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# Growth rates and mortality patterns of *Acacia mellifera* subsp. *detinens* in the semi-arid Highland Savanna, Namibia: Encroachment is not as rapid as previously believed

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## ABSTRACT

Perceptions regarding bush thickening in Namibia are rarely tested. It is often thought that bush thickening species such as *Acacia mellifera* subsp. *detinens* grow rapidly and recruit often. We estimated the growth rates and mortality patterns of 31 *A. mellifera* subsp. *detinens* shrubs in the arid Highland Savanna vegetation type in Namibia that were less than one metre tall in 1972, by remeasuring their heights in 2007. Some of these shrubs' heights were also remeasured in 1988. Growth rates were variable but on average individuals grew 3.19 cm in height per year. Growth was slower between 1972 and 1988 (2.67 cm per annum) than between 1988 and 2007 (3.85 cm per annum) for ten individuals remeasured in 1988. Based on these data, individuals of 1.5 m height are likely to be approximately half a century old, individuals of 2 m height 65 years old and individuals of 4 m height well over a century old. Mortality was high over the 35-year period (61.3%). Mortality between 1972 and 1988 was higher (45.2%) than between 1988 and 2007 (29.7%). These results support other studies that suggest that *A. mellifera* subsp. *detinens* is a very slow-growing species in the Highland Savanna, and that mature bush thickets observed today were mostly in existence as mature bush thickets half a century ago. *Acacia mellifera* subsp. *detinens* is a long-lived species, living for well over a century, but is susceptible to drought. Further implications of these findings are discussed.

**Keywords:** *Acacia mellifera* subsp. *detinens*; bush encroachment; drought; growth rate; Highland Savanna; mortality; Namibia; rainfall; recruitment; saplings; seedlings; vegetation

## INTRODUCTION

Much has been written about bush thickening globally, regionally and nationally (O'Connor et al. 2014), yet little empirical evidence is available to support several conflicting theories (Ward 2005, Joubert et al. 2008). For example, some researchers suggest that fire is necessary to stop the transition from a grassy state to a bushy state and that grass competition plays only a secondary role (Joubert et al. 2008, 2012). However, others believe that competition with grass is sufficient to reduce the growth of seedlings and saplings, and even kill them, provided that the perennial grass cover is high and the tufts are healthy (e.g. Ward & Esler 2011). There is also evidence to suggest that the increase in atmospheric carbon dioxide is a primary driver of bush thickening (Bond & Midgley 2012, Buitenwerf et al. 2012).

Recruitment is continuous in some species such as *Prosopis glandulosa* in semi-arid grasslands in Texas, USA (Brown & Archer 1999). However,

Watson et al. (1997) have demonstrated both episodic peaks in, and continuous recruitment of, two shrub species in arid rangelands in western Australia. In other species such as *Acacia (sensu lato) mellifera* subsp. *detinens* in semi-arid southern African savannas recruitment occurs episodically and infrequently, during periods of protracted above average annual rainfall (Joubert et al. 2008, 2013). Seymour (2008) showed episodic recruitment from seed, but a continuously recruiting sapling bank, for *A. erioloba* in an arid South African savanna. These observed responses in arid rangelands clearly show that the process of thickening does not follow a uniform pattern among all species and in all environments.

Little has been written on the growth rates of individuals of encroaching species. This is despite its importance in understanding encroachment rates and patterns and in improving predictions of sustainability for wood industries such as charcoal production and biomass energy plants. This short paper presents growth data of *A. mellifera* subsp. *detinens* from height measurements taken in 1972,

and repeated in 1988 and 2007, in the Highland Savanna. It also reports on the mortality rates of the measured individuals over the same period.

## METHODS

### Study Site

The Highland Savanna vegetation type (Giess 1998) has a mean annual rainfall of approximately 360 mm (CV=40%) in Windhoek. It is characterised by woody species including *A. hereroensis*, *A. hebeclada* subsp. *hebeclada*, *A. mellifera* subsp. *detinens*, *A. reficiens*, *Euclea undulata*, *Dombeya rotundifolia*, *Tarchonanthus camphoratus*, *Searsia marlothii*, *Albizia anthelmintica* and *Ozoroa crassinervia*. Valuable climax grasses include *Brachiaria nigropedata*, *Antheophora pubescens*, *Heteropogon contortus* and *Digitaria eriantha*, but *Eragrostis nindensis* (considered a subclimax grass) is usually the most abundant (Joubert 1997). On many farms, the climax grasses are largely absent, being replaced by *Michrochloa caffra* and various annual pioneer species including *Aristida adscensionis*.

This study was undertaken on the farm Sonnleiten which is located about 20 km east of Windhoek in the Highland Savanna vegetation type. The farm has been in existence for over a century, and over this period has been used primarily for cattle production. In the 1970s, Mr. Argo Rust, the farm owner, adopted the high intensity, short duration method of grazing, as prescribed by the principles of Holistic Resource Management (Savory 1991). Wildlife species include kudu (*Tragelaphus strepsiceros*), oryx (*Oryx gazella*) and springbok (*Antidorcas marsupialis*). Currently, the farm is largely encroached with *A. mellifera* thickets and very little perennial grass cover occurs.

Sonnleiten is approximately 5,000 ha in extent, which is a typical size for a commercial farm in the Highland Savanna. The farm is characterised by steep mountainous slopes that are covered by lithic leptosols, typically <30 cm in depth, with a high proportion of schist and quartz gravel, as well as deeper eutric leptosol pediments below these slopes (Mendelsohn et al. 2002).

### Data Collection

Rainfall data were obtained from Neudamm Agricultural Office, approximately 5 km from the study site on Sonnleiten Farm.

In 1972, four permanent circular plots with a radius of 5 m (78.54 m<sup>2</sup>) each were established on Sonnleiten by Argo Rust. Within each plot, all woody, perennial plants were mapped and their heights measured. In 1988, one of the plots, which

contained 10 surviving individuals of *A. mellifera* subsp. *detinens*, was remeasured. By November 2007, a total of 31 individuals remained alive in the four plots and these were remeasured. Although height is not as good an indicator of biomass growth as are other parameters (Hasen-Yusuf et al. 2013), the continued use of height was necessary for comparisons with 1972. Mean annual growth rates were determined from the height data. Mortality was also recorded. It was possible to determine the growth rates and mortality patterns for two periods for ten of these trees (Period 1: 1972-1988 [16 years]; Period 2: 1988-2007 [19 years]). The difference in growth rates during these two periods was tested for significance using the Wilcoxon Matched Pairs Test (Statsoft 2004) to account for non-normally distributed data. The mean sizes of the ten trees that were measured in all three years (1972, 1988, 2007) were fitted to a linear model (Statsoft 2004) in order to get an approximate establishment time for the individuals, and a prediction for future size of individuals in 25 years.

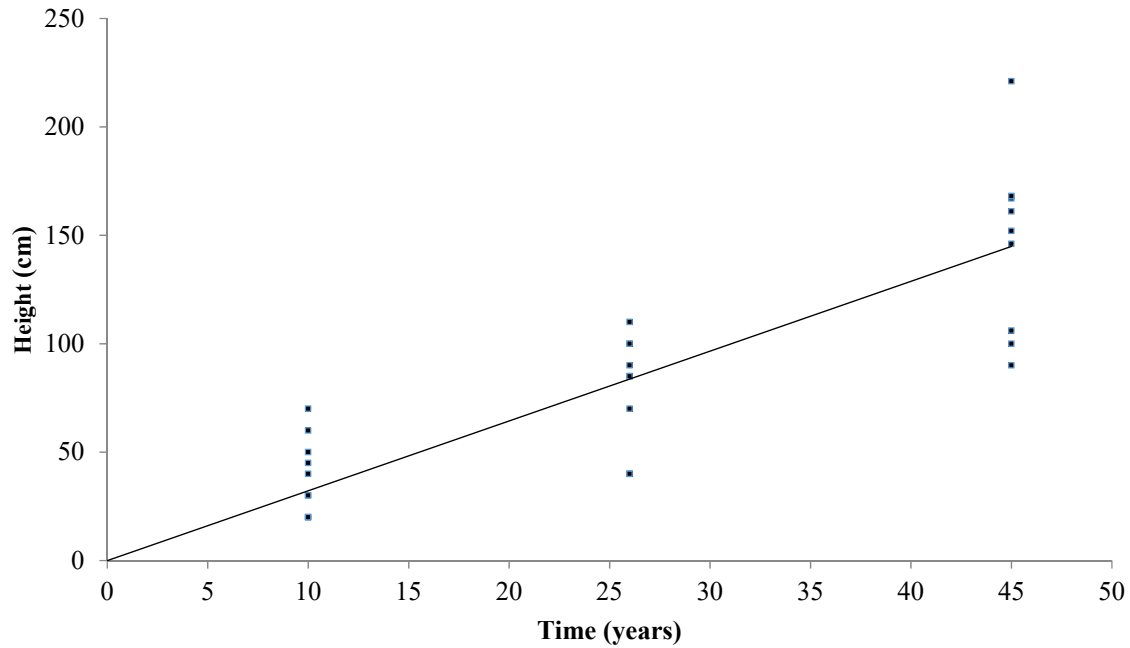
## RESULTS AND DISCUSSION

The data show a very slow increase in height of *A. mellifera* subsp. *detinens* shrubs over 35 years (Table 1). The maximum growth rate of just over 7 cm / annum was for an individual growing in an area of minimal competition, on a pediment soil that was much deeper than in the other areas. The next highest growth rate was 4.6 cm / annum. The mean annual growth rate for the ten individuals remeasured in 1988 was significantly lower in the first period (1972-1988) than in the second period of measured growth (1988-2007) (Table 1). The absolute growth rate of larger and older individuals would tend to be greater than smaller and younger individuals. None of the individuals was judged to be senescent.

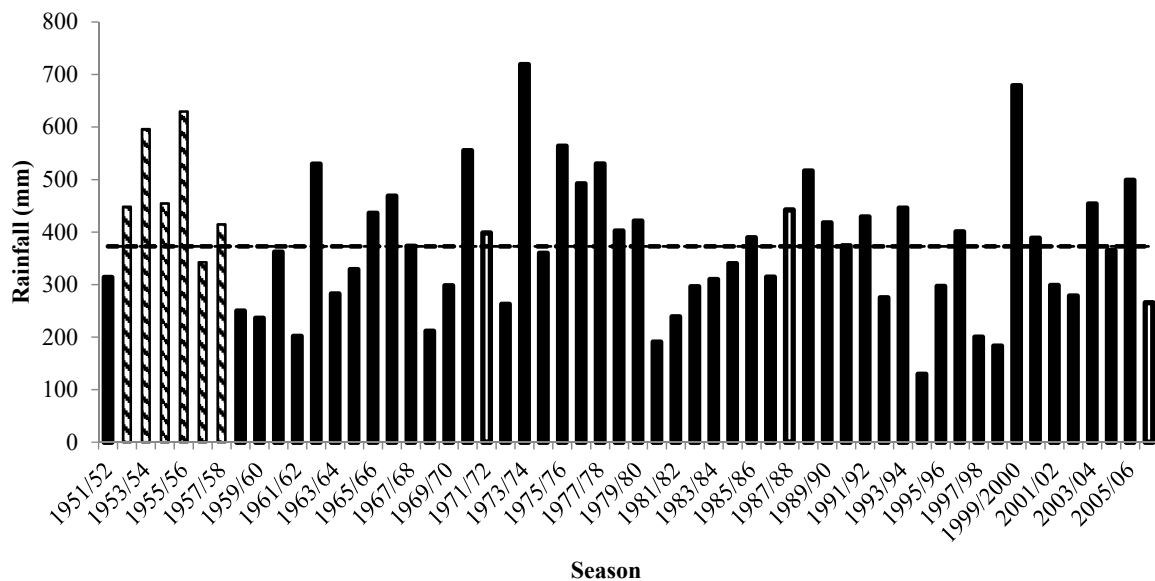
Figure 1 shows the change in height of the ten individuals measured at all three time-steps (1972, 1988, 2007). Extrapolation of the regression equation suggests that for saplings that were measured in 1972

**Table 1:** Annual growth rates of *Acacia mellifera* subsp. *detinens* shrubs between 1972 and 2007. Size range 1972 = 20-100 cm; 2007 = 106-288 cm

Period	N	Mean (cm)	Min (cm)	Max (cm)
1972-2007 (35 years)	16	3.19 (± 1.50)	0.95	7.09
1972-2007 (35 years)	10	3.12 (± 0.94)	1.43	4.6
1972-1988 (16 years)	10	2.34 (± 1.38)	-0.63	3.75
1988-2007 (19 years)	10	3.78 (± 1.05)	2.63	6.37



**Figure 1:** Regression line fitting the heights of ten *Acacia mellifera subsp. detinens* shrubs between 1972, 1988 and 2007. The equation is Stem height (cm) = 3.219 x years. The adjusted  $R^2$  value is 0.712. Some individuals have the same values which is why fewer than ten data points are illustrated at each time step.



**Figure 2:** Rainfall data from Neudamm Agricultural College from 1951-1952 to 2006-2007. The patterned bars denote the rainfall during the likely time of establishment of the cohort of plants measured. Open bars denote the rainfall at the time the plants were first measured (1972), measured again (1988) and for the last time (2007).

germination occurred ten years previously (i.e. in 1962), assuming a linear pattern in height growth. However, germination of the 31 individuals measured is much more likely to have occurred between 1952 and 1958, when there were two (1952-1953 and 1953-1954) and three (1955-1956; 1956-1957 and 1957-1958) successive seasons of above average rainfall (Figure 2).

Recruitment is dependent upon rare, protracted above average rainfall periods (Joubert et al. 2008, 2013), as was experienced in the 1950s (Figure 2). If this is the case, then the saplings measured in 1972 would have been between 14 and 19 years old. Saplings that germinated in 2001 on Krumhuk farm nearby were typically less than 10 cm high after six years of growth, with mean stem diameters of around 5 mm



(Joubert et al. 2013). However, seedlings at Neudamm College, germinating in 2007, grew faster (Joubert 2014), suggesting large variability in early growth rates, dependent upon rainfall. The growth rates after recruitment are likely to be slower in absolute terms, but variable, and highly dependent upon soil moisture. It is likely that the individuals measured at Sonnleiten were between 49 and 54 years old in 2007. Shrubs between 1 m and 2 m are typically believed by rangeland managers to be around 10 years old. This misconception could be due to: i) the observation of trees with faster growth rates in micro-environments where water accumulates and ii) the rapid growth rate of resprouts from the stumps of trees either cut or burnt or suckering from lateral roots after extreme disturbance, for example in borrow pits. Forecasting with the regression model shown in Figure 1 suggests that the individuals would only reach a height of approximately 2 m at approximately 65 years. Growth rates measured at Sonnleiten, however, are from individuals growing in shallow soils under more arid conditions than many areas where encroachment is a more serious problem. For example, heavily encroached areas in Thornbush Savanna typically have >400 mm annual rainfall and are characterised by having deeper soils with a higher clay content than are found at the study site at Sonnleiten. Thus, growth rates in these areas of higher rainfall and finer and deeper soils are more likely to be closer to the maximum (7.09 cm per annum) measured for a single individual growing on deeper soils at Sonnleiten.

The mortality rates observed in this study (Table 2) were surprisingly high. It is not known how much of the mortality was due to self-thinning, but drought stress and fungal dieback are likely to have contributed greatly. Mortality in the first period was much higher than in the second period. Unpublished data (Joubert) and farmer knowledge suggest that sapling mortality can be very high, particularly in dry years. There were five consecutive below average rainfall years (1980-1981 to 1984-1985) in the first period, as opposed to three periods of two consecutive years of below average rainfall (1994-1995 to 1995-1996; 1998-1989 to 1999-2000 and 2001-2002 to 2002-2003). These protracted periods of below average rainfall are likely to have been the major cause of mortality.

The slow growth rates shown here, the high mortality rates, and the rareness of recruitment events, as shown in other studies (Joubert et al. 2008) support the authors' view that woody plant encroachment is generally a much slower process than previously supposed. Joubert (2014), using aerial photos, showed that a 25-50% increase in cover occurred in the 50-year span between 1958 and 2007 on Sonnleiten Farm. However, the mean increase on

**Table 2:** Mortality patterns of 31 *Acacia mellifera* subsp. *detinens* shrubs on Sonnleiten Farm.

Time period	Number of individuals or %
Population in 1972	31
Survivors from 1972 in 2007	12
Died by 1988	14
Died between 1988 and 2007	5
Mortality overall	61.3%
Mortality 1972 to 1988	45.2%
Mortality 1988 to 2007	29.4%

three farms studied was 14-26% and some mature thickets decreased by 17-35% during this time.

In another study using matched photography spanning 130 years, Rohde and Hoffman (2011) found an overall increase in woody cover of 1.5% (range -5% to 15%) for shrubs <1.5 m and 22% (range -5% to 48%) for trees >1.5 m in what they termed "tree and shrub savannas" which broadly corresponds with the Highland Savanna in this study. This would be viewed as surprisingly modest by most rangeland managers and scientists in Namibia today. The current perception that existing thickets of large trees were initiated during a drought period coinciding with a foot-and-mouth outbreak in the late 1950s is disproven by these observed slow growth rates of species. Based on this study, thickets with 4 m trees are likely to have established more than a century ago.

Growth rates in deeper soils, with higher clay and nutrient contents, and in higher rainfall areas are likely to be significantly higher, as is reflected by the higher growth rate of the individual growing in deeper soils in this study (more than double the mean). Joubert et al. (2013) show that seedling and sapling growth, in the first six years, is much slower than the growth of the mature shrubs in this study.

## CONCLUSION

*Acacia mellifera* subsp. *detinens* is a slow-growing species, contrary to current popular perceptions of the species being an "aggressive invader". Combining this with the fact that recruitment is highly episodic (Joubert et al. 2008, 2013), it appears that preventative management of bush thickening by this species is easier than was previously supposed. Currently observed thickets of mature *A. mellifera* subsp. *detinens* trees are likely to be much older than previously thought. The perception that bush thickening is the single most important factor explaining the perceived decline in rangeland productivity since the late 1950s (De Klerk 2004) is challenged by this study, and other recent studies (e.g. Rohde & Hoffman 2011, Joubert 2014).

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# Oxpeckers in Namibia: A review of their status and distribution in 2017

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## ABSTRACT

Red-billed Oxpecker, *Buphagus erythrorhynchus*, and Yellow-billed Oxpecker, *B. africanus*, occur in Namibia with populations restricted to the extreme north-eastern parts (both species) and north-western parts of the country (Yellow-billed Oxpecker). The Red Data status of Red-billed Oxpecker in Namibia is Rare or Peripheral while Yellow-billed Oxpecker is considered Endangered. Assessments of the status of Namibian populations were made in 1983, 1984-85 and 1997-98. The aim of this study was to determine if their status and distribution had changed since the previous assessments and to ascertain the range of hosts used by the two species. The Red-billed Oxpecker population appears to have remained unchanged in terms of both numbers of birds (around 3,600 birds) and distribution. Cattle remain their most important host although a range of other hosts are used, including 10 game species, two equine species and two livestock species. The Yellow-billed Oxpecker population in the north-east may have increased but remains low (at around 2,300 birds) and there is evidence to suggest that its range has contracted. For the Yellow-billed Oxpecker in the north-east, the primary hosts are cattle and buffalo, *Syncerus caffer*, with these birds using seven game species, one equine species and one livestock species. In the north-west, Yellow-billed Oxpecker appears to have expanded its range and, as no assessment of the numbers of birds was made here, it is suggested that future efforts focus on determining population estimates for this population which was previously considered to be marginal in Namibia.

**Keywords:** *Buphagus africanus*; *Buphagus erythrorhynchus*; distribution; host species; Namibia, oxpecker; population status

## INTRODUCTION

Oxpeckers feed primarily on ectoparasites, mucus and necrotic tissue present on ungulate hosts, and previous studies have recorded birds on a wide range of ungulate species (Stutterheim 1981, Stutterheim & Panagis 1985a, Hustler 1987, Hart et al. 1990, Mooring & Mundy 1996, Koenig 1997).

In Namibia, oxpeckers are currently limited to the extreme north-eastern and north-western parts of the country, although their historical distribution was much broader. The Red Data status of Yellow-billed Oxpecker, *Buphagus africanus*, in Namibia is listed as Endangered while Red-billed Oxpecker, *B. erythrorhynchus*, is considered Rare and Peripheral (Simmons et al. 2015). Both species of oxpecker are found in Kavango East, eastern Kavango West and Zambezi regions, while the Yellow-billed Oxpecker is also found in Kunene Region.

Declines in bird numbers and their distribution in southern Africa since the mid-1900s have been largely attributed to the use of arsenic-based cattle dipping schemes (Stutterheim & Brooke 1981,

Stutterheim 1982). These methods of parasite control have since been replaced in Namibia. A recent reintroduction was undertaken in Otavi district in Namibia where 15 Red-billed Oxpeckers were released in an attempt to re-establish this species in part of its former range (Nebe & Nebe 2015). In 2017 the first case of successful breeding was recorded (B Nebe pers com).

Assessments of the status of Red-billed Oxpecker and Yellow-billed Oxpecker in Namibia were undertaken in 1983 (Brown & Brown 1987), 1984-85 (Stutterheim & Panagis 1985b), and 1997-98 (Robertson & Jarvis 2000). These studies dealt specifically with oxpeckers in the Zambezi region and predominantly with their presence in relation to cattle hosts. No recent oxpecker specific surveys in Namibia have been published, however some indication of relative abundance and distribution is available from the ongoing Southern African Bird Atlas Programme (SABAP) run by the Animal Demography Unit at the University of Cape Town (Harrison et. al. 1997, <http://sabap2.adu.org.za>).

In this study we report on surveys of both species of oxpecker, carried out in Kavango East and Zambezi regions (hereafter referred to as the north-east) and Kunene region (hereafter referred to as the north-west) in Namibia between 2010 and 2017. We present results from two types of counts and (a) assess host preferences between the two species, (b) provide estimates of bird numbers across the main host species, and (c) map the current distributions of each oxpecker species, comparing these with both SABAP1 (Mundy 1997a, 1997b) and SABAP2 distribution maps (<http://sabap2.adu.org.za>).

## METHODS

### Sampling

Between 2010 and 2017 field records were collected whilst driving to and from locations in Kavango East and Zambezi regions and Kunene region. In total, 49 counts were undertaken, comprising both *casual* counts (29 counts) where only host groups that contained oxpeckers were counted and *inclusive* counts (20 counts in the north-east) where all potential hosts and any oxpeckers present were counted. In the north-east, inclusive counts were undertaken across the entire Zambezi Region (former Caprivi strip), both inside and outside the state protected areas and including all game and domestic host species. Inclusive counts are important for the determination of oxpecker:host ratios which can be applied to host population figures to estimate bird numbers across an area. Inclusive counts were undertaken along the margins of the Kwando and Okavango rivers in 2010-2013, and along all the main road networks of eastern Zambezi region in 2017.

While casual counts alone cannot provide population estimates because they do not include all potential host animals, when combined with inclusive counts they allow a more robust assessment to be made of:

- 1) the range of host species used by each oxpecker species (with an indication of preference), and
- 2) the spatial distributions of the two species.

In the north-west of the country, only casual counts were made across an area ranging from the western boundary of Etosha National Park to Namibia's northern boundary with Angola. For all counts, host animals and oxpeckers were counted and the following details were recorded: date, time, GPS coordinates, host species, number of host animals, and number of oxpeckers of each species. For inclusive counts, the start and stop GPS coordinates were also recorded.

### Analysis

Data were summarised for each oxpecker species separately, for both inclusive and casual counts. Data were tabulated to tally host groups, host numbers and

bird numbers for each game and domestic host species.

For inclusive counts bird:host ratios were determined. As inclusive counts were only undertaken in the north-east, population estimates for each bird species could only be obtained for that area. Counts between 2010 and 2013 were primarily undertaken in wildlife areas proximal to the Okavango and Kwando river systems. These counts were therefore amalgamated and used to derive bird:host ratios on game species. In 2017 a comprehensive inclusive count along approximately the same road network counted in previous studies (Brown & Brown 1987, Sutterheim & Panagis 1985b, Robertson & Jarvis 2000) was conducted with the primary aim to determine bird presence and bird:host ratios on livestock.

The most recently available data on cattle numbers in Zambezi Region and East Kavango were obtained from the Department of Veterinary Services. Estimated numbers of wildlife hosts were obtained from numbers of wildlife seen during the Zambezi game count of 2016 (<http://www.nacso.org.na>).

All counts (both casual and inclusive) were used to map the presence of each bird species at the resolution of monad (1 minute x 1 minute) grid squares. Data from both the SABAP1 and SABAP2 programmes were compiled for purposes of comparison, however these data were only available at a coarser level of resolution i.e. quarter degree squares for SABAP1 and pentad squares (5 minute x 5 minute) for SABAP2.

## RESULTS

### Oxpeckers and host species

In the north-east a total of 969 Red-billed Oxpeckers was recorded on 307 host groups and 739 Yellow-billed Oxpeckers were recorded on 142 host groups. In the north-west, casual counts recorded 198 Yellow-billed Oxpeckers on 61 host groups (Table 1).

### Host utilisation and preference - North-east

Red-billed Oxpecker was recorded on 10 game species, two equine species (donkey, horse) and two livestock species (cattle, goat). In both casual and inclusive counts kudu, impala and buffalo were the most common game hosts (Table 2a, 2b, scientific names of game species are given in Table 2a). Cattle were the most important domestic livestock hosts, followed by donkeys (Table 2c).

Yellow-billed Oxpecker was recorded on seven game species, one equine species (donkey) and one livestock species (cattle). Both buffalo and kudu

accounted for most birds on game hosts in casual counts (Table 2a) although birds were detected only on buffalo in inclusive counts despite a large number of kudu groups (85) being recorded (Table 2b). As with Red-billed Oxpecker, cattle were the most important livestock hosts, with a few birds being recorded on donkeys (Table 2c).

### Host utilisation and preference - North-west

Only Yellow-billed Oxpecker is present in the north-west and it was only recorded on equine and bovine hosts (horse, donkey and cattle) (Table 3). No birds were observed on game animals (Table 4) despite the fact that substantial numbers of a wide range of potential game host species were observed each year.

**Table 1:** Count summary: number of host groups and numbers of Red-billed and Yellow-billed Oxpeckers recorded between 2010 and 2017, in the north-east and north-west of Namibia.

Year	Total number of Host Groups	North-East				North-West	
		Red-billed Oxpecker		Yellow-billed Oxpecker		Yellow-billed Oxpecker	
		Host Groups with birds	Number of birds	Host Groups with birds	Number of birds	Host Groups with birds	Number of birds
2010	189	56	182	28	133		
2011	185	38	114	11	143	21	35
2012	146	105	350	52	196	15	45
2013	373	65	202	27	172	10	32
2014	27	15	50	13	42	1	5
2015	13	6	23	8	28	14	81
2017	178	22	48	3	25		
	<b>1111</b>	<b>307</b>	<b>969</b>	<b>142</b>	<b>739</b>	<b>61</b>	<b>198</b>

**Table 2a:** Number of hosts, host groups and number of associated Red-billed and Yellow-billed Oxpeckers recorded during casual counts in the north-east of Namibia between 2010 and 2017.

Total Hosts				RBO where present			YBO where present		
Host	Total Groups	Total Hosts	Avg. Group size	Host Groups	Hosts	RBOs	Host Groups	Hosts	YBOs
Roan antelope <i>Hippotragus equinus</i>	7	34	5	5	20	23	3	20	10
Giraffe <i>Giraffa camelopardalis</i>	1	7	7	1	7	8			
Hippo <i>Hippopotamus amphibius</i>	3	20	7	3	20	14	1	4	1
Sable antelope <i>Hippotragus niger</i>	6	82	14	4	56	16	2	26	6
Tsessebe <i>Damaliscus lunatus</i>	1	6	6	1	6	1			
Warthog <i>Phacochoerus africanus</i>	5	23	5	3	12	5	2	11	6
Zebra <i>Equus quagga</i>	4	50	12	4	50	12			
Kudu <i>Tragelaphus strepsiceros</i>	52	280	5	39	198	110	15	94	57
Buffalo <i>Syncerus caffer</i>	20	1159	58	6	159	31	15	1059	188
Impala <i>Aepycerus melampus</i>	14	332	24	14	332	42	1	41	2
	<b>113</b>	<b>1993</b>	<b>143</b>	<b>80</b>	<b>860</b>	<b>262</b>	<b>39</b>	<b>1255</b>	<b>270</b>
Horse	2	4	2	2	4	9			
Donkey	15	44	3	11	34	40	5	13	15
Goat	3	50	17	3	50	7			
Cattle	217	4406	20	157	2988	487	85	2130	303
	<b>237</b>	<b>4504</b>	<b>42</b>	<b>173</b>	<b>3076</b>	<b>543</b>	<b>90</b>	<b>2143</b>	<b>318</b>

**Table 2b:** Number of game hosts and groups and number of associated Red-billed and Yellow-billed Oxpeckers recorded during inclusive counts in the north-east of Namibia between 2010 and 2013.

Total Hosts				RBO where present			YBO where present			Birds per host	
Host	Total Groups	Total Hosts	Avg. Group size	Host Groups	Hosts	RBOs	Host Groups	Hosts	YBOs	RBO	YBO
Roan	6	25	4				1	11	12		0.48
Giraffe	8	32	4	2	9	8				0.25	
Hippo	11	33	3								
Sable	10	38	4	1	6	2	1	1	2	0.0526	0.0526
Tsessebe	8	49	6								
Warthog	30	86	3								
Zebra	16	162	10								
Kudu	85	316	4	7	37	16				0.0506	
Buffalo	27	834	31	2	102	11	4	377	91	0.0132	0.1091
Impala	154	1706	11	3	62	10				0.0059	

**Table 2c:** Number of livestock hosts and groups, and number of associated Red-billed and Yellow-billed Oxpeckers recorded during inclusive counts in the north-east of Namibia in 2017.

Total Hosts				RBO where present			YBO where present			Birds per host	
Host	Total Groups	Total Hosts	Avg. Group size	Host Groups	Hosts	RBOs	Host Groups	Hosts	YBOs	RBO	YBO
Goat	34	540	16								
Cattle	139	2079	15	22	344	48	3	76	25	0.0231	0.012

The average group size of both species of oxpeckers on different host species ranged from 1 to 5.5, with the exception of Yellow-billed Oxpeckers on buffalo, which averaged about 15. This high average was influenced by a group of 70 Yellow-billed Oxpeckers recorded on a herd of buffalo in the north-east (Table 4).

#### Estimates of bird numbers in the north-east

By using known or estimated host numbers, and applying the bird:host ratios obtained from the inclusive counts, we derived population estimates for both oxpecker species. Buffalo are transient in the north-east and associated with the perennial water courses. Their numbers in Namibia vary over time as they move to and from neighbouring countries. Wetland counts undertaken between 2004 and 2014 along the Kwando river showed a range in animals seen from 1,297 – 6,556 over four counts (Du Preez et al. 2015). The number of 3,173 buffalo seen in the 2016 Zambezi game count falls in the middle of these values. Cattle numbers in the north-east have remained relatively stable over the long term but show some fluctuation between years. In order to derive a typical number for cattle in the north-east we

used an average of values available for Zambezi and Kavango East during the time span of this study (Table 5).

Where available, bird:host ratios for each game species and cattle were applied to derive estimates of Red-billed Oxpecker and Yellow-billed Oxpecker populations in the north-east (Table 6).

It is estimated that there are at least approximately 3,600 Red-billed Oxpeckers in the north-east with the vast majority of these birds being associated with cattle (Table 6). The estimate for Yellow-billed Oxpecker in the north-east is around 2,290. In contrast to Red-billed Oxpecker there are around half

**Table 3:** Number of livestock hosts and groups, and number of associated Yellow-billed Oxpeckers recorded during casual counts in the north-west of Namibia between 2010 and 2015.

Host	Where birds were present		
	Host Groups	Host Numbers	YBO
Donkey	21	81	58
Horse	2	10	8
Cattle	36	412	125

**Table 4:** Range in group size and average group size of oxpeckers on all host species in the north-east and north-west of Namibia between 2010 and 2017.

Host	North-East				North-West	
	Red-billed Oxpecker		Yellow-billed Oxpecker		Yellow-billed Oxpecker	
	Range	Average	Range	Average	Range	Average
Roan antelope	2-12	4.6	2-12	5.5		
Giraffe	3-8	5.3				
Hippo	4-6	4.7	1	1.0		
Sable antelope	1-7	3.6	1-5	2.7		
Tsessebe	1-1	1.0				
Warthog	1-2	1.7	1-5	3.0		
Zebra	1-9	3.0				
Kudu	1-10	2.7	1-13	3.8		
Buffalo	1-12	5.3	2-70	14.7		
Impala	1-7	3.1	2	2.0		
Horse	3-6	4.5			4	4.0
Donkey	1-10	3.6	1-5	3	1-8	2.8
Goat	1-4	2.3				
Cattle	1-12	3.0	1-19	3.8	1-28	3.5

**Table 5:** Cattle numbers in north-east Namibia between 2010 and 2014. Data were provided by the Department of Veterinary Services, Ministry of Agriculture, Water and Forestry, Namibia.

Year	Cattle	Sheep	Goat	Horse	Donkey
2014	162,007	15	4,812	12	31
2013	141,634	35	3,703	0	103
2012	136,603	161	11,589	15	267
2010	152,823	163	10,443	12	117
Average	148,270	90	7,640	10	130

**Table 6:** Current estimates of Red-billed and Yellow-billed Oxpecker numbers on game and cattle in north-east Namibia.

Hosts		Red-billed Oxpecker		Yellow-billed Oxpecker**	
Game*	Animals	Birds per host	Estimate	Birds per host	Estimate
Roan	263			0.48	126
Giraffe	120	0.25	30		
Sable	691	0.0526	36	0.0526	36
Kudu	891	0.0506	45		
Buffalo	3,173	0.0132	42	0.1091	346
Impala	2,183	0.0059	13		
			166		508
Livestock					
Cattle	148,270	0.0231	3,425	0.012	1779
Overall			3591		2287

\* Numbers for game species are actual numbers sighted in transect counts across the region and are therefore an underestimate of available hosts. The numbers of oxpeckers on game animals are therefore likely to be higher than the estimates presented here.

\*\* The ratio for Yellow-billed Oxpeckers was derived from bird sightings on only three host groups.

**Table 7:** Comparison of current and historical bird:host ratios and estimates for Red-billed and Yellow-billed Oxpecker on cattle hosts in the north-east of Namibia.

	1983-84 <sup>a</sup>		1997-98 <sup>b</sup>		2017	
	Bird:host	Estimate	Bird:host	Estimate	Bird:host	Estimate
RBO	0.0519-0.0845	2285-3780	0.0293-0.0396	3627-4902	0.0231	3,425
YBO	0.0461-0.0491	2062-2200	0.0027*	334	0.012**	1,779

<sup>a</sup> From Brown & Brown 1987 and Stutterheim & Panagis 1985b. Estimates were derived in Robertson & Jarvis 2000.

<sup>b</sup> From Robertson & Jarvis 2000

\* Ratio derived from sightings of YBO on only six groups over three counts

\*\* Ratio derived from only three bird groups one of which contained 16 birds. This ratio should be considered with caution.

as many Yellow-billed Oxpeckers on cattle/livestock and the greater importance of game hosts, particularly buffalo, kudu (from casual counts), roan and sable antelope to this oxpecker species is evident.

A comparison of past bird:host ratios on cattle (Table 7) suggests a gradual reduction since the 1980s for Red-billed Oxpecker although the gradual increase and then stabilisation of cattle numbers in the region has had the effect of keeping the oxpecker population relatively constant. Relative to the Red-billed Oxpecker the sighting frequency of Yellow-billed Oxpecker on cattle remains low. In the three counts done by Robertson and Jarvis in 1997 and 1998 Yellow-billed Oxpecker was only observed on one, three and two host groups respectively. In the current count oxpeckers were only observed on three cattle groups.

### Oxpecker distributions

Red-billed Oxpecker was observed throughout both the western and eastern parts of the north-east and in the multiple-use zone of Bwabwata National Park (Figure 1). Its distribution has shown effectively no change since the first bird atlas programme was concluded in the mid-1990s and our sightings are similar in distribution to those of the ongoing SABAP2 programme. Indications are that the range and size of this population has remained stable over time.

Yellow-billed Oxpecker was observed primarily along the Kwando River system and in close association with the Chobe and Linyanti river channels (Figure 2). There were only a few observations of birds in the western parts of Zambezi Region. The distribution of sightings in this study matches the general pattern of the SABAP1

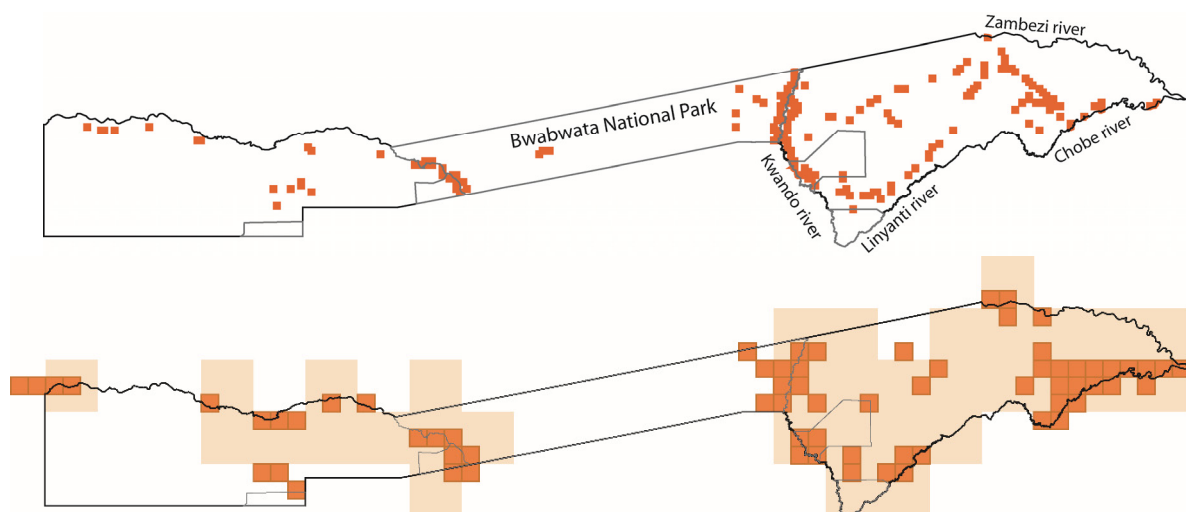
programme however it is evident that there has been a reduction in numbers of birds in the section of the region east from where the Zambezi River forms the northern border of the country. This loss is also evident from the sightings recorded by the ongoing SABAP2 programme.

In the north-west of Namibia, where only Yellow-billed Oxpecker occurs, the distribution of birds has shown a dramatic expansion (Figure 3). Sightings were made as far south as Puros Conservancy in the west and in Ehi-Rovipuka Conservancy in the east. This is in sharp contrast to the distribution recorded by the SABAP1 programme where birds were limited to the margins of the Kunene River and northern border of the country east of Ruacana. Sightings from SABAP2 programme match areas recorded in SABAP1 but also show an expansion in range previously not recorded.

### DISCUSSION

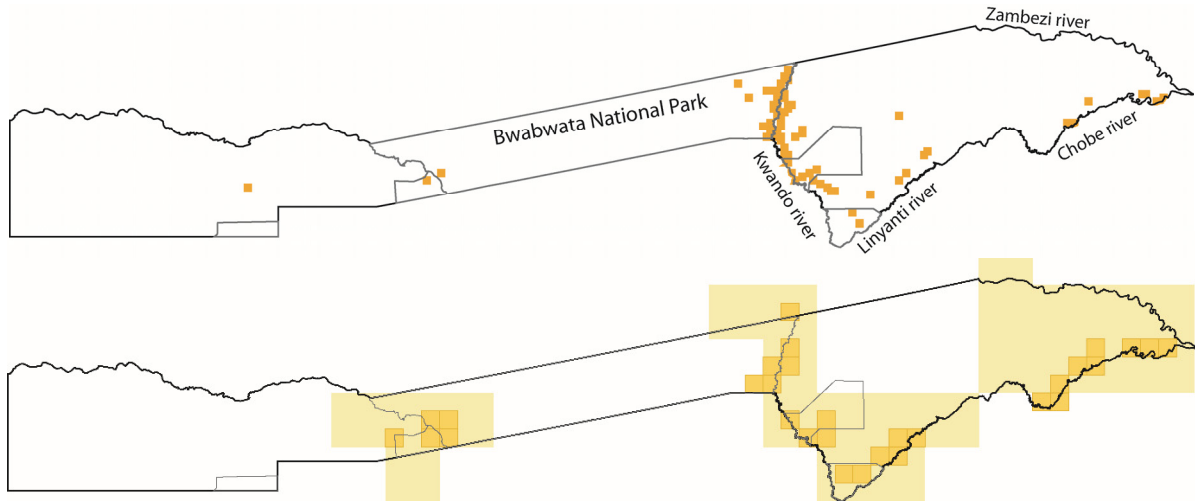
Host preferences on game species observed from this work concur with those found previously in Hwange National Park, Zimbabwe (Hustler 1987) and in Moremi Game Reserve, Botswana (Stutterheim & Panagis 1985a) and with a study of tick yields on game animals in Kenya (Hart *et al.* 1990).

The Red-billed Oxpecker population in the north-east appears to have remained stable. While bird:host ratios have shown a small decline our estimates suggest that there are similar numbers of birds present now as there were in the 1990s. Cattle remain the most important host for these birds however they do also use a broad variety of game hosts. They are distributed widely throughout the region and there is no evidence that their range has contracted.



**Figure 1:** Top: sightings of Red-billed Oxpecker in this study 2010-2017, bottom: sightings from SABAP1 (large blocks) and SABAP2 (small blocks).





**Figure 2:** Top: sightings of Yellow-billed Oxpecker in this study 2010-2017, bottom: sightings from SABAP1 (large blocks) and SABAP2 (small blocks).



**Figure 3:** Top: sightings of Yellow-billed Oxpecker in this study 2010-2015, bottom: sightings from SABAP1 (large blocks) and SABAP2 (small blocks).

The Yellow-billed Oxpecker population in the north-east remains precarious. While cattle are an important host for these birds the bird:host ratio remains low. The estimates presented here should be considered with much caution. The restricted range of these birds with their greater apparent association with river systems suggests an alternative approach should be used to assess population size more accurately for these birds, perhaps based on zonation. A limited range of game species, in particular buffalo, are important for these birds and they host a significant proportion of the Yellow-billed Oxpecker population.

The Yellow-billed Oxpecker population in the north-west has shown a significant expansion of its range since the 1990s. This population was previously considered marginal in Namibia. However, it has now become more widespread and a series of *inclusive* counts are required throughout the north-west to derive robust bird:host ratios on livestock and game species for these birds, using a stratification based on livestock density.

The occurrence of Yellow-billed Oxpecker in the wetter north-east and the arid north-west provides an opportunity for further research to try to explain the factors limiting populations of this species in these quite different habitats. More detailed studies of feeding and breeding behavior, perhaps linked to climatic conditions and flooding of the large river systems of both the north-western and north-eastern populations, may provide useful insights into the drivers behind population growth or decline in Namibia for this endangered species.

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# Vegetation of the Auas-Oanob Conservancy in the Khomas Hochland of Namibia

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## ABSTRACT

The Auas-Oanob Conservancy is situated in the very rugged, botanically highly diverse Khomas Hochland in central Namibia. A number of vegetation-related studies have been undertaken in this highland before, but none covering the full extent of the conservancy, and with different interpretations of the syntaxonomy of the vegetation. The current study aimed to describe and map the vegetation of the conservancy for practical management purposes. A total of 229 relevés was compiled within the conservancy, and 72 relevés from other studies (falling either within the conservancy, or the nearby Auas Mountain range) were added. The data were classified with a Modified TWINSpan. Three large groups were identified: (a) the high-altitude veld of the Auas Mountain range, with three associations being recognised on the basis of altitude and aspect; (b) the lowlands and valley veld with five associations, and (c) the Khomas Hochland veld with five associations. These 13 associations are described, mapped and compared to descriptions of the vegetation in the vicinity of the conservancy. A higher-order syntaxonomy, with three orders, one subdivided into two alliances, is suggested. This higher-order syntaxonomy needs to be further investigated, taking into account vegetation descriptions of the greater Khomas Hochland. The high-altitude veld of the Auas Mountain range has a unique composition with a high degree of endemism and a sub-alpine character. Due to its restricted range, there is an urgent need for formal protection. Two associations within the lowland and valley veld are under threat, being the *Platycarphella carlinoides*—*Chloris virgata* association of the *omiramba* (erosion) and the *Schmidtia kalahariensis*—*Acacia erioloba* association in the bottom lands at Omeya (development). Bush encroachment is a general threat to various upland associations.

**Keywords:** Auas Mountains; freehold conservancies; Highland Savanna; Khomas Hochland; phytosociology; vegetation classification; Namibia

## INTRODUCTION

The Auas-Oanob Conservancy consists of a cluster of freehold (commercial) farms south and south-west of Windhoek (Figure 1a), with a total area of 105,644 ha (Shaw & Marker 2010). The farms are used mainly for cattle farming, but horse breeding, game farming, hunting and ecotourism are also practised. In forming a freehold conservancy, the owners subscribe to the principle of “...a legally protected area of a group of land-occupiers practicing co-operative management based on a sustainable utilisation strategy, promotion of the conservation of natural resources and wildlife, and the desire to reinstate the original biodiversity with the basic goal of sharing resources amongst all members” (Shaw & Marker 2010). With more than 88% of all wildlife in Namibia occurring outside national parks, it was deemed necessary to form such conservancies to aid in the protection and sustainable utilisation of this renewable natural resource (NACSO 2010, Shaw & Marker 2010).

The Auas-Oanob Conservancy is situated in the *Khomas Hochland* (‘Khomas Highland’, often referred to as the ‘Central Highlands’) in central Namibia. This very rugged, high altitude (reaching 2,000 m asl) landscape was formed through extensive faulting and erosion of the soft schists of the Damara Sequence, and forms a broad south-west to north-east oriented band through central Namibia (Swart & Marais 2009). The ruggedness of the landscape gives rise to a plethora of habitats, which, in turn, result in a high diversity of plant and animal species (Craven 2001, Mendelsohn et al. 2002). Giess (1998) referred to the vegetation of this landscape as the ‘Highland savanna’.

To date, three studies have been undertaken to describe portions of this Highland savanna vegetation type: (a) Volk and Leippert (1971) studied the farms Binsenheim and Voigtland to the south-east and east of Windhoek; (b) Kellner (1986) reported on the Daan Viljoen Game Reserve, as well as portions of the farms Claratal and Bergvlug (west, south-west and east of Windhoek) and (c) Burke and Wittneben

(2007) investigated the Auas Mountain range south of Windhoek. Burke and Wittneben (2007) give an overview of composition and structure of the high-altitude Auas Mountain range vegetation, but provide no formal classification. Both Volk and Leippert (1971) and Kellner (1986) provide phytosociological descriptions of the vegetation at lower elevations in similar habitats, however, they differ in interpretation and syntaxonomy of the vegetation. Kellner (1986) describes the vegetation of the northern part of the farm Claratal as dominated by the *Acacia hereroensis*—*Brachiaria nigropedata*<sup>1</sup> savanna. This is the only part of the Auas-Oanob Conservancy that has an existing description of the vegetation.

This study aims at describing and mapping the vegetation of the entire Auas-Oanob Conservancy in order to serve as baseline information for the management of this important resource.

## STUDY AREA

### Topography and geology

The Lichtenstein Mountains, which form a southern extension of the Auas Mountain range, dominate the landscape of the central-eastern part of the conservancy. The highest peaks of the Lichtenstein mountains rise to 2,400 m, whereas the Molkteblick, as highest peak in the Auas Mountains, rises to 2,450 m (Jarvis et al. 2008). To the east, these taper out into the Omeya valley, which is closed in the far south-east of the study area by the Oamites Mountain.

To the west of the Lichtenstein Mountains, an undulating plateau forms the upper catchment of the Oanob River. This catchment is characterised by a number of *omiramba* (flat watercourses without discernable gradients - King 1963, Strohbach 2008), which feed into the endoreic Oanob river. The Oanob plateau ends in the north-west along the watershed to the Kuiseb valley, which forms the typical moderately steep to steep mountainous highland of the Khomas Hochland. To the south and south-west of the Oanob plateau, the mountainous highland of the Khomas Hochland sets in again, less steeply, forming a rolling landscape.

The geology is dominated by various formations of the Damara Sequence. The Oanob plateau and adjacent Kuiseb catchment are formed by highly erodible quartz-biotite ('mica') schists of the Kuiseb Formation (Geological Survey 1980, South African Committee for Stratigraphy 1980). The Auas Mountains and Lichtenstein Mountains form a band of the Auas Formation, which consists of interbedded

layers of quartzite. To the south of the Auas Formation is an extensive band of the Chuos Formation, bounded by a narrow band of the Kudis Subgroup. Both these formations are also characterised by schists, whilst the Chuos Formation contains a variety of other lithological layers (South African Committee for Stratigraphy 1980). The Aris / Omeya valley is underlain by gneisses and schists of the Hohewarte Complex, which is to the south, overlain by quaternary alluvial deposits (South African Committee for Stratigraphy 1980). In the far south, the Oamites mountain forms part of the Billstein Formation, characterised by grey quartzites and acidic igneous rocks (South African Committee for Stratigraphy 1980).

### Climate

The climate of the Auas-Oanob Conservancy can best be described as a subtropical steppe climate (BS), following Köppen (1936). The mean annual rainfall ranges from below 300 mm in the far south-west to about 350 mm in the north-east (Figure 1a). No information is available on the orographic effect of the high mountain ranges on the precipitation, but it is expected to be considerable. Temperatures can rise to about 36°C in summer, whilst frost can be expected during the winter months from May till as late as October (Figure 2).

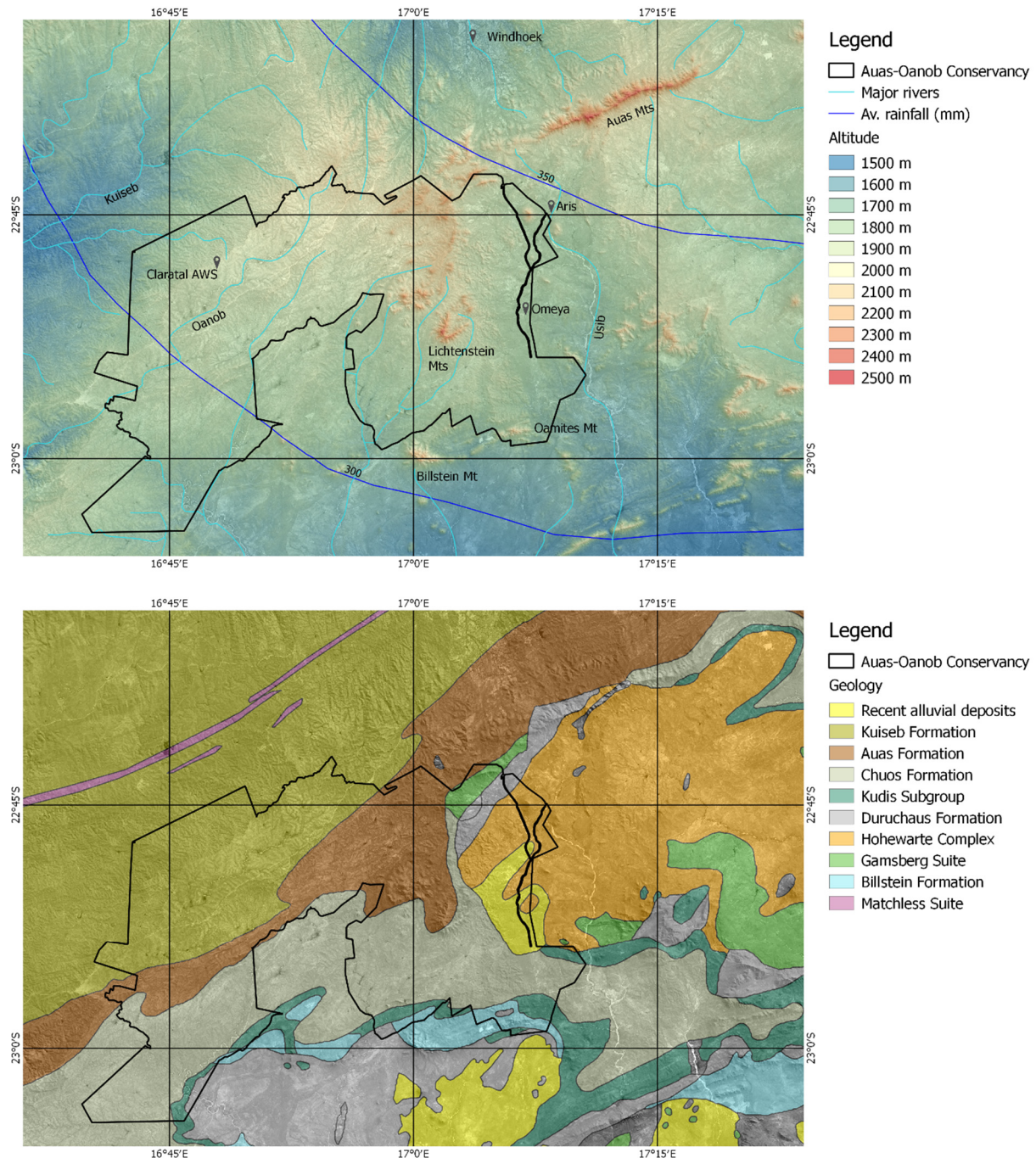
## METHODS

### Field survey

Surveying followed the general method employed for the Vegetation Survey of Namibia project (Strohbach 2001, 2014). Sampling of 229 plots throughout the study area took place during April 2000 and again in April 2002. At each survey plot of 20 x 50 m, a Braun-Blanquet type relevé was compiled. All vascular plant species occurring were noted down, as well as their typical growth forms and estimated crown cover. Habitat descriptors included: the position, by way of a GPS-reading (referenced to Schwarzeck), the landscape, local topography, slope and aspect, lithology, degradation indicators, as well as a photograph. Unknown species and reference specimens were collected, identified and deposited at the National Herbarium of Namibia (WIND). The relevé data were captured on TurboVeg (Hennekens & Schaminée 2001). The data form part of GIVD AF-NA-001 (Dengler et al. 2011, Strohbach & Kangombe 2012). Two further sets of data were available for the northern part of the conservancy: 19 relevés compiled by Kellner (1986) on the farm Claratal (also part of GIVD AF-NA-001), as well as

<sup>1</sup> Recent name changes to the genus *Acacia* (Kyalangalilwa et al. 2013) have not been recognised by prevailing taxonomic literature in Namibia (Klaassen & Kwembeya 2013). To avoid confusion, also in the reference to older

vegetation descriptions, the name *Acacia* is thus used throughout this paper. Alternative names are provided in the appendices.

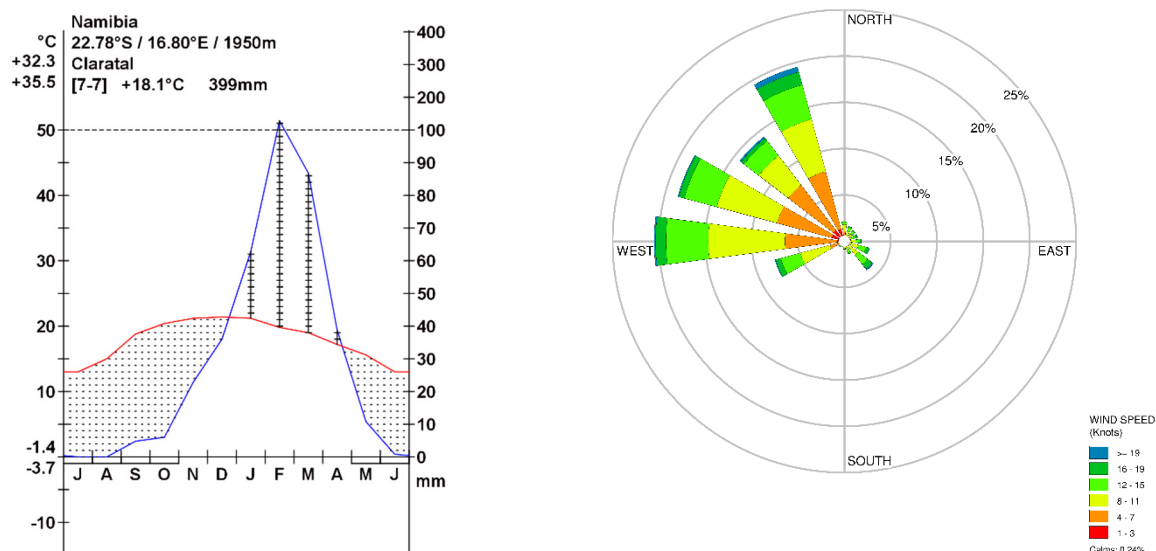


**Figure 1a** (above): Altitude and topography of the study area. **1b** (below): Geological formations of the study area. Data sources: background image: LandSat TM scene path 178 row 076 band 7, dated 4 May 2001; Awas-Oanob Conservancy boundaries: “Environmental Information Service Namibia” (2014); DEM for altitude map: Jarvis et al. (2008); rainfall isohyets and rivers: (NARIS 2001); geological information: Geological Survey (1980).

19 relevés from the Claratal BIOTA observatory (Jürgens et al. 2010, forming part of AF-00-003 - Muche et al. 2012), collected in 2005 by the author of this paper. As the high Lichtenstein mountains form a southern extension of the Awas mountain range, 34 relevés available as part of GIVD AF-NA-001 from Burke and Wittneben (2007), collected in 2004 in the Awas mountains, were also included in the extended data set.

Essential differences in data collecting were as follows: Kellner (1986) collected his relevés on a 625 m<sup>2</sup> plot (25 x 25 m). His original field notes were not available: the data were captured from phytosociological tables. No GPSs were available at the time of collecting, meaning that the exact location of his sample plots is unknown. Burke and Wittneben (2007) collected their relevés on undefined plots, essentially a short transect, along an altitudinal gradient (A Burke pers. com). This was necessary





**Figure 2:** Climate diagram following the convention of Walter et al. (1975) and wind rose of Claratal within the Auas-Oanob Conservancy. Data source: SASSCAL (2014).

due to the extremely steep and rough terrain. GPS positions for the start of each transect are known, and the habitat descriptions followed largely the standards of the Vegetation Survey. Missing habitat data (e.g. geology) could be extracted from GPS sources based on the GPS position. The Claratal BIOTA observatory relevés were collected on 20 x 50 m plots, following the standards of the Vegetation Survey of Namibia project. The only difference is that their position was predetermined, and the plots strictly oriented in an east-west direction. This results in an occasional mixture of habitats. In this way one of the original 20 relevés from the observatory had to be excluded. To avoid observer bias, data were cleaned before processing following methods described by Strohbach (2014).

The final, extended data set consists of 301 relevés, with 514 species.

### Data analysis

Modified TWINSPLAN (Roleček et al. 2009), using average Sørensen dissimilarity as diversity measure, was used for classification. For the first classification, pseudospecies cut levels were set at 0 and 10% since many species occurred widely spread, but abundance was low outside their typical niche habitats. Crispness values (Botta-Dukát et al. 2005) for this first classification indicated a highly reliable subdivision into three clusters. Due to highly varying internal diversity within each cluster, it was necessary to split the data set into three subsets, and classify each subset separately. These three subsets represent (a) the high-altitude mountain veld of the Auas range, (b) the lowlands and valley habitats and (c) the upland Khomas Hochland veld.

The three subsets were further classified as follows: For the first cluster (high mountain veld of the Auas range), best results were achieved without the use of pseudospecies. Classification procedures were stopped with four subdivisions based on crispness values (Botta-Dukát et al. 2005). Pseudospecies (set at 0 and 10%) were utilised in the classification of the subsequent two data sets. Classification of the second subset was stopped, again based on crispness values, after five subdivisions. However, especially the riparian vegetation proved to be ill-defined. For this reason, Cocktail (Bruehlheide & Flintrop 1994) was utilised to select relevés with a combination of *Ziziphus mucronata*, *Searsia lancea*, *Cynodon dactylon* and a high abundance of *Acacia karroo*, based on a description by Kellner (1986).

Classification of the third subset was stopped after three subdivisions, as these represented the best ecologically interpretable results. However, rocky outcrops, highlighted by both Volk and Leippert (1972) and Kellner (1986) as distinctly different from the vegetation of the rolling hills of the Khomas Hochland, were not defined at all. Cocktail (Bruehlheide & Flintrop 1994) was thus employed to select 11 relevés from cluster 2 to define a new group based on the occurrence of *Manuleopsis dinteri*, *Triraphis rammosissioma*, *Pennisetum foermeranum* and *Ozoroa crassinervia*. Furthermore, during field work it was observed that the vegetation of the far south-western uplands subtly changed to a more xeric form, with *Panicum arbusculum* replacing *Brachiaria nigropedata*. This trend was subsequently confirmed with further field work to the west and south of the present study area, but did not reflect in the present classification of the third data subset (uplands of the Khomas Hochland). Therefore, Cocktail (Bruehlheide & Flintrop 1994) was again

employed to define a new group based on the presence of *Aizoon schellenbergii*, *Aptosimum albomarginatum*, *Panicum arbusculum* and *Hibiscus discophorus*. Seven relevés were selected this way from group four and made into an additional group five.

Diagnostic species were determined using the phi coefficient of association (Chytrý et al. 2002). For this calculation the numbers of relevés were standardised following Tichý and Chytrý (2006). Species with  $\phi \geq 40$  were considered as diagnostic and with  $\phi \geq 60$  as highly diagnostic; however, species with a non-significant fidelity at  $\alpha = 0.05$  using Fisher's exact test were omitted. Species occurring with at least a 60% frequency were regarded as constant and with at least an 80% frequency as highly constant.

The average structure for each grouping (i.e. average tree, shrub, dwarf shrub, perennial grass, annual grass and herb cover) was calculated using the available growth form data. The Shannon Index (as an index of evenness) ( $H' = -\sum p_i \ln p_i$ ) and Simpson's Index (as an index of dominance) ( $D = \sum (n_i/N)^2$ ) (Peet 1974) were calculated for each relevé using Juice (Tichý et al. 2011). For the species density (number of species per 1,000 m<sup>2</sup>), the relevé data from Kellner (1986) were excluded, as these were sampled on 625 m<sup>2</sup> plots (25 x 25 m), not 1,000 m<sup>2</sup> plots as all other relevés.

A Nonmetric Multidimensional Scaling ordination (NMS) (Kruskal 1964) was calculated with PC-ORD version 6.08 (McCune et al. 2002). The data set was reduced by removing the data from Kellner (1986), as these had the most incomplete habitat data, and no lat/long localities to derive data from GIS sources (e.g. altitude, annual precipitation, slope, etc.) The resulting database had 282 relevés with 14 environmental factors, in addition to the classification results. Average Sørensen dissimilarity was used as distance measure, and the ordination was calculated in three dimensions (i.e. three resulting axes), based on an initial scree plot of stress versus dimensions (McCune et al. 2002, Peck 2010). The solutions were calculated with 250 iterations using real data. To aid the interpretation of the resulting scatter plots of the ordination results, the environmental parameters were overlain in a joint plot.

## Mapping

Mapping was performed using a Landsat 7 ETM satellite scene path 178 row 076, dated 4 May 2001. The satellite image was clipped to an area slightly larger than the study area before further processing. This clip was imported into the Definens software package (Definens 2006) and segmented into relatively homogenous areas. The segments were classified using the classified sample sites as ground

truth data. In a final step, the resulting shape file was clipped according to the farm boundaries and areas for each landscape calculated using the QGIS 2.4.0 Chugiak software package (QGIS 2014).

## RESULTS

### Classification results

The classification results are depicted in Figure 3 as a dendrogram.

Observer bias could not be detected, with relevés from various data sources being classified in eight of 13 associations (Table 1). No outlying groups (which is a tell-tale indication of observer bias) could be detected (Figure 6a) (Edwards 2000, Strohbach 2014). It needs to be remembered that Burke and Witteben (2007) focused specifically on the High Mountain Veld, and their relevés thus dominate the classification results of those associations. Conversely, two of their relevés classified into the Khomas Hochland Veld, representing the foothills to the High Mountain Veld. Also, Kellner's focus (1986) was specifically on the Khomas Hochland part of Claratal, and his relevés were accordingly classified into the associations of the Khomas Hochland Veld.

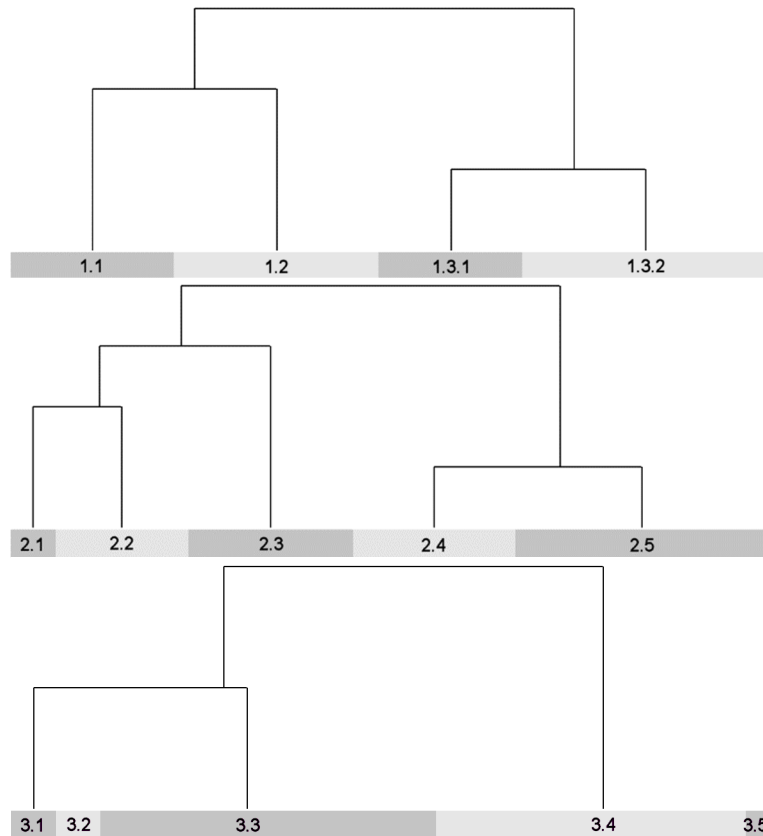
The full phytosociological table is presented as downloadable online [Appendix 1](#), the synoptic table as downloadable [Appendix 2](#). The classification yielded 13 associations in three higher-order syntaxa, one with two subassociations. The associations are not described formally according to the International Code for Phytosociological Nomenclature (ICPN) (Weber et al. 2000), pending further descriptions of the vegetation of the greater Khomas Hochland as well as a review of the syntaxonomy of these. Within the description, highly diagnostic species (with phi coefficient  $> 60$ ) and highly constant species (occurring in more than 80% of relevés) are indicated in **bold**.

### 1 *Digitaria eriantha*—*Osyris lanceolata* high mountain veld

Provisionally recognised at the level of an alliance, the high mountain veld (above 2,000 m asl) represents a unique cluster of associations related to high altitudes. Three associations, one with two subassociations related to aspect, were recognised.

#### 1.1 *Eriocephalus dinteri*—*Danthoniopsis ramosa* high altitude mountain veld of the south-facing slopes

A total of eight relevés has been classified into this association. It is characterised by the diagnostic species *Eriocephalus dinteri*, *Jamesbrittenia pallida*, *Cheilanthes hirta*, *Selago alopecuroides*,



**Figure 3:** Dendrograms depicting the classification results. (a) subset 1, representing the high Auas mountain vegetation; (b) subset 2, representing the vegetation of the lowlands and valleys, and (c) subset 3, representing vegetation of the uplands of the Khomas Hochland.

**Table 1:** Overview of classification results, indicating the source of relevé data on which the description of associations is based.

Veld type	Association (no of relevés)	Data source			
		Strohbach	Kellner (1986)	Burke & Wittneben (2007)	BIOTA (Jürgens et al. 2010)
High Mountain Veld	1.1 (8)			8	
	1.2 (10)			10	
	1.3 (19)	5		14	
Lowlands and valley habitats	2.1 (12)	11			1
	2.2 (6)	6			
	2.3 (9)	9			
	2.4 (8)	7			1
	2.5 (16)	16			
Khomas Hochland Veld	3.1 (12)	12			
	3.2 (13)	10	3		
	3.3 (99)	65	16	2	16
	3.4 (80)	79			1
	3.5 (9)	9			

*Frankenia pomonensis*, *Leucas glabrata*, *Senecio inaequidens*, *Namacodon schinzianum*, *Eragrostis rigidior* and *Tagetes minuta*. These are constantly associated with *Hypoestes forskoolii*, *Digitaria eriantha*, *Danthoniopsis ramosa*, *Oxalis purpurascens*, *Tarchonanthus camphoratus*, *Eragrostis scopelophila* and *Eragrostis nindensis*. A total of 47 species has been observed in this association, with, on average, 27 species per

1,000 m<sup>2</sup>. This association occurs on the upper southern slopes of the Auas mountain range, from about 2,150 m upwards. The southern slopes are extremely steep (>50°), often with steep cliff faces. The relatively high shrub cover, mostly confined to rock crevices, and the virtual lack of annual grass species are conspicuous (Figure 4a).



### 1.2 *Danthoniopsis ramosa*—*Olea europaea* high altitude mountain veld of the north-facing slopes

Ten relevés have been classified into this association. It is characterised by the diagnostic species *Selago angustibractea*, *Lopholaena cneorifolia*, *Cheilanthes multifida*, *Adromischus* species, *Jamesbrittenia hyperioides*, *Eriocephalus scariosus*, *Babiana hypogaea*, *Anthospermum* species, *Cineraria canescens*, *Calostephane marlothiana*, *Hypoxis iridifolia*, *Felicia muricata*, *Wahlenbergia denticulata*, *Tristachya superba*, *Silene burchellii* var. *burchellii*, *Helichrysum obtusum*, *Gladiolus saccatus*, *Crassula capitella* subsp. *nodulosa*, *Monsonia burkeana*, *Euphorbia spartaria*, *Thesium lacinulatum*, *Brachiaria serrata*, *Ebracteola montismoltkei* and *Cotyledon orbiculata*. These are constantly associated with *Tarchonanthus camphoratus*, *Acacia hereroensis*, *Hypoestes forskoolii*, *Digitaria eriantha*, *Danthoniopsis ramosa* and *Chascanum pinnatifidum*. A total of 88 species has been observed in this association, with an average of 48 species per 1,000 m<sup>2</sup>.

This association occurs on the upper northern slopes of the Auas mountain range, again from 2,200 m upwards. The slopes are very steep, flattening off near the mountain peak (but never forming a 'flat' plateau). The soil surface is rock-strewn, with a 50-80% cover of blocky rock boulders. With increasing altitude, the shrub layer (up to 3 m high) recedes and the vegetation becomes dominated by grasses (Burke & Wittneben 2007) (Figures 4b and 5a).

### 1.3 *Acacia hereroensis*—*Tarchonanthus camphoratus* mid-altitude mountain veld

A total of 19 relevés has been classified into this association. It is characterised by the diagnostic species *Anthepphora pubescens*, *Kyphocarpa angustifolia*, *Heteropogon contortus*, *Combretum apiculatum* subsp. *apiculatum* subsp. *apiculatum*, *Ziziphus mucronata*, *Andropogon chinensis*, *Brachiaria nigropedata*, *Schmidtia pappophoroides*, *Microchloa caffra*, *Phyllanthus pentandrus*, *Hibiscus sulfuranthus*, *Dombeya rotundifolia*, *Talinum caffrum*, *Cenchrus ciliaris*, *Geigeria ornativa*, *Hibiscus pusillus*, *Fingerhuthia africana*, *Dyschoriste pseudirecta*, *Oxygonum* species and *Eragrostis echinochloidea*. These are constantly associated with *Tarchonanthus camphoratus*, *Acacia hereroensis*, *Pellaea calomelanos*, *Hypoestes forskoolii*, *Eragrostis scopelophila*, *Eragrostis nindensis*, *Searsia marlothii*, *Digitaria eriantha* and *Commelina africana*. A total of 125 species has been observed in this association, with, on average, 47 species per 1,000 m<sup>2</sup>.

This association occurs on the mid-slopes of the Auas mountain range, at a mean altitude from 2,000 - 2,250 m asl. The slopes are steep to very steep (up to 50°),

but less rock-strewn than at higher altitudes (rock and large stone cover each up to 40%). Gravel cover on these slopes becomes more prominent, also up to 40%. The association can be subdivided into two subassociations based on aspect.

#### 1.3.1 *Acacia hereroensis*—*Tarchonanthus camphoratus*—*elephantorrhiza suffruticosa* mountain veld of the northern slopes

A total of seven relevés has been classified into this subassociation. It is characterised by the diagnostic species *Elephantorrhiza suffruticosa*, *Vigna frutescens*, *Eragrostis porosa*, *Elephantorrhiza elephantina*, *Trochomeria macrocarpa*, *Solanum delagoense*, *Albica* species, *Cyperus margaritaceus*, *Tephrosia rhodesica*, *Sida chrysanthia*, *Evolvulus alsinoides*, *Ipomoea holubii*, *Lantana dinteri*, *Montinia caryophyllacea*, *Lycium eenii*, *Dicoma macrocephala*, *Stipagrostis uniplumis* var. *uniplumis*, *Dipcadi glaucum* and *Ipomoea obscura* var. *obscura*. These are constantly associated by *Tarchonanthus camphoratus*, *Searsia marlothii*, *Pellaea calomelanos*, *Melinis repens* subsp. *repens*, *Hypoestes forskoolii*, *Heteropogon contortus*, *Eragrostis scopelophila*, *Eragrostis nindensis*, *Commelina africana*, *Brachiaria nigropedata*, *Acacia hereroensis*, *Digitaria eriantha* and *Chascanum pinnatifidum*. A total of 105 species has been observed in this subassociation, with, on average, 58 species per 1,000 m<sup>2</sup>.

This subassociation occurs on the northern slopes of the Auas Mountain range. Shrub cover is relatively low, forming an open shrubland between 2-3 m high. The perennial grass cover is variable, but can reach more than 50% (Burke & Wittneben 2007) (Figure 4c).

#### 1.3.2 *Acacia hereroensis*—*Tarchonanthus camphoratus*—*monelytrum luederitzianum* mountain veld of the southern slopes

A total of 12 relevés has been classified into this subassociation. It is characterised by the diagnostic species *Monelytrum luederitzianum*, *Thesium xerophyticum*, *Bidens biternata*, *Raphionacme velutina*, *Phyllanthus* species, *Melinis repens* subsp. *grandiflora*, *Eragrostis lehmanniana*, *Solanum capense*, *Pogonarthria squarrosa*, *Hermannia affinis*, *Drimia sanguinea* and *Aristida effusa*. These are constantly associated by *Tarchonanthus camphoratus*, *Acacia hereroensis*, *Pellaea calomelanos*, *Melinis repens* subsp. *repens*, *Hypoestes forskoolii*, *Eragrostis scopelophila*, *Eragrostis nindensis*, *Digitaria eriantha*, *Searsia marlothii*, *Solanum lichtensteinii* and *Anthepphora pubescens*. A total of 104 species has been observed in this subassociation, with, on average, 40 species per 1,000 m<sup>2</sup>.

This subassociation occurs on the southern and western mid-slopes of the Auas Mountain range including the Lichtenstein mountains. Shrub cover is denser than on the upper reaches, forming a moderately closed, tall shrubland. Grass cover is less variable, from 50 to 60% cover, but occasionally even reaching 70% cover (Figures 4d and 5b).

## 2. Lowlands and valley habitats

The vegetation of the lowlands, particularly the riverine vegetation, alluvial plains of the Omeya valley as well as the foot slopes of the Khomas Hochland form a mixture of vegetation types, which cannot all logically be associated with a clear, single syntaxon. Five associations have been identified here as follows:

### 2.1 *Acacia karroo*—*Cynodon dactylon* riparian vegetation

Twelve relevés have been classified into this association. It is characterised by the diagnostic species *Gomphocarpus fruticosus*, *Cynodon dactylon*, *Ziziphus mucronata*, *Achyranthes aspera* var. *sicula*, *Searsia lancea*, *Felicia muricata* and *Acacia karroo*. These are constantly associated with *Bidens biternata*, *Tagetes minuta*, *Chloris virgata*, *Cenchrus ciliaris* and *Schkuhria pinnata*. A total of 127 species has been observed in this association, with an average of 39 species per 1,000 m<sup>2</sup>.

This association typically occurs as a moderately closed bushland to sub-continuous thicket *sensu* Edwards (1983), dominated by *Acacia karroo*, *Ziziphus mucronata* and *Searsia lancea* trees between 8 to 15 m high along the banks of ephemeral rivers both within the steeper Khomas Hochland landscapes as well as the flatter plains landscapes (Figure 4e and 5c). The herbaceous layer is often dominated by weedy (often annual) and/or shade-loving species. As this association occurs along riverbeds, seeds and other propagules from other, nearby associations are often washed in, resulting in a highly diverse species composition. In this way, even typical sand bank species like *Stipagrostis namaquensis* and *Schmidtia kalahariensis* (which are not known shade species) occur here occasionally.

### 2.2 *Stipagrostis namaquensis* sand banks

Six relevés have been classified into this association. It is characterised by the diagnostic species *Stipagrostis namaquensis*, *Felicia clavipilosa*, *Indigofera alternans*, *Hermbsstaedtia odorata* and *Melanthus comosus*. These are constantly associated with *Acacia karroo*, *Pogonarthria fleckii*, *Bulbostylis hispidula*, *Bidens biternata* and *Acrotome* species. A total of 38 species has been observed in this association, with, on average, 21 species per 1,000 m<sup>2</sup>.

This association occurs within the river bed on sand banks that are stable enough to support a fairly dense grass layer, but frequently threatened by flash floods, so that only a few trees become established. *Acacia karroo* is thus a typical pioneer species in this community, occurring mostly only as juvenile plants or small shrubs. The structure is typically a tall, moderately-closed grassland *sensu* Edwards (1983) (Figure 4f and 5d).

### 2.3 *Platycarphella carlinoides*—*Chloris virgata* floodplains and omiramba

A total of nine relevés was classified into this association. It is characterised by the diagnostic species *Hypertelis salsoloides*, *Tragus berteronianus*, *Talinum cafferum*, *Oxalis depressa*, *Tragus racemosus*, *Monsonia angustifolia*, *Eragrostis pilgeriana*, *Aptosimum glandulosum*, *Platycarphella carlinoides*, *Talinum arnotii*, *Digitaria eriantha*, *Pentzia incana*, and *Eragrostis nindensis*. These are constantly associated by *Kyllinga alata*, *Ursinia nana*, *Pogonarthria fleckii*, *Eragrostis echinochloidea*, *Chloris virgata*, *Mollugo cerviana*, *Melinis repens* subsp. *grandiflora*, *Geigeria pectidea*, *Eragrostis porosa*, *Commelina benghalensis* and *Acacia karroo*. A total of 109 species has been observed in this association, with, on average, 43 species per 1,000 m<sup>2</sup>.

This association is found in the shallow drainage systems (*omiramba*) within the Oanob plateau. These end in rivers and tributary streams of the Oanob River. Typically, they are dominated by a dense, tall grass sward consisting of a combination of annual and perennial species (Figure 4g and 5e). Occasional tree clumps (*Acacia karroo* and/or *Acacia erioloba*) indicate the close relationship to the riverine system, in particular the *Cynodon dactylon*—*Acacia karroo* riverine woodlands. The soils are fine-grained with virtually no stone cover.

### 2.4 *Pupalia lappacea*—*Acacia mellifera* bush-encroached lowlands

A total of eight relevés has been classified into this association. It is characterised by the diagnostic species *Boscia albitrunca*, *Gisekia africana*, *Phyllanthus pentandrus*, *Pupalia lappacea*, *Senecio consanguineus*, *Citrullus lanatus*, *Acacia mellifera* subsp. *detinens*, *Otoptera burchellii*, *Melolobium macrocalyx*, *Catophractes alexandri*, *Albizia anthelmintica* and *Phaeoptilum spinosum*. These are constantly associated with *Pogonarthria fleckii*, *Mollugo cerviana*, *Lycium bosciifolium*, *Schmidtia kalahariensis*, *Oxygonum* species, *Ocimum americanum* var. *americanum*, *Nidorella resedifolia*, *Lycium eenii*, *Eragrostis porosa*, *Commelina benghalensis*, *Acrotome* species and *Acacia erioloba*. A total of 109 species has been observed in this

association, with, on average, 41 species per 1,000 m<sup>2</sup>.

This association occurs in various habitats, often on footslopes of the mountainous highland of the Khomas Hochland, in patches on the Oanob plateau as well as in the undulating landscape of the southern Khomas Hochland. Several large expanses of this vegetation type are visible along the main road (B1) between Windhoek and Rehoboth near the Oamites Mountain, as well as along the C26. These tall, moderately closed shrublands are dominated by *Acacia mellifera* subsp. *detinens* and *Catophractes alexandri*, with *Leucosphaera bainesii* conspicuous in the understorey. The grass layer is generally very weakly developed, with a variety of herbaceous, often weedy, species (Figure 4h and 5f).

## 2.5 *Schmidtia kalahariensis*—*Acacia erioloba* woodlands of the Omeya valley

A total of 16 relevés has been classified into this association. It is characterised by the diagnostic species *Hypertelis bowkeriana*, *Nidorella resedifolia*, *Helichrysum candolleanum*, *Ifloga glomerata*, *Geigeria ornativa*, *Eragrostis cylindriflora*, *Acacia erioloba*, *Geigeria pectidea*, *Kyphocarpa angustifolia* and *Aptosimum albomarginatum*. These are constantly associated with *Pogonarthria fleckii*, *Mollugo cerviana*, *Lycium bosciifolium*, *Kyllinga* species, *Ursinia nana*, *Eragrostis lehmanniana*, *Commelina benghalensis*, *Chloris virgata* and *Melinis repens* subsp. *grandiflora*. A total of 111 species has been observed in this association, with, on average, 40 species per 1,000 m<sup>2</sup>.

These tall, semi-open woodlands are dominated by *Acacia erioloba* (Figure 4i and 5g). The grass layer is dominated by the annual *Schmidtia kalahariensis* in association with *Aristida congesta* subsp. *congesta* and *Antheophora schinzii*. The poisonous species *Geigeria pectidea* and *Elephantorrhiza elephantina* are conspicuous in the herbaceous layer. The association has a park-like appearance (resulting in the development of a golf-estate within this association), but because of the highly unpalatable, annual grass sward and high abundance of poisonous species (*Geigeria pectidea* and *Elephantorrhiza elephantina*) it has low potential for livestock farming.

## 3. *Acacia hereroensis* veld of the Khomas Hochland

*Acacia hereroensis* is the characteristic species of the Highland savanna *sensu* Giess (1998). This third group of associations all contain this characteristic species. The other unifying feature is the undulating to steep hilly and mountainous landscape, generally with shallow, stony soils.

## 3.1 *Ornithoglossum calcicola*—*Fingerhuthia africana* mountain veld of the Oamites mountain

A total of 12 relevés has been classified into this association. It is characterised by the diagnostic species *Ornithoglossum calcicola*, *Melhania damarana*, *Crotalaria kurtii*, *Thesium xerophyticum*, *Monechma genistifolium* subsp. *genistifolium*, *Enneapogon scoparius*, *Cleome suffruticosa*, *Stipagrostis hirtigluma*, *Zygophyllum pubescens*, *Sarcostemma viminale*, *Euphorbia lignosa*, *Polygala pallida*, *Pelargonium otaviense*, *Stipagrostis ciliata*, *Peliostomum leucorrhizum* and *Euclea undulata*. These are constantly associated with *Searsia marlothii*, *Fingerhuthia africana*, *Eragrostis nindensis*, *Acacia hereroensis*, *Enneapogon cenchroides*, *Talinum caffrum*, *Otoptera burchellii*, *Cenchrus ciliaris* and *Acacia mellifera* subsp. *detinens*. A total of 101 species has been observed in this association, with, on average, 37 species per 1,000 m<sup>2</sup>.

This tall, semi-open shrubland is typically dominated by *Acacia mellifera* subsp. *detinens*, *Combretum apiculatum* subsp. *apiculatum* and *Acacia hereroensis* in the shrub layer, and *Fingerhuthia africana*, *Stipagrostis hirtigluma*, *Monelytrum luederitzianum* and *Enneapogon desvauxii* in the grass layer (Figures 4j and 5h). The association occupies very steep slopes (>60%) with stony soils. Up to 40% of the soil surface is covered with rocks, with smaller fragments also covering up to 40% in total.

## 3.2 *Triraphis ramosissima*—*Manuleopsis dinteri* veld of the rocky outcrops

A total of 13 relevés has been classified into this association. It is characterised by the diagnostic species *Pennisetum foermeranum*, *Cheilanthes marlothii*, *Bidens biternata*, *Manuleopsis dinteri*, *Pellaea calomelanos*, *Combretum apiculatum* subsp. *apiculatum* subsp. *apiculatum*, *Ozoroa crassinervia*, *Triraphis ramosissima*, *Eragrostis scopelophila*, *Kalanchoe lanceolata* and *Phyllanthus pentandrus*. These are constantly associated by *Searsia marlothii*, *Cenchrus ciliaris*, *Melinis repens* subsp. *grandiflora*, *Antheophora pubescens*, *Acacia hereroensis*, *Eragrostis nindensis*, *Stipagrostis uniplumis* var. *uniplumis*, *Enneapogon cenchroides*, *Tarchonanthus camphoratus*, *Talinum caffrum* and *Digitaria eriantha*. A total of 110 species has been observed in this association, with, on average, 45 species per 1,000 m<sup>2</sup>.

This association is found on the rocky outcrops scattered throughout the Khomas Hochland. Although by definition a short, moderately closed bushland, the small sizes of these outcrops (on average less than 0.25 ha) and patchy distribution in the landscape do not warrant this description (Figures

4k and 5i). These rocky outcrops, due to the layered nature of the schists, are well-fissured and provide habitats for smaller, shade-loving plants like ferns (*Pellaea calomelanos*, *Cheilanthes* species). Although not exclusively, most of these rocky outcrops are south facing.

### 3.3 *Brachiaria nigropedata*—*Acacia hereroensis* veld of the central Khomas Hochland

A total of 99 relevés has been classified into this association. It is characterised by *Ziziphus mucronata* and *Brachiaria nigropedata*, which are constantly associated by *Searsia marlothii*, *Antheophora pubescens*, *Acacia hereroensis*, *Eragrostis nindensis*, *Melinis repens* subsp. *grandiflora*, *Stipagrostis uniplumis* var. *uniplumis*, *Monelytrum luederitzianum*, *Aristida meridionalis*, *Schmidtia pappophoroides*, *Kyphocarpa angustifolia* and *Cenchrus ciliaris*. A total of 152 species has been observed in this association, with, on average, 39 species per 1,000 m<sup>2</sup>.

These semi-open to moderately closed tall shrublands, or occasionally, low bushlands are dominated by *Acacia hereroensis* in the tree and shrub layer, and *Eragrostis nindensis*, *Aristida adscensionis*, *Brachiaria nigropedata*, *Monelytrum luederitzianum*, *Aristida meridionalis* and *Stipagrostis uniplumis* in the grass layer (Figure 4l). The altitude ranges from 1,760 to 2,100 m asl, whilst slopes are moderately steep to steep (Figure 5j). No affinity to a particular aspect could be detected. Stone and rock cover is variable, but a high cover of quartz pebbles (40-80%) (referred to as pebble mulch) is conspicuous.

### 3.4 *Panicum lanipes*—*Pentzia incana* veld of the Oanob Plateau

A total of 80 relevés has been classified into this association. It is characterised by *Panicum lanipes*, *Plinthus sericeus*, *Pentzia incana*, *Tragus berteronianus*, *Blepharis integrifolia*, *Barleria rigida* and *Ipomoea bolusiana*. These are constantly associated with *Acacia mellifera* subsp. *detinens*, *Eragrostis nindensis*, *Kyphocarpa angustifolia*, *Aristida adscensionis*, *Microchloa caffra*, *Talinum caffrum*, *Schmidtia pappophoroides*, *Melinis repens* subsp. *grandiflora*, *Hirpicium gazanioides*, *Hermannia modesta*, *Eriocephalus luederitzianus*, *Chascanum pinnatifidum*, *Leucosphaera bainesii*, *Pogonarthria fleckii*, *Phaeoptilum spinosum*, *Stipagrostis uniplumis* var. *uniplumis*, *Monelytrum luederitzianum*, *Searsia marlothii* and *Fingerhuthia africana*. A total of 206 species has been observed in this association, with, on average, 53 species per 1,000 m<sup>2</sup>.

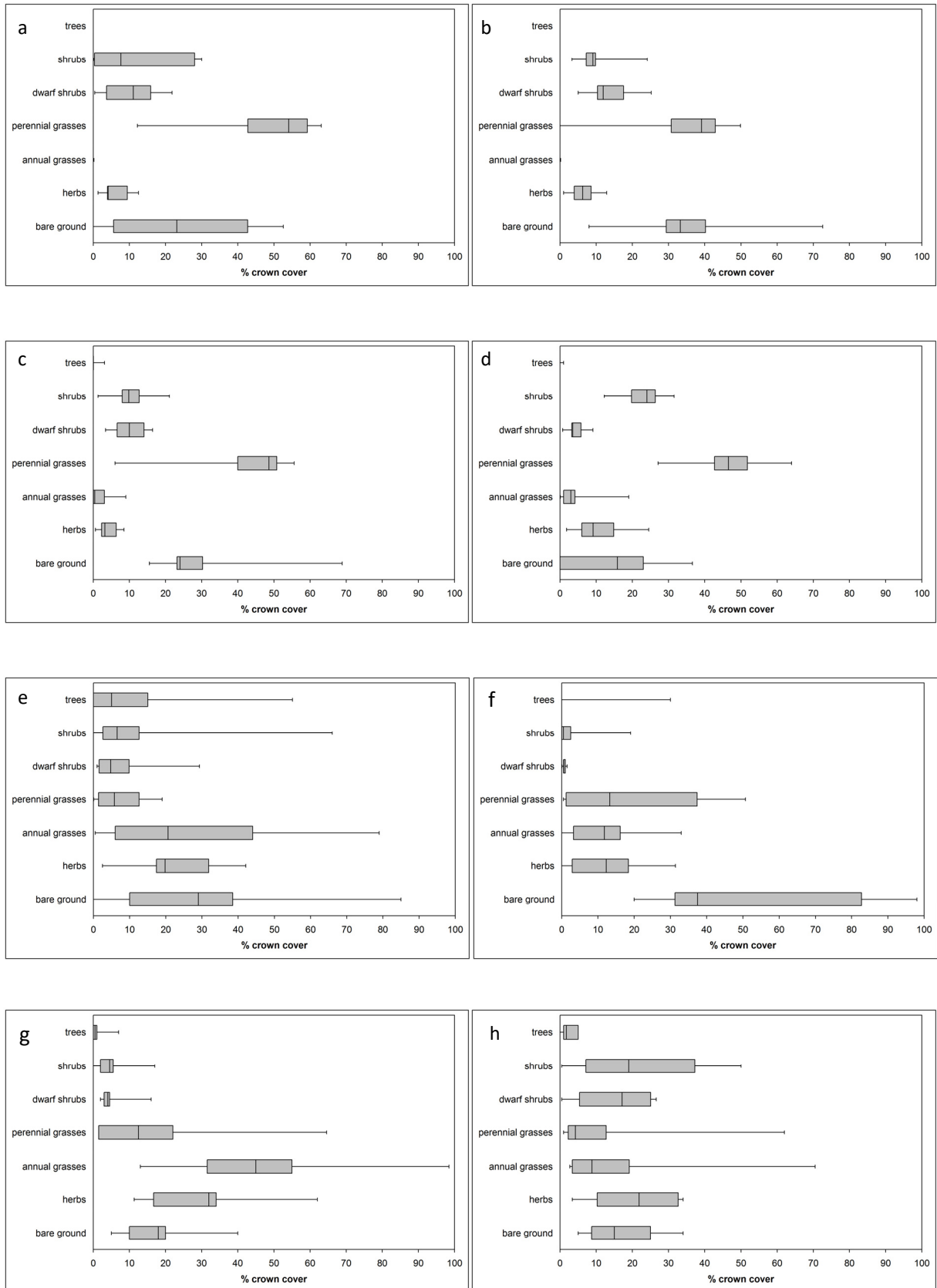
The Oanob plateau has a very distinct karroid structure (Figure 4 m), with the dwarf shrub species

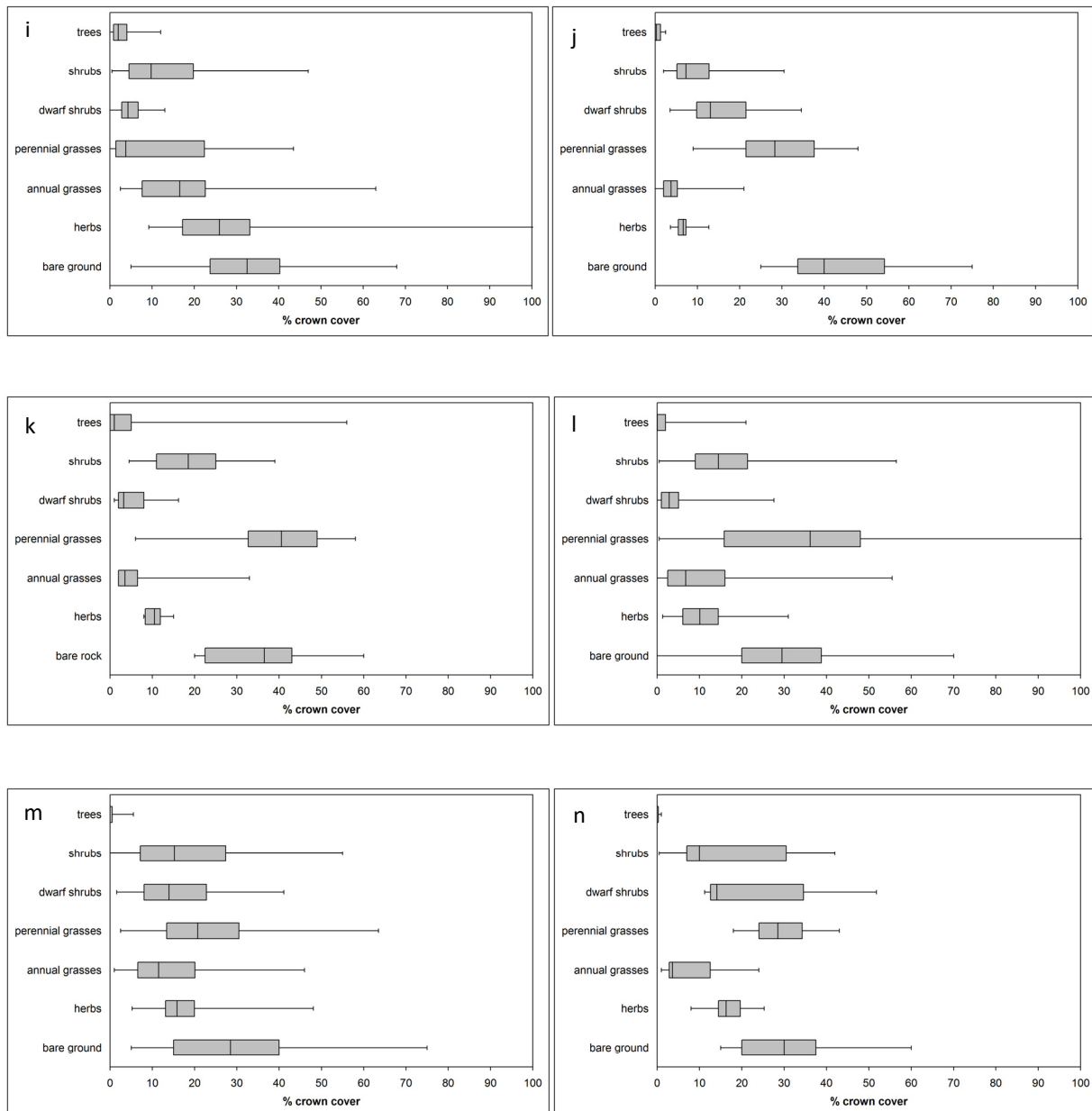
*Leucosphaera bainesii*, *Eriocephalus luederitzianus* and *Pentzia incana* dominating. Larger shrubs are generally fairly sparse, with a tendency for species from the adjacent *Brachiaria nigropedata*—*Acacia hereroensis* association to encroach. In patches, *Acacia mellifera* subsp. *detinens* forms dense bushlands within this landscape. The altitude of the Oanob plateau ranges from 1,680-2,040 m asl., with an undulating to rolling relief (<10° slopes) (Figure 5k). The lithology is similar to the adjacent association, dominated by schists of the Kuiseb, Auas and Chuos Formations. Rock outcrops are generally absent, with relatively few large stones. The pebble mulch typical of the Khomas Hochland is, however, present (40-80% cover). The plateau is interspersed with numerous *omiramba*, as described for association 2.3 (Figure 8).

### 3.5 *Panicum arbusculum*—*Acacia mellifera* veld of the southern Khomas Hochland

A total of nine relevés has been classified into this association. It is characterised by the diagnostic species *Kohautia cynanchica*, *Aizoon schellenbergii*, *Aptosimum albomarginatum*, *Panicum arbusculum*, *Justicia guerkeana*, *Catophractes alexandri*, *Ipomoea obscura* var. *obscura*, *Heliotropium ciliatum*, *Aptosimum lineare*, *Pentzia monocephala*, *Hibiscus discophorus*, *Phyllanthus maderaspatensis*, *Phaeoptilum spinosum*, *Ptychlobium biflorum* subsp. *angolensis*, *Ocimum americanum* var. *americanum*, *Aptosimum spinescens*, *Eriospermum flagelliforme*, *Asparagus exuvialis*, *Melhaniea virescens*, *Leucosphaera bainesii*, *Eriocephalus luederitzianus* and *Pogonarthria fleckii*. These are constantly associated by *Stipagrostis uniplumis* var. *uniplumis*, *Kyphocarpa angustifolia*, *Fingerhuthia africana*, *Eragrostis nindensis*, *Acacia mellifera* subsp. *detinens*, *Pogonarthria fleckii*, *Microchloa caffra*, *Melinis repens* subsp. *grandiflora*, and *Hermannia modesta*. A total of 118 species has been observed in this association, with, on average, 60 species per 1,000 m<sup>2</sup>.

These moderately closed, high shrublands are dominated by *Acacia mellifera* subsp. *detinens*, *Catophractes alexandri* and *Phaeoptilum spinosum*, with the occasional *Rhigozum trichotomum*. The absence of *Brachiaria nigropedata* and the presence of *Panicum arbusculum*, a grass species known as a climax grass in the rocky slopes of the Nama-Karoo in southern Namibia, is conspicuous (Figure 4n and 5l). The landscape is generally rolling (<10°) and not as steep as the central Khomas Hochland. Stone cover, with the exception of pebble mulch, is also lower than in similar landscapes within this cluster of vegetation associations. The altitude ranges from 1,670-1,810 m asl.





**Figure 4:** Box-and-whisker plots of the typical structure of the various associations and subassociations. a) *Eriocephalus dinteri*—*Danthoniopsis ramosa* association; b) *Danthoniopsis ramosa*—*Olea europaea* association; c) *Acacia hereroensis*—*Tarchonanthus camphoratus*—*elephanthorrhiza suffruticosa* subassociation; d) *Acacia hereroensis*—*Tarchonanthus camphoratus*—*monelytrum luederitzianum* subassociation; e) *Acacia karroo*—*Cynodon dactylon* association; f) *Stipagrostis namaquensis* association; g) *Platycarphella carlinoides*—*Chloris virgata* association; h) *Pupalia lappacea*—*Acacia mellifera* association; i) *Schmidtia kalahariensis*—*Acacia erioloba* association; j) *Ornithoglossum calcicola*—*Fingerhuthia africana* association; k) *Triraphis ramosissima*—*Manuleopsis dinteri* association; l) *Brachiaria nigropedata*—*Acacia hereroensis* association; m) *Panicum lanipes*—*Pentzia incana* association; n) *Panicum arbusculum*—*Acacia mellifera* association.

## Environmental Gradients

The NMS produced an ordination in three dimensions, with the final stress for the best solution being 17.248, and an instability of 0.000, after 76 iterations. The high mountain veld is distinctly separated from the remaining clusters along Axis 1, whilst the lowlands and valley habitats are clearly separated from the Khomas Hochland veld along Axis 3 (Figure 6a). The overlap between association

2.3 (*Pupalia lappacea*—*Acacia mellifera* bush-encroached lowlands) and the Khomas Hochland veld indicates the close relationship between these two groupings, with association 2.3 most likely being an encroached / degraded form of the Khomas Hochland veld.

Axis 1 presents an altitudinal and precipitation gradient (Figure 6b and c). Altitude is correlated to Axis 1 with  $r=0.802$ , whilst precipitation is correlated









**Figure 5:** Typical views of the various associations and subassociations. a) *Danthoniopsis ramosa*—*Olea europaea* association; b) *Acacia hereroensis*—*Tarchonanthus camphoratus*—*monelytrum luederitzianum* subassociation; c) *Acacia karroo*—*Cynodon dactylon* association; d) *Stipagrostis namaquensis* association; e) *Platycarphella carlinoides*—*Chloris virgata* association; f) *Pupalia lappacea*—*Acacia mellifera* association; g) *Schmidtia kalahariensis*—*Acacia erioloba* association; h) *Ornithoglossum calcicola*—*Fingerhuthia africana* association; i) *Triraphis ramosissima*—*Manuleopsis dinteri* association; j) *Brachiaria nigropedata*—*Acacia hereroensis* association; k) *Panicum lanipes*—*Pentzia incana* association; l) *Panicum arbusculum*—*Acacia mellifera* association. Source: Photo 5a: Dr A. Burke, all others by the author.

to this axis with  $r=0.632$ . Axis 2 and 3 represent gradients related to primarily geology, which also manifest themselves in slope and stoniness gradients (Figure 6b and 6c). In Axis 2 a clear split between relevés compiled on the Kuiseb - and the Chuos Formations is evident (Figure 6b), whilst relevés compiled on the Auas Formation form a link between these two groupings. Axis 3 depicts a gradient between metamorphic rock types (mostly schists and quartzites, but also marbles) and more recent fluvial sedimentary deposits (Figure 6c). This could also be interpreted as a gradient from more skeletal soils with a high stone or rock content to deeper soils with a high gravel content, typical of alluvial deposits.

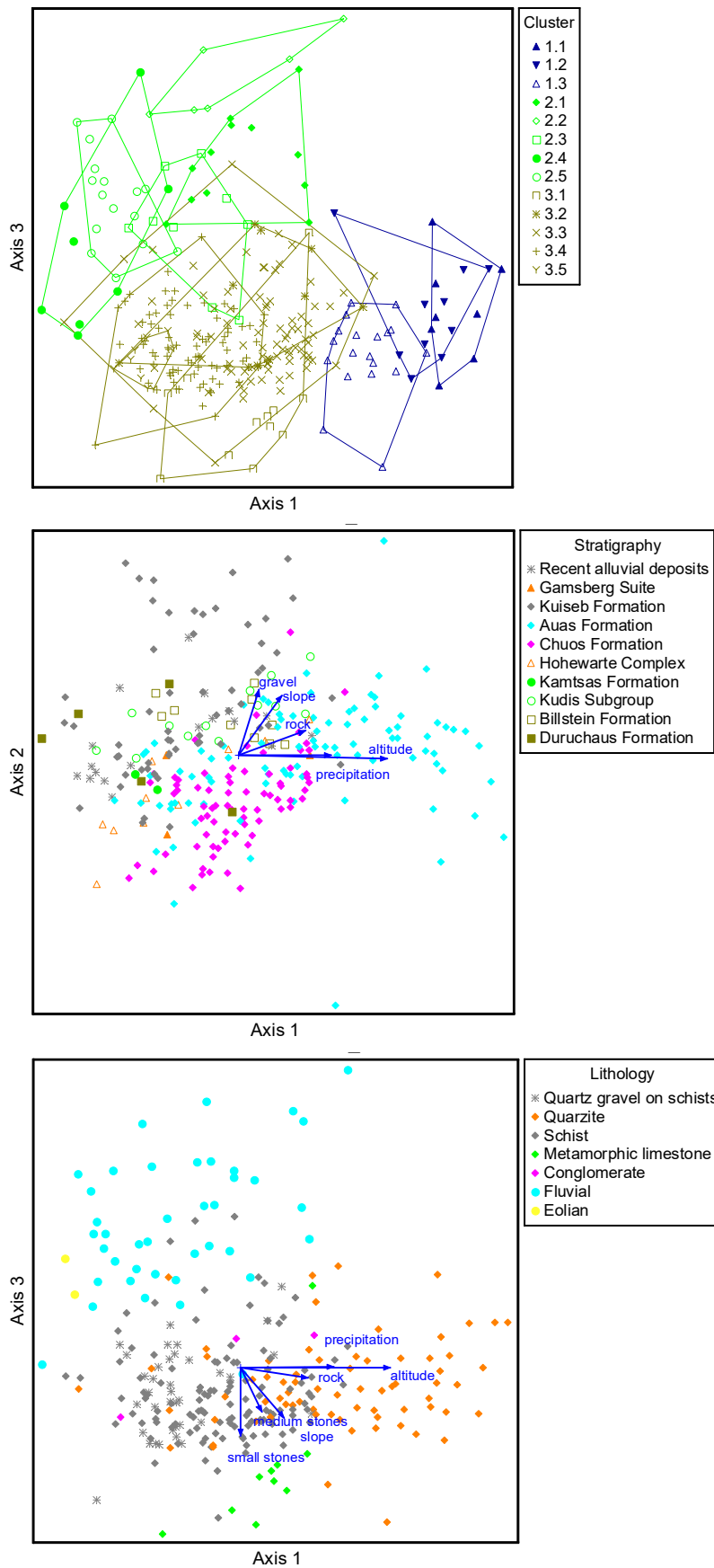
### Biodiversity indicators

The species richness per 1,000 m<sup>2</sup>, Shannon Index and Simpson's Index are depicted in Figure 7.

### Vegetation map

It was not possible to map all individual vegetation associations separately, especially as a number of these occur as small patches, embedded within larger associations (Figure 8). The roughness of the terrain, coupled with numerous shade areas complicated mapping further. For this reason, larger landscape units were mapped, incorporating the major (dominating) vegetation associations, with included smaller associations (Table 2). This follows the SOTER approach of landscape mapping (FAO 1995). A similar approach has been followed by, amongst others, van Rooyen et al. (2008) and Hüttich et al. (2009).





**Figure 6:** Scatter plots of the NMS ordination: (a) with the classification results indicated as convex hulls; (b) indicating the geological stratification as well as environmental factors as biplots; and (c) indicating the lithology of the substrate as well as environmental factors as biplots.

**Table 2:** Vegetation associations included in the various mapping units depicted in Figure 8, as well as their measured area

Mapping unit	Major vegetation association	Included vegetation association	Area within conservancy (ha)
High Auas mountains	<i>Erioccephalus dinteri</i> — <i>Danthoniopsis ramosa</i> (1.1); <i>Danthoniopsis ramosa</i> — <i>Olea europaea</i> (1.2); <i>Acacia hereroensis</i> — <i>Tarchonanthus camphoratus</i> (1.3)		10,840.8
Rivers	<i>Cynodon dactylon</i> — <i>Acacia karroo</i> (2.1)	<i>Stipagrostis namaquensis</i> (2.2)	1,724.3
Omiramba	<i>Platycarphella carlinoides</i> — <i>Chloris virgata</i> (2.3)		4,298.8
Bush-encroached lowlands*	<i>Pupalia lappacea</i> — <i>Acacia mellifera</i> (2.4)	<i>Panicum arbusculum</i> — <i>Acacia mellifera</i> (3.5)	12,831.2
Omeya Camelthorn savanna	<i>Schmidtia kalahariensis</i> — <i>Acacia erioloba</i> (2.5)	<i>Pupalia lappacea</i> — <i>Acacia mellifera</i> (2.4)	7,818.0
Oamites mountain	<i>Ornithoglossum calcicola</i> — <i>Fingerhuthia africana</i> (3.1)		261.9
Central Khomas Hochland	<i>Brachiaria nigropedata</i> — <i>Acacia hereroensis</i> (3.3)	<i>Triraphis ramosissima</i> — <i>Manuleopsis dinteri</i> (3.2); <i>Pupalia lappacea</i> — <i>Acacia mellifera</i> (2.4)	41,907.9
Oanob plateau	<i>Panicum lanipes</i> — <i>Pentzia incana</i> (3.4)	<i>Pupalia lappacea</i> — <i>Acacia mellifera</i> (2.4)	26,745.8
Southern Khomas Hochland	<i>Panicum arbusculum</i> — <i>Acacia mellifera</i> (3.5)	<i>Pupalia lappacea</i> — <i>Acacia mellifera</i> (2.4)	18,038.8

\* The full extent of this unit is unclear.

## DISCUSSION

### Species Diversity

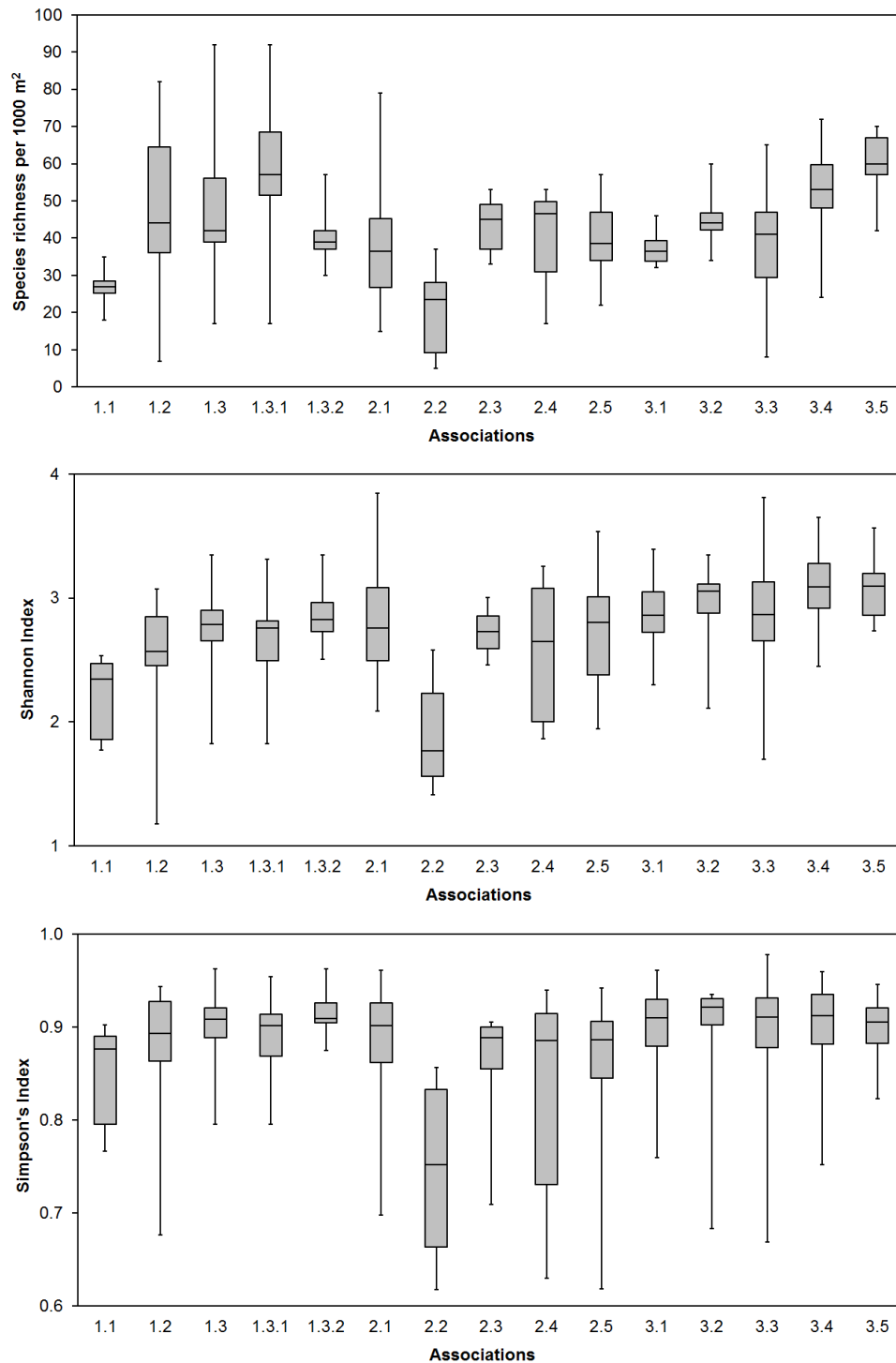
The low species richness of association 2.2 (*Stipagrostis namaquensis* sandbank vegetation), coupled with low Shannon and Simpson's indices, is conspicuous (Figure 7). These are indicative of the absolute dominance of the vegetation by this hard, reed-like grass (Figure 5d). The often- observed dominance by *Acacia karroo* in association 2.1 (riparian vegetation), and *Acacia erioloba* and *Schmidtia kalahariensis* for the Omeya valley veld (association 2.5), is also reflected in these Simpson's index ranges.

More diverse, but equally dominated by a single species, is association 2.4, the *Pupalia lappacea*—*Acacia mellifera* lowlands, according to the Simpson's Index (Figure 7c). The generally wide ranges in low Simpson index values for many associations are also conspicuous, specifically the Khomas Hochland mountain veld (associations 3.1 to 3.5). This is indicative of a tendency of these vegetation types to become bush-encroached (mostly by *Acacia mellifera*), in this way showing a high degree of dominance and less diversity. This is contrary to findings of De Klerk (2004), which indicate that the Khomas Hochland is largely unaffected by bush encroachment.

### Degradation trends

Bush encroachment is the biggest threat to the vegetation (and thus grazing resources) in central and northern Namibia (Bester 1998, De Klerk 2004). This problem was observed in the *Brachiaria nigropedata*—*Acacia hereroensis*, the *Panicum lanipes*—*Pentzia incana* and the *Panicum arbusculum*—*Acacia mellifera* associations. A large number of plots from these three associations are encroached by dense stands of *Acacia mellifera* and would fit, from a structural point of view, into the *Pupalia lappacea*—*Acacia mellifera* association. For this reason, the definition of this association is to be regarded as incomplete, and the mapping of it questionable.

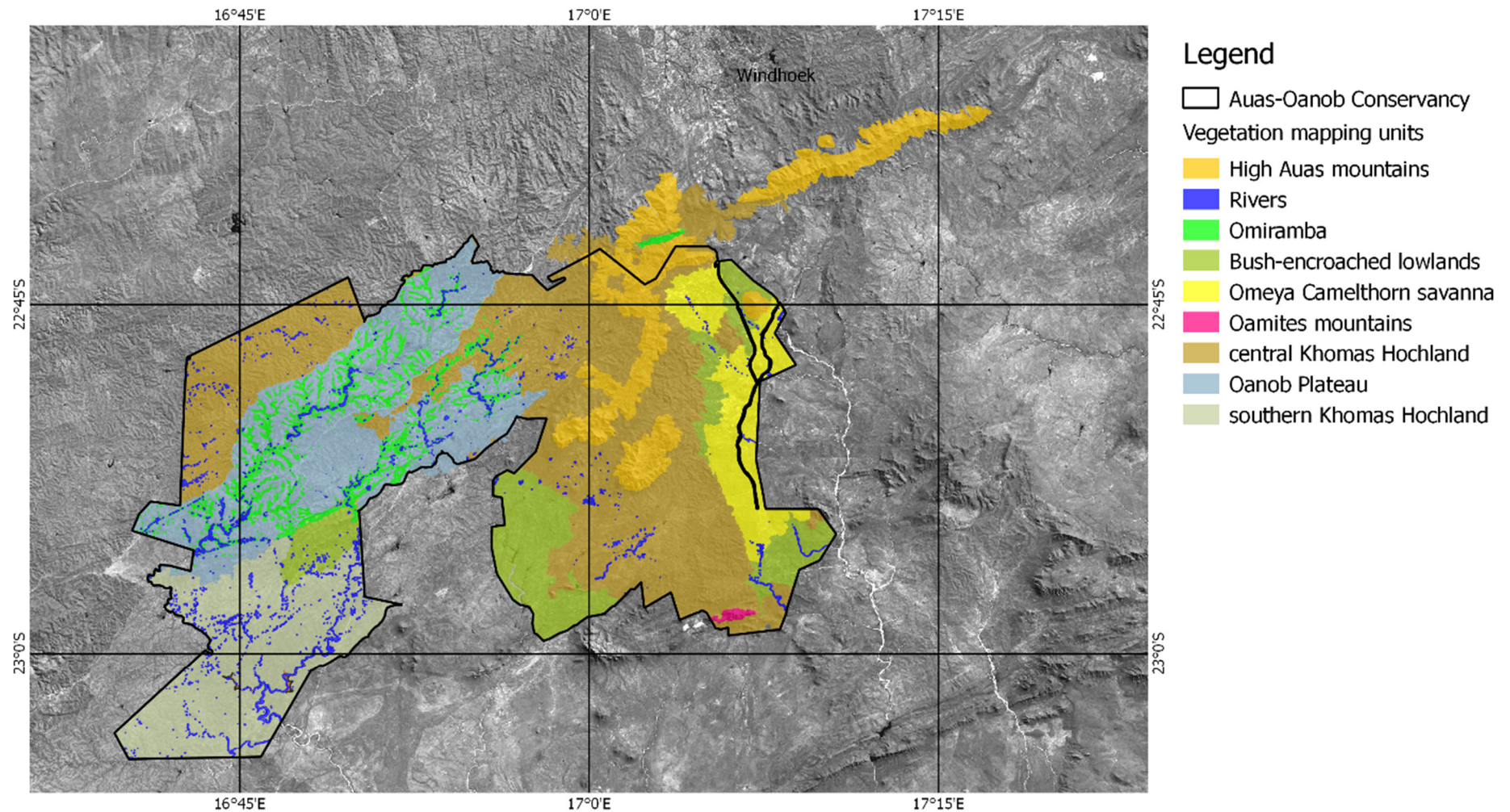
The *omiramba*, in their original state, formed extensive seasonal wetlands with extremely high soil moisture, supporting dense perennial grassland. However, already in the early 1970s, Volk and Leippert (1971) commented that the vegetation of the *omiramba* (their *Themeda triandra* association) was badly degraded, with the only example of the original vegetation remaining in an enclosure on the farm Voigtland. Within the study area, the *Platycarphella carlinoides*—*Chloris virgata* association is also dominated by grasses, albeit a mixture of perennial and annual grasses, with a strong component of pioneer grasses (e.g. *Aristida*



**Figure 7:** Diversity indicators for the various associations described. a) Species richness; b) Shannon index and c) Simpson's Index.

*congesta*, *Tragus racemosus*, *Eragrostis pilgeriana*, *Pogonarthria fleckii*, *Antheophora schinzii*, *Chloris virgata*, *Melinis repens* subsp. *grandiflora* and *Eragrostis porosa*) (Müller 2007). Climax grasses like *Eragrostis superba*, *Digitaria eriantha*, *Cymbopogon pospischilii* and *Themeda triandra* only occur occasionally in protected niches. The soils are highly erodible, forming extensive gullies (Volk & Leipert 1971, Shamathe et al. 2008). In

this way, the rivers are extending steadily uphill into this association. A similar situation has been described for western Australian, where erosion is leading to extensive landscape desiccation (Pringle & Tinley 2003). This means that erosion of the omiramba will not only lead to further degradation of this ecosystem, but will be likely to also have a detrimental effect on the adjacent upstream landscapes by drying these up.



**Figure 8:** Map of the landscapes associated with dominant vegetation types listed in Table 1 within the Awas-Oanob Conservancy and Awas Mountain range. For an A0 size version of this map see downloadable [Appendix 3](#).

## Species conservation

The Auas Mountain range has been identified as a botanically important area in need of protection, because it supports rare, endemic plant species as well as relicts of Grassland and Succulent Karoo vegetation within high altitude habitats (Hofmeyr 2004). Within the three vegetation associations on this mountain range 217 species occur, of which 15 (or 6.9%) are endemic to Namibia. The Khomas Hochland vegetation contains 23 species endemic to Namibia, which is proportionally less (5.8% of 399 species), but also nine exotic species (or 2.3%). Two of these, *Opuntia stricta* and *Prosopis glandulosa*, have been classified as the most aggressive invaders in Namibia (Brown et al. 1985, Bethune et al. 2004,). Neither has reached problematic proportions within the conservancy yet, but the danger of spreading is real.

In conclusion, the Auas-Oanob Conservancy contains several unique landscapes, which can be characterised by high species diversity, a high number of endemic species, high-altitude mountainous habitats, but also arid wetland habitats in the form of the *omiramba*. Current land use is no direct threat to the species richness, but, through various forms of land degradation like erosion, bush encroachment and/or the threat of alien invasive species, will be likely to cause a change of habitat, which in turn, will threaten the plant species diversity and consequently also the habitat of wild and domesticated animal species. The presented description, as well as the original relevé data with associated photographs, will serve as a valuable resource in monitoring changes in the vegetation within this conservancy.

## ACKNOWLEDGEMENTS

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# A comparison of the community dynamics of bioturbating small mammals between livestock and wildlife farming areas in the Kalahari, Namibia

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## ABSTRACT

The study compared abundance and diversity of small mammals between a commercial livestock and neighbouring game farm in the Kalahari Thornveld of Namibia's Omaheke region. Sherman traps baited with standard small-mammal attractants were set out in grids in similar habitats on each land use for four trap-nights during the growing season and the non-growing season of 2015. In total, 174 individuals of five species of small mammals were trapped, 118 on the livestock farm, and 56 on the game farm. Species richness totalled five species of the order Rodentia of which the bioturbating species *Gerbillurus paeba* represented 79.9% (n=139). All five species were trapped on the livestock farm, and only three on the game farm. With similar stocking rates but different grazing management strategies, it is expected that rangeland condition and perennial grass cover differences influenced the densities and species richness of small mammals. This suggests that ecosystem services associated with these mammal species would be more effective on the livestock farm, leading to better soil moisture infiltration and retention, as well as more effective soil nutrient cycling and seed dispersal.

**Keywords:** abundance, bioturbator, diversity, ecosystem services, Kalahari, land use, Namibia, nutrient cycle, rodentia, seed dispersal, small mammals, soil moisture, community dynamics

## INTRODUCTION

Small mammals play a vital role in ecosystem functioning through the dispersal of seeds, seed predation, nutrient cycling through ground bioturbation, specialist and primary consumption, predation and as prey for animals on a higher trophic level (Avenant 2000). This study concentrated on the impacts of livestock and game farming on the abundance and diversity of subterranean-living small mammals, known as bioturbators. Bioturbation is the tilling and mixing of soils by living organisms, usually resulting in heterogeneity of soil structure and aeration (Gabet et al. 2003). This disturbance of soils by burrowing small mammals can positively improve the status of degraded land by supporting key geobiosphere feedbacks (Eldridge & Leys 2003). In particular, their burrow systems increase the surface macro-pores, which can accelerate water infiltration back into the ground and improve plant-available soil water (Eldridge & James 2009). Borchard and Eldrich (2011) found that foraging small mammals turn over between 1,000 to 3,000 kg of soil per year. This process can aid the germination and penetration of seedlings from soils surfaces considered to be crusted and impermeable.

Overgrazing of rangelands and its consequences for vegetation cover in the Namibian savannas are well

documented (Seely & Jacobson 1994). The introduction of wildlife to arid parts of the Namibian savanna has resulted in particular challenges such as selective grazing and shortage of drought reserve grazing (Bothma & du Toit 2010). Farms with excessive numbers of ungulates (both livestock and wildlife) sustain extensive damage to vegetation from trampling and grazing pressure. These land-use induced changes in vegetation cover and composition influence soil temperature and structure (Joubert & Ryan 1999) and consequently affect the habitat assemblage of small mammals (Giere & Zeller 2005). Indirectly, this is likely to feed back to plants as lower aeration and moisture content of soil. Bush thickening, a major form of savanna rangeland degradation, may be a double-edged sword for small mammals (Blaum et al. 2007b). Bush thickets in the midst of open savannas provide important functions for biodiversity (Blaum et al. 2007b) and probably improve soil productivity. They do, however, lower overall rangeland productivity (Tainton 1999) and can lead to fragmentation and a loss of species diversity (Blaum et al. 2007a, 2007b, 2009, 2012).

This study investigated differences in the abundance and diversity of small mammals in two major land uses in the Namibian Kalahari, namely livestock (cattle, horse and sheep) and wildlife (hunting and tourism). It explored the reasons for, and

consequences of, the differences. The study selected small mammals as model organisms since they rapidly respond to sudden changes in their environment (Avenant & Cavallini 2007) and have been successfully used in ecosystem service assessments (Avenant 2000, Avenant et al. 2008). The study further identified key ecosystem services likely to be impacted by the differences in density and diversity of small mammals.

## METHODS

### Study site

The study was carried out in the south-eastern Kalahari Sandveld of Namibia's Omaheke Region at two adjacent farms namely, Kuzikus (23°14.214'S, 18°23.435'E) and Ebenhaezer (23°13.127'S, 18°26.769'E). Kuzikus is a wildlife sanctuary with consumptive and non-consumptive tourism activities. Dominant wildlife species are eland (*Tragelaphus oryx*), oryx (*Oryx gazella gazella*), kudu (*Tragelaphus strepsiceros*) springbok (*Antidorcas marsupialis*), blesbok (*Damaliscus albifrons*), red hartebeest (*Alcelaphus buselaphus*), Burchell's zebra (*Equus quagga burchellii*), giraffe (*Giraffa giraffa angolensis*), black rhinoceros (*Diceros bicornis bicornis*) and ostrich (*Struthio camelus*). Dominant grasses include *Aristida congesta*, *Aristida stipitata* and *Schmidtia kalahariensis*, indicating poor rangeland condition (van Oudtshoorn 2002).

Ebenhaezer is a livestock farm, sustaining herds of cattle, karakul sheep and horses. The farm practices rotational grazing as a management tool to avoid overgrazing (PH Hugo pers. com.). Dominant grasses include *Stipagrostis uniplumis* and *Stipagrostis ciliate*, indicating a veld in good condition (van Oudtshoorn 2002), although *S. kalahariensis*, *A. congesta* and *A. stipitata* also occur.

### Trapping of small mammals

To determine species diversity, richness and abundance of small mammals, four grids (two grids in each farm) consisting of 40 collapsible Sherman aluminium live-traps (23 x 8 x 9 cm) were set concurrently on both farms. Traps were set at 10 m spacing interval (Leirs et al. 1995, Avenant 2000, Avenant & Cavallini 2007). Grid sites were selected with similar soil, vegetation and topographical characteristics, and equidistant from water-points. Trapping was done over four consecutive nights per session during the months of March (growing season), and June (non-growing season) in 2015. These months were chosen because populations of small mammals are most likely to be at their highest during the growing season (Blaum et al. 2007a), and decrease in winter (June), therefore the rate of population decline could be determined. According

to Giere and Zeller (2005) a trap-night signifies one trap set out for 24 hours, hence 160 traps used in the study yielded 640 trap-nights per trapping session. Bait consisted of a mixture of peanut butter, rolled oats, Bovril™ (for insectivores) and sunflower oil. Traps were inspected every morning at 06h00, emptied and then left closed. They were re-baited every afternoon at 17h00 and left open, consequently sampling only nocturnal and crepuscular species. The study excluded diurnal trapping because the extreme maximum temperatures of the Kalahari, which reach up to 45°C in summer (Mendelsohn 2010), would have resulted in unnecessary mortalities of trapped small mammals.

The possible impact of small-mammal burrowing activity on soil moisture infiltration was tested by simulating a 20 mm/hour rainfall event for 1 hour on a 1 m<sup>2</sup> site with burrow activity and on a similar site without burrows. Water was stained with bromothymol blue to record the depth of infiltration. The result of this experiment is descriptive only, as replication was not possible within the short study period.

### Data analysis

Trapped individuals were identified to species level, weighed, and sexed by visual dimorphism. Other morphometric measurements such as right hind-foot length (RHFL) were taken. Hind-foot/mass ratio was tested as a possible indicator of fitness (Krebs & Singleton 1993). Capture-mark-recapture was used, and retrapped individuals were removed from the abundance and fitness analyses. Normality of data was determined by the Shapiro Wilks W test. For non-parametric (all) comparisons between seasons and land uses the Kruskal Wallis Anova test was used. Statistica for Windows version 10 (StatSoft Inc. 2011) was used for statistical analysis. A 95% level of confidence was regarded as significant.

## RESULTS

In total, over both seasons, 174 individuals of five small mammal species were trapped. Of these, 56 (32.18%) individuals were trapped on the wildlife sanctuary (Kuzikus) and 118 (67.82%) on the livestock farm (Ebenhaezer). The livestock land use produced a total species richness of five rodents whereas only three species were trapped at the wildlife sanctuary (Table 1). Overall, *Gerbillurus paeba* represented 79.89% (n=139) and this species was trapped on both farms. Also captured on both properties were *Gerbilliscus brantsii* 13% (n=23), *Mus indutus* 3% (n=6), *Saccostomus campestris* 2% (n=4) and *Gerbilliscus leucogaster* 1% (n=2) were only trapped on the livestock farm (Table 1) and were absent in the wildlife sanctuary. Abundance of small mammals (mean per grid per trap-night) was significantly higher in the livestock farm, in both the



growing season (Fig. 1a) ( $H_{1,12}=4.41$ ,  $p<0.05$ ) and the non-growing season (Fig. 1b) ( $H_{1,12}=16.00$ ,  $p<0.01$ ).

The fitness of *G. paeba* individuals (mass / right hind foot ratio) was higher on the livestock farm, and marginally lower in winter. These differences were, however, not statistically significant. Low numbers of trapped individuals from other species precluded any statistical comparison of fitness.

A single soil moisture infiltration experiment (20mm/hour rainfall event) provided a preliminary indication that rainfall infiltrates deeper in soils with burrows than without. Figure 3 illustrates water infiltration up to a depth of 300 mm in an area without burrowing activity, and 400 mm in an area with small-mammal burrowing.

Species richness (mean per trap-night) was significantly higher on the livestock farm (Ebenhaezer) in both the growing season (Fig. 2a) ( $H_{1,12}=4.41$   $p<0.05$ ) and the non-growing season (Fig. 2b) ( $H_{1,2}=5.75$   $p<0.01$ ).

## DISCUSSION

Significantly higher abundance and species richness of small mammals were found on the livestock farm in both the growing and non-growing season of 2015. This corresponds with earlier work by Caro (2001) and Muck and Zeller (2006) who showed that

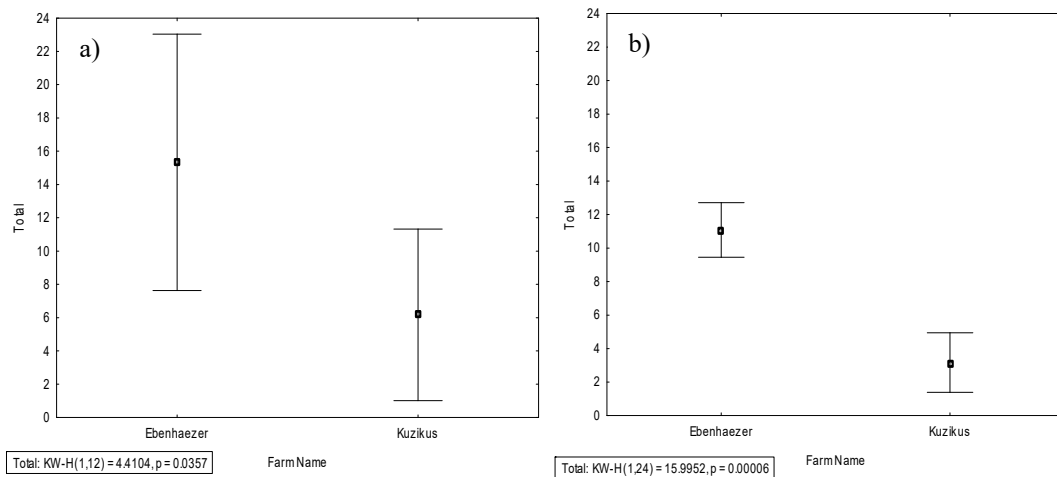
abundance and diversity of small mammals were higher in areas adjacent to protected areas as opposed to inside. Though the cause of this phenomenon is unclear, Caro (2001) speculated that inter-specific competition between continually grazing ungulates and small mammals as well as reduced cover by mega-herbivore grazing could be one of the reasons.

Our study also complements the findings of a study at Namibian airports which showed that abundance of small mammals was higher in un-mowed area than in mowed areas (Hauptfleisch 2014). This Namibian study suggested that mowing the airport properties and areas adjacent to the runways could reduce the density and diversity of small mammals (Hauptfleisch & Avenant 2015).

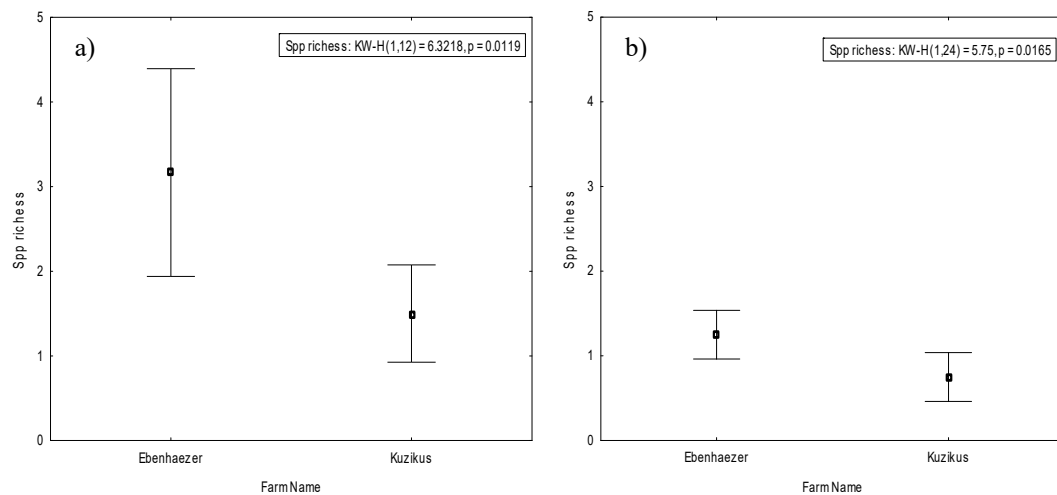
Wild ungulate grazing on enclosed farms such as Kuzikus results in continuous and selective grazing (Trollope 1990) leading to consistently reduced cover, as well as reduced rangeland productivity and sustainability (McGranahan 2008). This results in reduced overall vegetation cover, known to be unfavourable for small mammals (Muck & Zeller 2006, Hauptfleisch & Avenant 2015). This is evident on Kuzikus, the wildlife farm used in this study (BR Reinhardt pers. com.). Conversely, the livestock farmer is able to regulate grazing pressure through rotational grazing and drought reserve planning (PH Hugo pers. com.). Furthermore, the grazing

**Table 1:** Small mammals trapped at Kuzikus and Ebenhaezer during the growing and non-growing seasons of 2015 (Numbers of retrapped individuals are indicated in parentheses).

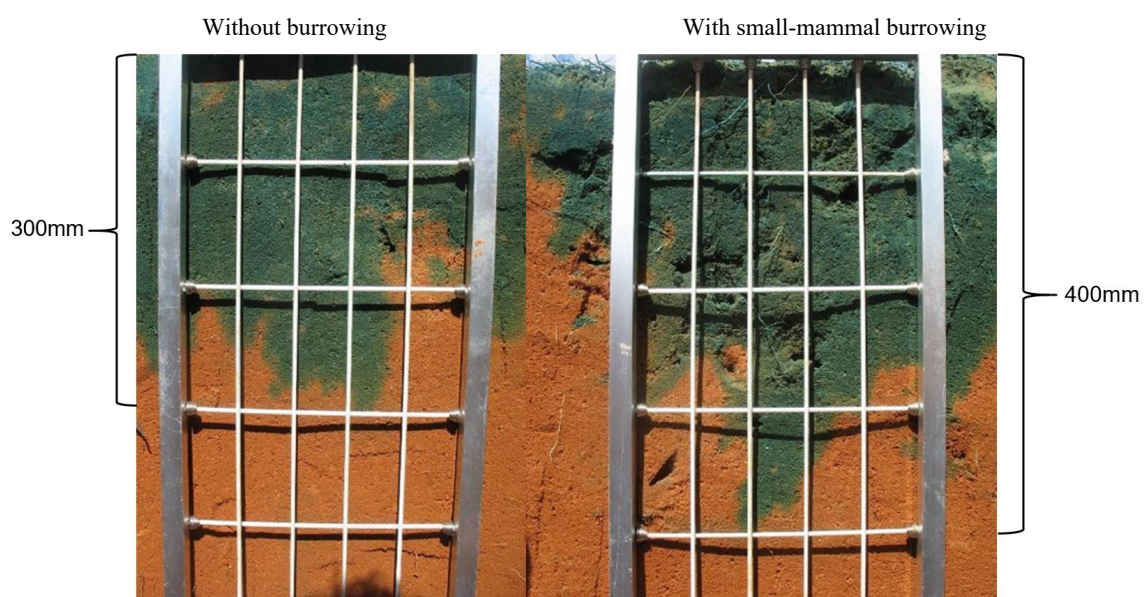
		Ebenhaezer (livestock)			Kuzikus (wildlife)		
Season	Species	Grid A	Grid B	Total	Grid A	Grid B	Total
Summer	<i>Gerbilliscus leucogaster</i> (Bushveld gerbil)	0 (0)	2 (2)	2	0 (0)	0 (0)	0
Winter		0 (0)	0 (0)	0	0 (0)	0 (0)	0
Summer	<i>Gerbilliscus brantsii</i> (Highveld gerbil)	12 (5)	9 (2)	21	1 (1)	1 (0)	2
Winter		0 (0)	0 (0)	0	0 (0)	0 (0)	0
Summer	<i>Gerbillurus paeba</i> (Hairy-footed gerbil)	23 (9)	19 (2)	42	23 (2)	7 (1)	30
Winter		26 (25)	19 (13)	45	7 (6)	15 (11)	22
Summer	<i>Mus indutus</i> (Desert pygmy mouse)	1 (1)	2 (2)	3	0 (0)	2 (0)	2
Winter		1 (1)	0 (0)	1	0 (0)	0 (0)	0
Summer	<i>Saccostomys campestris</i> (African pouched mouse)	1 (1)	3 (3)	4	0 (0)	0 (0)	0
Winter		0 (0)	0 (0)	0	0 (0)	0 (0)	0
Summary							
Overall	Total captured	64	54	118	31	25	56
Summer	Total captured	37	35	72	24	10	34
Winter	Total captured	27	19	46	7	15	22
Summer	Species Richness	4	5		2	3	
Winter	Species Richness	2	1		1	1	
Overall	Species Richness	4	5		2	3	
Summer	Shannon Diversity	0.86	1.2		0.17	0.8	
Winter	Shannon Diversity	0.16	0.2		-	-	
Overall	Shannon Diversity	0.69	0.95		0.14	0.44	



**Figure 1:** Abundance of small mammals (mean per trap-night) in the growing (a) and non-growing season (b) of 2015.



**Figure 2:** Species richness of small mammals (mean per trap-night) in the growing (a) and non-growing season (b) of 2015.



**Figure 3:** Soil moisture infiltration following a simulated one-hour rainfall event of 20 mm.

technique of cattle, feeding on central tillers of tall perennial grass, is known to create corridors for small mammals to move among collapsed outer tillers (Mentis 1981), reducing the risk of predation to small mammals.

Secondly, active meso-carnivore control is practiced on the livestock farm (PH Hugo pers. com.). This includes species such as black-backed jackal (*Canis mesomelas*) and African wildcat (*Felis silvestris lybica*), both important rodent predators (Chimimba 2005). This provides sanctuary for populations of small mammals, a phenomenon also found at Namibian airports surrounded by livestock farming (Hauptfleisch 2014, Hauptfleisch & Avenant 2015).

In recent years more livestock farmers are converting to wildlife ranching as a better option of land use, largely because such practices are perceived to be less costly, more resilient to climate change and more natural for the maintenance of healthier ecosystems and biodiversity (Cloete et al. 2007). However, findings of this study indicate that this may not always be the case, and that wildlife farming with limited grazing management options could be detrimental to biodiversity.

Small mammals have a crucial role in dispersal of seeds and diet for higher predators within the Kalahari (Blaum et al. 2007a). These ecosystem services are likely to be less effective as a result of reduced abundance and species richness of small mammals in the wildlife land use. The loss of bioturbation may reduce soil moisture infiltration (Figure 3) and retention, and important pedological characteristic affecting the performance of arid rangelands.

Dynamic densities and diversity of small mammals found in this study, seemingly perpetuated by the land use practiced on the neighbouring farms, emphasises the value of small mammals as indicators of ecosystem integrity and rangeland condition, supporting other southern African findings (Muck & Zeller 2006, Avenant 2011). Although there are many ways to measure land degradation, the role of small mammals as biological indicators (Avenant & Cavallini 2007, Avenant 2011) could be assimilated in rangeland management practices for savanna ecosystems in Namibia.

While the data collection period of this study was of short duration (one summer and one winter only), the comparative nature of the study provides useful insights. Soil moisture infiltration experiments need to be repeated and key soil and vegetation condition properties should be quantified in order to be able to assess the impacts of the activity of small mammals on soil and vegetation productivity.

## CONCLUSION

The study found significantly higher abundance and diversity of small mammals on a livestock farm compared to its neighbouring wildlife farm. This difference occurred in the growing and non-growing seasons of 2015. No significant differences in the fitness of the small mammals could be determined however. The burrowing hairy-footed gerbil, *G. paeba*, was found to be the dominant species on both land uses, with two other gerbils, *G. leucogaster* and *G. brantsii*, occurring on both land uses. The mice *S. campestris* and *M. indutus* were trapped only on the livestock farm. Our results suggest that continuous and selective grazing on the wildlife farm resulted in consistently lower vegetation cover than on the rotationally-grazed livestock farm, thereby reducing cover for small mammals to thrive. The removal of meso-carnivores from the livestock farm further reduced predation of small mammals. The reduction of numbers of small mammals on the wildlife farm is expected to result in the loss of important ecosystem services such as soil moisture infiltration and retention, soil aeration and seed dispersal. With very little trapping of small mammals currently being conducted in southern Namibia, the survey provides current records for updating knowledge on the distribution of nocturnal small mammals in Namibia.

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# Responses and feedbacks of burrowing mammals under differently managed rangelands

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## ABSTRACT

Bioturbating organisms are known for their benefits to landscapes and ecosystems. Studies have to date largely focussed on invertebrates with very little known about the role burrowing mammals potentially play, especially nocturnally active species. They are thought to be vulnerable to land degradation - such as shrub encroachment and livestock overgrazing - leading to increased negative effects on land productivity through the loss of their associated ecosystem services. In the Kalahari Desert ecosystem of Namibia's Omaheke Region this study compared the abundance and diversity of burrowing medium-sized nocturnal mammals between neighbouring livestock and wildlife land use types. It postulated that bioturbation by nocturnal mammals is an important feedback mechanism leading to improved soil conditions and therefore improved vegetation productivity. The study used nocturnal road strip counts during the growing and non-growing seasons of 2016 to quantify differences in medium-sized mammal population dynamics. Using high resolution multispectral unmanned aerial vehicle imagery, burrow size and abundance as well as vegetation productivity was estimated. The study found a higher diversity of nocturnal medium-sized mammals on the wildlife reserve. Furthermore, clear seasonal patterns were observed. Whereas total sighting number was similar in the growing season and winter on the wildlife reserve; on the livestock farm, there were significantly more mammals spotted in summer, and far fewer in winter. Notably, we revealed that some species of mammal have clear habitat preferences during the different seasons. Results showed that shrub encroachment had a negative relationship with burrow number on both sites, with the livestock farm particularly susceptible. Importantly some benefits were indicated by areas around larger burrows showing higher vegetative productivity. Overall, the study provided valuable insights into the movements, strategies and potential benefits of these mammals. Further research is needed to determine the precise mechanisms by which the burrowers may provide ecosystem functioning benefits to the land users.

**Keywords:** bioturbation, ecosystem engineer, ecosystem services, medium-sized mammals, nocturnal, rangeland productivity, Namibia.

## INTRODUCTION

Land-use type and management are currently predicted to be one of the greatest impacts in global-change biology, particularly in dry environments (Sala et al. 2000). Poor management practices or inappropriate land-use can often lead to ecosystem degradation, such as over-grazing and shrub encroachment, leading to fragmentation and loss of biodiversity (Millennium Ecosystem Assessment 2005, Blaum et al. 2007, 2009). Compounding impacts such as loss of soil fertility, moisture and vegetation productivity are well documented (Prose et al. 1987, Belsky 1994, Fleming et al. 2014). However, a potentially important, yet under-studied aspect of ecosystems in dry regions, are the medium-sized burrowing mammals and mesocarnivores (Blaum et al. 2009), which provide important ecosystem services and are thought to be highly vulnerable to land degradation.

Mesocarnivores are small to medium-sized carnivores of less than 15 kg (Roemer et al. 2009) and medium-sized mammals are classified as mammals with burrow openings of 8-100 cm in diameter (Skinner & Smithers 1990). A potentially important role of these burrowing mammals in ecosystems is bioturbation – the manipulation and movement of soil by biota (Meysman et al. 2006, Fleming et al. 2014). Increasing demand for grassland habitats for livestock has resulted in conflict with medium-sized, herbivorous bioturbators, and global bioturbator numbers are declining (Davidson et al. 2012, Fleming et al. 2014). Although the benefits of burrowing mammals in an ecosystem have been documented (Meadows 1991, Zhang et al. 2003, Meysman et al. 2006, Davidson et al. 2012, Fleming et al. 2014), many livestock and crop farmers are not aware of their importance to rangeland productivity. Poor management therefore has the potential to provide feedback mechanisms that result in poorer and less productive rangelands.

Charles Darwin was the first researcher to observe the importance of burrowing animals by describing the impact of earthworms on landscape function (Meysman et al. 2006). The important role that bioturbation plays in landscape formation and evolution, through soil formation, erosion, soil stabilisation and soil fertility, has only been fully realised in recent years, but has been neglected in Namibia. International studies have found that bioturbating invertebrates improve soil fertility, increase water infiltration into the soil by 4-10 times, improve moisture retention and aerate soil (Edwards & Bohlen 1996, Gabet et al. 2003, Meysman et al. 2006, Bonachela et al. 2015).

There is, however, little empirical information regarding the importance of bioturbating mammals on ecosystems, and especially bioturbating medium-sized mammals and mesocarnivores, which play an important role in ecosystem functioning as ecosystem engineers (Jones et al. 1994, Gabet et al. 2003, Blaum et al. 2007, Roemer et al. 2009, Fleming et al. 2014). Burrowing mammals are defined as allogenic engineers that modify the environment by mechanically changing materials into different physical states (Jones et al. 1994). Ecosystem services provided either directly or indirectly by bioturbating mammals are present in most parts of the world but are often underestimated (De Groot et al. 2002, Roemer et al. 2009, Fleming et al. 2014). These services include habitat creation, soil formation, nutrient cycling, food provision, climate regulation, water regulation and even cultural and/or aesthetic values.

The study aimed to quantify differences in the abundance and diversity of medium-sized burrowing mammals between a wildlife reserve and a livestock farm in Namibia's Omaheke region, and possible impacts of their activity on rangeland productivity.

## METHODS

### Study sites

In this study, two neighbouring sites, with different management practices, were compared: Kuzikus Wildlife Reserve and Ebenhaezer livestock farm (Figure 1a). The study sites were located in the Kalahari sandveld of Namibia (23°12'S, 18°26'E). In general, the study area falls within the Southern Kalahari vegetation type in the broader Tree-and-shrub Savanna biome (Mendelsohn et al. 2002). The Southern Kalahari covers about 12.4 million hectares of land in southern Africa, which includes Botswana, South Africa and south-eastern Namibia (Leistner & Werger 1973). In Namibia, the average annual rainfall in the area ranges from 200-350 mm while average evaporation ranges from 2,000-2,500 mm per year (Mendelsohn et al. 2002). The dominant soils are arenosols, which consist of more than 70%

wind-blown sand. As a result of these factors, water infiltration is rapid, water retention is generally low and nutrients are readily leached out of the soil. Longitudinal, vegetated dunes and open grassland with scattered *Acacia* (*sensu lato*) trees are the characteristic vegetation types found in the area. Growing seasons fall in the summer, starting at the onset of rain, usually between October and June.

### Kuzikus Wildlife Reserve

The 10,500 ha reserve is situated on the edge of the central Kalahari, 180 km southeast of Windhoek (Kuzikus Wildlife Reserve 2010) and at an altitude of 1380 m (Reinhard et al. 2009). The landscapes of the reserve include Kalahari savannah, saltpans with dwarf shrubland, thornbush encroached areas and low, vegetated dunes. The dominant woody vegetation on the reserve includes *Acacia erioloba*, *Acacia karroo*, *Grewia flava*, *Acacia mellifera* subsp. *detinens*, and the dominant grass species are *Aristida* and *Stipagrostis* species. Kuzikus supports about 3,000 grazing and browsing mammals of 20 species such as black rhino, giraffe, common eland, Burchell's zebra, oryx, blesbok, blue and black wildebeest and red hartebeest (Kuzikus Wildlife Reserve 2010). Wildlife continuously graze the reserve as there are no inner fences, which has resulted in over-grazed veld and subsequent increase in bush density in some areas (personal observation). A 2.4 m high game proof fence separates Kuzikus from the eastern neighbouring farm, Ebenhaezer (Reinhard et al. 2009), with which it was compared for this study.

### Ebenhaezer Livestock Farm

Ebenhaezer is a 2,200 ha mixed livestock farm with karakul sheep, cattle and horses being farmed commercially (Vinte 2015). The vegetation type, rainfall, evaporation and soil texture and structure are identical to Kuzikus. The grass sward is however dominated by *Stipagrostis uniplumis*, which in this ecosystem indicates veld in good condition. Rotational grazing is practised by the farm management to prevent over-grazing, and predator control is practised to prevent sheep losses (PH Hugo pers. com.).

### Night survey methodology

Night surveys were conducted to determine and compare species diversity and abundance of nocturnal mammals between the two land-uses. Road strip count routes (Bothma & Toit 2010) transversed both properties in the two dominant habitats (bush encroached and open grassland) (Figure 1b). A fixed, three-hour route was driven at 20 km/h for five consecutive nights. The strips were equidistant on each property and random start and end-points were chosen to eliminate temporal bias of sightings. This



was done both in the growing (25-29 March 2016) and non-growing season (26-30 August 2016) of 2016. A minimum of three people was required to conduct the surveys each night: a driver and two observers/recorders (Sliwa et al. 2014). The two observers each used a spotlight of 1 million candlepower or higher and observed the road on both sides. Each medium-sized bioturbating mammal or mesocarnivore sighted was recorded, including the date and time sighted, GPS coordinates of their location, perpendicular distance estimated from vehicle and the habitat in which they were observed.

### Habitat survey/response

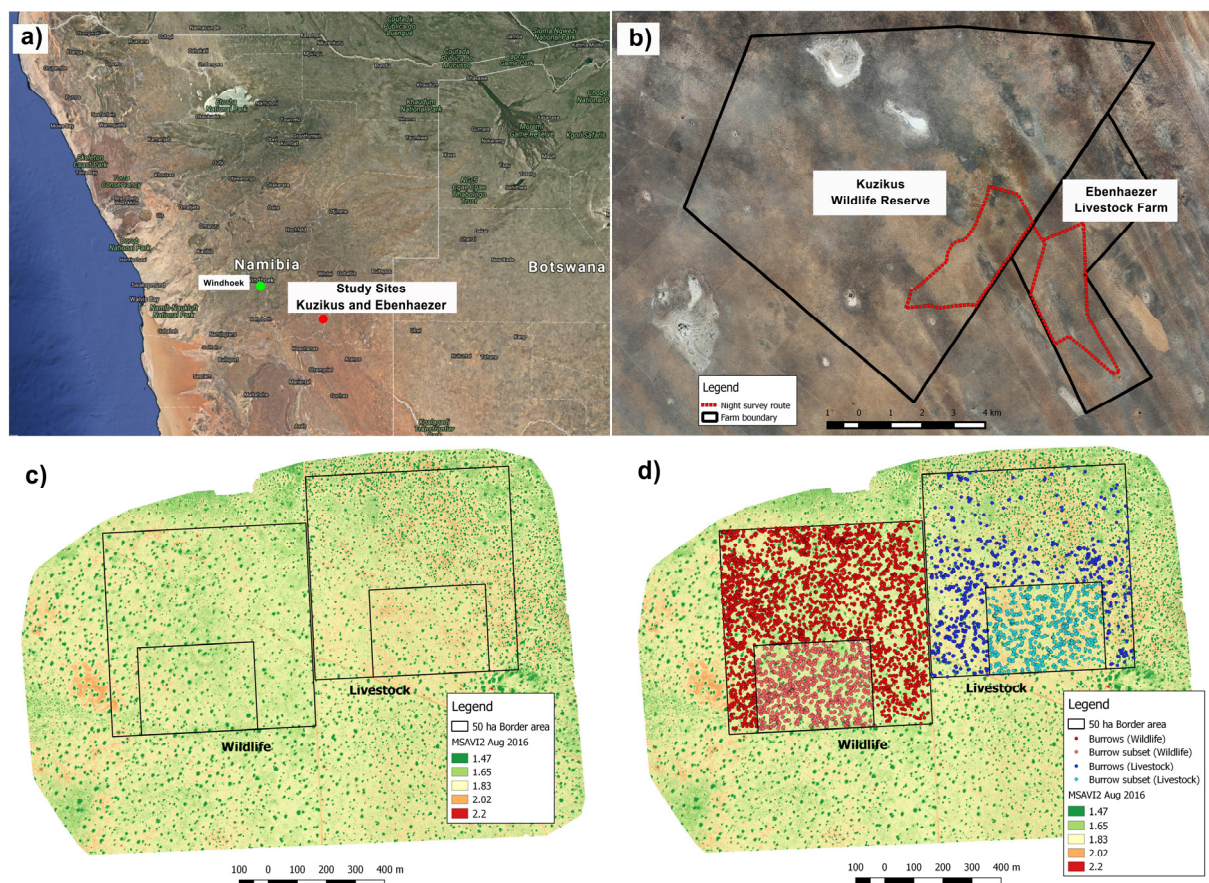
Both sites were surveyed aerially to obtain multispectral imagery of the two land-uses (Figure 1c, d). A senseFly eBee drone was set up, using eMotion 2 software, to fly an area of 100 ha on both properties (senseFly 2015) (Figure 1c, d). Visual (red-green-blue) and near infrared (NIR) georeferenced images of 4 cm pixel resolution were produced. The images were processed using the Postflight Terra 3D software and were used to

quantify the medium-sized mammal burrow density, shrub density and vegetation productivity.

Grid cells of 50 x 50 m were overlaid onto the two sites using QGIS software (Anonymous 2016a) to further analyse the different habitats. Areas of two different sizes were quantified per site. Firstly, approximately the full 100 ha area ("whole survey") per site, containing 196 cells in total; and secondly, due to extensive shrub encroachment, a subset of each area where shrubs were less abundant ("Less shrubby subset" – 48 cells per site) (Figure 1c, d).

### Burrow and shrub density

The medium-sized mammal burrows were marked as points using QGIS and the diameter of each burrow on the images was measured. Burrow size-classes were determined by species use and classified as small, medium and large. Burrow diameters of 8-14 cm (striped polecat, *Ictonyx striatus*) were classified as small, 15-30 cm (springhare, *Pedetes capensis*; black-footed cat, *Felis nigripes*; African wild cat, *Felis silvestris lybica*; small-spotted genet,



**Figure 1:** Two neighbouring study sites of different management types: Kuzikus Wildlife Reserve and Ebenhaezer livestock farm. a) Location map of sites within Namibia. b) Outer borders of sites, location of fences, and night animal survey routes taken throughout growing and winter seasons. c) Defined areas for habitat survey, taken by aerial drone, including the two-sized areas: larger "whole survey" and smaller "less shrubby subset". Including productivity index MSAV12 where lower values signify greater productivity. d) Location of burrows identified on images in the two sites.

*Genetta genetta*) as medium, and 31-100 cm (bat-eared fox, *Otocyon megalotis*; Cape fox, *Vulpes chama*; pangolin, *Manis temminckii*; Cape porcupine, *Hystrix africaeaustralis*; aardwolf, *Proteles cristatus*; aardvark, *Orycteropus afer*) as large (Skinner & Smithers 1990). A subset of burrows was ground-truthed to confirm size classifications (Goodchild 1994).

Similarly, shrubs were also marked on the images using QGIS software. The diameters were measured using the same protocol as for burrows, returning information for shrub number and shrub area per survey area or per cell.

### Site productivity

Multispectral images taken by the eBee drone were processed and analysed using the Postflight Terra 3D software (senseFly 2015) to determine plant productivity across the two sites, as well as productivity around burrows (Bonachela et al. 2015) (Figure 1c). The Modified Soil-Adjusted Vegetation Index (MSAVI2) was used to assess productivity (Huete 1988). This is a commonly-used index that is a version of the Normalised Difference Vegetation Index (NDVI), but additionally corrects for atmospheric conditions, soil and the sun's angle. The MSAVI2 index is a ratio of the reflected visible and NIR light by vegetation (Weier & Herring 2000). Through the process of photosynthesis, visible light is strongly absorbed by chlorophyll inside green plant leaves, however NIR light is strongly reflected due to the structure of the leaves. Most of the visible light is absorbed by healthy plant biomass and, in return, a large amount of NIR light is reflected. In unhealthy or sparse plant biomass, less visible light is absorbed and, in return, less NIR light is reflected. Importantly, MSAVI2 is inversely related to productivity/green biomass. Therefore, lower values of MSAVI2 signify higher values of productivity (usually in the range of 0.5 (high productivity) to 2.5 (low productivity)).

Productivity of vegetation around burrows identified from the multispectral images was also calculated. Buffer areas (5 segments, undissolved) of five meters were created around each individual burrow identified using QGIS. A buffer polygon layer and MSAVI2 overlay was produced for each property and season. The layers were then run through the QGIS "Zonal Statistics" plugin to extract median pixel values for each burrow radius.

### Statistical Analyses

All statistical analyses were performed in R version 3.2.2 (Anonymous 2016b).

The animal night-drive observation data were analysed using Generalised Linear Models (GLM) fitted to a negative binomial distribution, using R

package MASS (Venables & Ripley 2002). In total, there were five nights (replications) of recordings per site per season.

Firstly, the total number of mammal observations was analysed. The response variable for this statistical model was therefore "total mammal observations", and explained with the two-level categorical explanatory variable Site (wildlife reserve or livestock farm), the two-level categorical variable Season (growing or winter), and the interaction of Site x Season.

Secondly, the single-species observational data were also analysed. In total, nine species were observed across both sites, and all are described in the tables and figures for interest (see Appendix 1 for species details). However, four of the species occurred in very small numbers, therefore, statistical analysis was only performed for a subset of the five most common species (African wild cat, aardwolf, bat-eared fox, springhare, and small-spotted genet). Similar to the total observations, the negative binomial GLM for single-species observations included the Site (wildlife reserve or livestock farm), Season (growing or winter), but additionally the five-level categorical explanatory variable Species. Also included were all three two-way interactions, and the three-way interaction Species x Site x Season.

Thirdly, as a measure of site diversity, we applied the commonly used Shannon-Wiener Diversity Index (Shannon 1948). This was modelled using linear models in the R basic stats package (Anonymous 2016b), testing the explanatory variables Site, Season and the interaction Site x Season.

For visual representation of the species observation data, log ratios were calculated to reveal the relative change in number of observations across sites in a season. Log ratios were calculated as the natural logarithm (+0.2) of the mean observations per survey in the wildlife reserve minus the log (+0.2) observations in the livestock farm. Positive values therefore signified relatively greater presence on the wildlife reserve, and negative values greater presence on the livestock farm.

For the habitat survey, many of the aerial outputs were calculated at a site level, and were therefore simply reported descriptively and not statistically analysed. However, using linear models in the R basic stats package (Anonymous 2016b), differences in productivity (median MSAVI2) around the three size categories of burrows (small, medium or large) were tested. This was performed on the whole survey and the less-shrubby subset. Replication for each site and each size area was dependent on the number of burrows observed.



Additionally, using 50 x 50 m cells per surveyed area, relationships between productivity (median MSAVI2 as response variable) and number or area of burrows per cell (log-transformed continuous explanatory variables) were tested. Finally, linear models were used to test for relationships between number or area of burrows and the number and area of shrubs.

## RESULTS

### Animal foraging/observations

The total number of night-time medium-sized mammal sightings was highly dependent on season. The main effect of season was statistically significant (deviance=18.86;  $p<0.001$ ) with more mammal sightings in the growing season than winter (Figure 2a). However, this growing-season increase was only evident on the livestock farm, as indicated by a statistically significant interaction in the GLM of site by season (deviance=33.30;  $p<0.001$ ; Figure 2a). On the wildlife reserve, sightings of mammals were similar in both seasons. Furthermore, while the overall number of sightings was similar at both sites, the sightings of mammals was higher on the livestock farm in the growing season, and higher on the wildlife reserve during winter (Figure 2).

Season also played a large role in determining the number and locality of observations of each single species (Figure 3, Table 1 and Table 2). The statistics and figures reveal that total species richness was higher on the wildlife reserve compared to the livestock farm (5 versus 2 in the growing season and 5 versus 1 in the winter).

Springhare was a key species in the interaction terms of both the single species (Table 2) comparison and total sightings (Figure 2). It switched seasonally in terms of where it was observed more frequently (Site

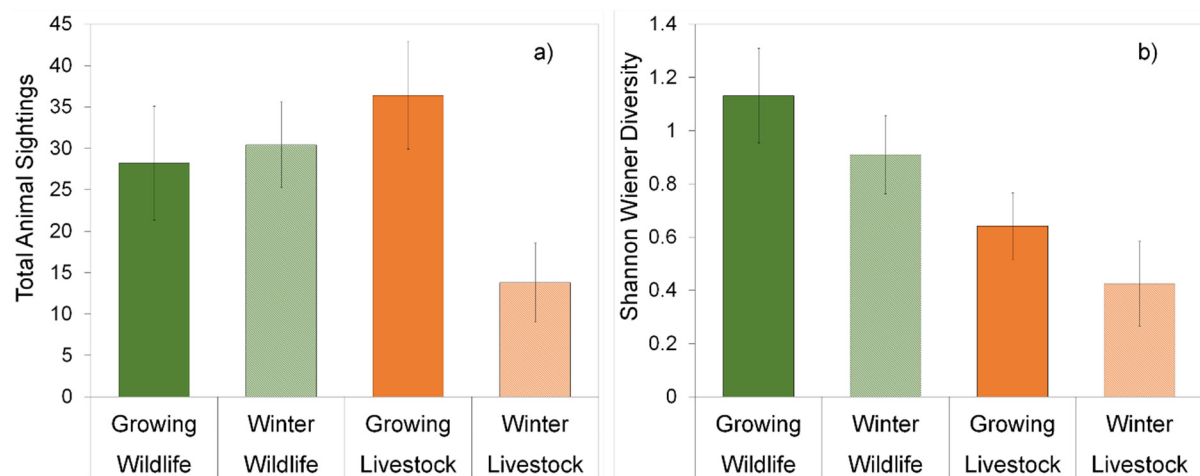
x Season effect: deviance=29.29;  $p<0.001$ ). Overall it was observed marginally more frequently on the livestock farm (Table 1), where it was seen in greater numbers during the growing season (Figure 3) but less frequently in the non-growing season.

In a similar seasonal effect, African wild cat was more abundant on the wildlife reserve during winter compared to growing season (Table 1), whereas on the livestock farm it was only observed in the growing season (Table 1) (note that this is not possible to see in Figure 3). Bat-eared fox was the only species to be seen more frequently on the livestock farm both in the growing season and in the winter (Table 1; Figure 3).

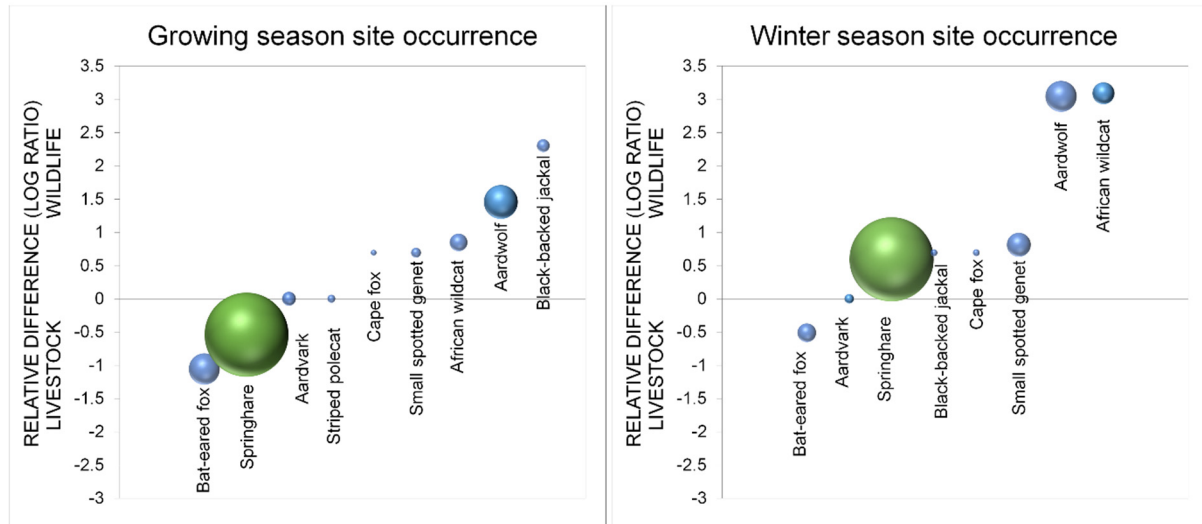
The Shannon-Wiener Diversity Index (SWDI) indicated that diversity was higher on the wildlife reserve (mean SWDI=1.02) than the livestock farm (mean SWDI=0.53) (Figure 1b; Site effect:  $F=20.29$   $p<0.01$ ) when seasons were combined. However, there was statistically no significant difference in diversity related to season ( $F=3.69$   $p=0.08$ ) or the interaction between Site and Season ( $F=0.95$   $p=0.34$ ).

### Habitat response/observations

The analysis of the aerial imagery revealed some important differences between the two sites in terms of number of burrows, shrubs and productivity (Tables 3, 4; Figure 1c, d). Burrows were more abundant on the wildlife reserve over the fully surveyed area and the less shrub-encroached patch (Table 3). Interestingly, the distribution of burrow sizes varied greatly between sites (Table 4) with small burrows being more abundant while medium- and large-sized burrows were much more similar in number at the two sites.



**Figure 2:** a) Mean number of total medium-sized mammal sightings (per night) and b) Mean Shannon Wiener Diversity Index, surveyed during night-time drives. Five night-time drives were conducted at each site of different management type (Wildlife reserve or Livestock farm) during the growing season and winter season. Bars indicate standard errors on the normal scale.



**Figure 3:** Relative frequency of single species sightings during night-time drives. Y-axis gives the log ratio of mammal sightings during the growing season (left) or winter (right) at the two sites of different management types (Wildlife reserve or Livestock farm). Positive values indicate greater frequency of sightings during a season on the wildlife reserve, negative values indicate greater frequency of sightings during a season on the livestock farm. Species are ranked along each separate x-axis by strength of log ratio. Bubble sizes are related to overall frequency of sightings of a given species.

**Table 1:** Mean frequency of medium-sized mammal sightings during five night-time drives per site (Wildlife reserve, Livestock farm) per season (growing season, winter season). Values in italic font indicate standard deviations.

Species	Wildlife Reserve				Livestock Farm			
	Growing		Winter		Growing		Winter	
African wild cat	1.4	<i>0.89</i>	2.2	<i>1.79</i>	0.6	<i>0.89</i>	0.0	<i>0.00</i>
Aardvark	0.6	<i>0.55</i>	0.2	<i>0.45</i>	0.6	<i>0.89</i>	0.2	<i>0.45</i>
Aardwolf	6.0	<i>4.00</i>	4.2	<i>4.97</i>	1.4	<i>0.89</i>	0.2	<i>0.45</i>
Bat-eared fox	1.6	<i>2.07</i>	0.6	<i>0.89</i>	4.6	<i>3.78</i>	1.0	<i>1.00</i>
Black-backed jackal	1.0	<i>0.71</i>	0.2	<i>0.45</i>	0.0	<i>0.00</i>	0.0	<i>0.00</i>
Cape fox	0.2	<i>0.45</i>	0.2	<i>0.45</i>	0.0	<i>0.00</i>	0.0	<i>0.00</i>
Small-spotted genet	0.4	<i>0.55</i>	1.8	<i>1.10</i>	0.2	<i>0.45</i>	0.8	<i>1.30</i>
Springhare	16.8	<i>1.10</i>	21.0	<i>2.35</i>	28.8	<i>4.97</i>	11.6	<i>3.72</i>
Striped polecat	0.2	<i>0.45</i>	0.0	<i>0.00</i>	0.2	<i>0.45</i>	0.0	<i>0.00</i>

**Table 2:** ANOVA table for single-species sighting data during night-time drive survey. Data were analysed using a negative binomial Generalised Linear Model testing for the effect of Species (5 most common species – see text), Site (Wildlife reserve or Livestock farm), Season (Growing or Winter), and all two- and three- way interactions.

\*\*\* indicates  $p < 0.001$ ; NS indicates not significant.

Variable	DF	Deviance	p
Species	4	752.80	***
Site	1	2.20	NS
Season	1	16.17	***
Species x Site	4	53.08	***
Species x Season	4	19.81	***
Site x Season	1	33.97	***
Species x Season x Site	4	3.52	NS

The livestock farm had many more shrubs than the wildlife reserve. There were approximately 3.5 times more shrubs on the livestock farm when viewing the whole surveyed areas, and although much fewer in terms of number, there were still 2.3 times more shrubs on the livestock farm on the less shrubby subset (Table 3). Within both sites, when analysing within cell patterns, burrow area had a negative relationship with shrub area (Figure 4; Wildlife  $F=12.62$   $p < 0.001$ ; Livestock  $F=71.15$   $p < 0.001$ ;  $DF=194$ ).

The wildlife area was more productive in terms of vegetation growth (lower MSAVI2) than the livestock farm (Table 3). Vegetation productivity also varied around burrows of different sizes (Table 4, Figure 5). Across the entire study site and “less shrubby subset” area on both the wildlife reserve and the livestock farm, MSAVI2 was significantly lower – and therefore had higher productivity – around large burrows compared to small burrows (Table 4; Figure 5). However, when analysing values within

**Table 3:** Habitat properties of two neighbouring sites of different management type (Wildlife reserve and Livestock farm) extracted from aerial drone images. Shown are details for burrows and shrubs for both total number and total area (m<sup>2</sup>), and mean number and mean area (m<sup>2</sup>) per 50 x 50 m cells. Additionally, the productivity index (MSAVI2 median) for the surveyed areas are presented, whereby lower values indicate higher productivity. All values shown are for two sizes of survey area: larger “Whole survey” (196 cells) and “Less shrubby subset” (48 cells).

		Whole survey		Less Shrubby Subset	
		Wildlife	Livestock	Wildlife	Livestock
Burrows	Number	4425	1340	1033	683
	Area	160.42	93.91	33.89	40.13
	Area	22.58	6.84	21.52	14.23
Burrows per cell	Number	0.82	0.48	0.71	0.84
	Area	653	2283	130	294
	Area	2,395.65	15,893.49	385.21	1,508.79
Shrubs	Number	3.33	11.65	2.71	6.13
	Area	12.22	81.09	8.03	31.43
	Area	1.7714	1.8176	1.7708	1.8435
MSAVI2 (med.)	Growing	1.7714	1.8176	1.7708	1.8435
	Winter	1.9699	1.9644	1.9619	1.9847

**Table 4:** Number and characteristics of burrows identified from aerial drone images. Burrows from two sites (Wildlife reserve and Livestock farm) and two sizes of survey area (Whole survey and Less shrubby subset) were classified into size categories: small (8-14 cm); medium (15-30 cm); large (31-100 cm). Shown are the number per site per survey size in each category, the proportion of the total number per site per survey size in each category, and the vegetative productivity in the 5 m surrounding each burrow, measured using MSAVI2 median, whereby lower values indicate higher productivity. Letters next to productivity measures indicate values that did not differ significantly when compared using a linear model.

		Whole Survey		Less Shrubby Subset	
		Wildlife	Livestock	Wildlife	Livestock
Number of Burrows	Burrow Size				
	Small	2071	144	522	88
	Medium	1865	847	405	449
Proportion of Burrows	Large	489	349	106	146
	Small	0.49	0.11	0.51	0.13
	Medium	0.40	0.63	0.39	0.66
MSAVI2 around Burrows	Large	0.10	0.26	0.10	0.21
	Small	1.772a	1.838b	1.774a	1.849b
	Medium	1.772a	1.822a	1.769ab	1.832a
MSAVI2 around Burrows	Large	1.767b	1.817a	1.761b	1.830a

cells, there was no relationship between number or area of burrows per cell and productivity in the less shrubby areas (Wildlife  $F=0.98$   $p=0.33$ ; Livestock  $F=0.578$   $p=0.45$ ).

## DISCUSSION

By comparing two neighbouring sites of different management types, our study was able to reveal interesting seasonal and land-use dependent dynamics of the under-studied nocturnal mammals of southern Africa. Both the abundance of active nocturnal medium-sized mammals and burrowing activity was higher within the wildlife reserve compared to the neighbouring livestock farm. Importantly, the study revealed that some mammals have clear habitat preferences during different seasons.

### Nocturnal activity

The observations carried out during night-time drives showed species-specific differences in frequencies of sightings related to management type, season and

their interaction. In general, we would consider the movement of this mainly nocturnal group of animals to be related to their foraging behaviour. Surprisingly, the overall total number of sightings was similar on both the wildlife reserve and the livestock farm. However, this was confounded by a large seasonal shift observed on the livestock farm. Here, numbers were significantly lower in the winter when potential food stocks are low. In contrast, frequencies of sightings were more intermediate, but similar in number, in both the growing season and winter on the wildlife reserve. It seems much more likely that the herbivorous mammals would seek shelter and food in these areas during winter, and that the predators may be attracted by the increased prey presence.

In agreement with our hypothesis, we showed that species diversity was higher on the wildlife reserve than the livestock farm. Five mammals were seen more frequently on the wildlife reserve compared to the livestock farm, where there were only two more frequent in the growing season and one more frequent in winter. The only mammal to be consistently seen

more frequently on the livestock farm was bat-eared fox. In contrast, aardwolf were more commonly observed on the wildlife reserve. This species had a higher foraging preference on the wildlife reserve in both the growing season and winter season, with similar seasonal proportions but lower sightings on the livestock farm (Williams et al. 1997, Blaum et al. 2009). African wild cat were mainly observed on the wildlife reserve and slightly more often during winter. This was probably due to the lack of vegetation cover on the livestock farm, where they were not observed at all in winter. Blaum et al. (2007) found that increased shrub cover affects African wild cat abundance negatively, which could explain their absence on the livestock farm, but increased foraging activity during winter could also explain the observations (Herbst & Mills 2010). Black-backed jackal were never observed on the livestock farm, probably as a result of them being persecuted for livestock losses (Blaum et al. 2009), and therefore avoiding this site. Jackal are known to be quick to learn avoidance behaviour to persecution (Brand et al. 1995, Kaunda 2000).

Springhare observations played a large role in the switching patterns revealed in the total number of observations and the species observations. During the growing season, it was seen regularly on the livestock farm, but during winter this species was seen in highest frequencies on the wildlife reserve. When considering the feeding preference of springhare for short and green grass (Augustine et al. 1995, Peinke et al. 2016), it could explain the switching patterns observed as the wildlife area showed more vegetation productivity than the livestock farm in the winter season. Springhare are also sensitive to human disturbances and increased human activities on the

livestock farm (active management, hunting/chasing) could also result in a preference for the wildlife area (Butynski 1984, Yellen 1991, Peinke et al. 2016).

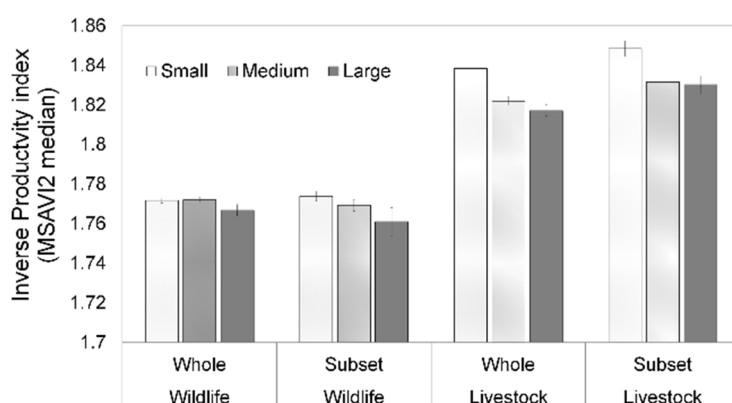
Overall, the general pattern seems to be that while many more species of these mammals were observed on the wildlife reserve, they were also observed to be foraging on the livestock farm, but mainly in the growing season when food and cover were more plentiful (Vinte 2015, Peinke et al. 2016).

### Number and size of burrows

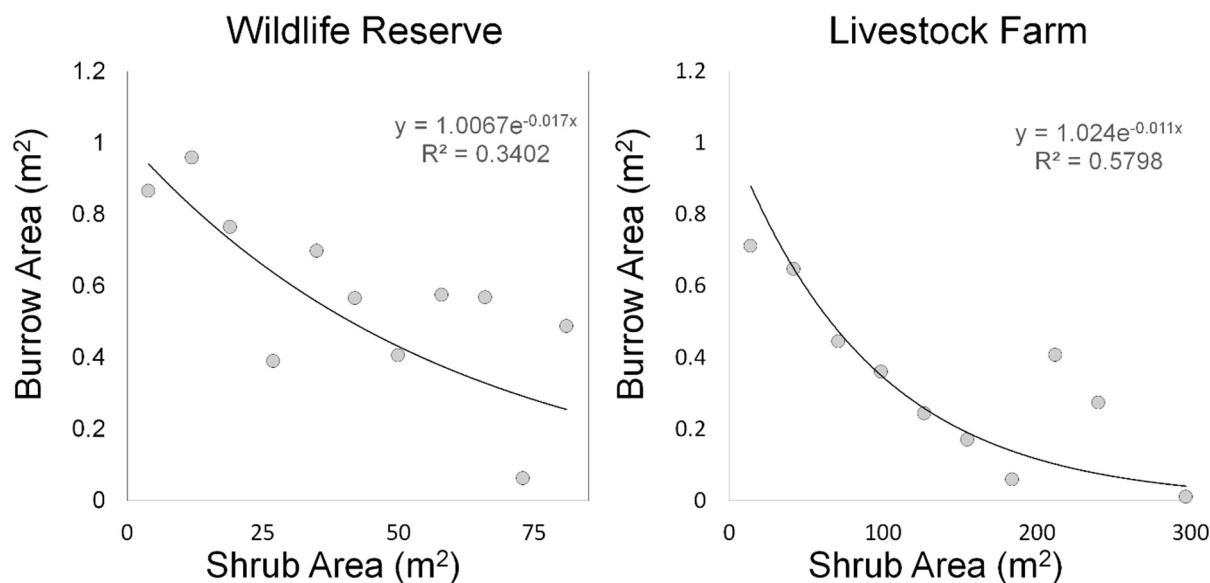
The aerial images produced for our study also revealed that not only were more species seen in the wildlife reserve, but more burrows were present there. Interestingly, the large burrows, classified in our study as between 31-100 cm, were in similar numbers on the two sites. These larger burrows are often home to mammals such as the bat-eared fox, Cape fox, pangolin, Cape porcupine, aardwolf, armadillo, and warthog (Skinner & Smithers 1990, Apps 2000, Stuart & Stuart 2013) and indeed this matches our sighting numbers well. While the aardwolf was much more often seen on the wildlife reserve, the bat-eared fox was seen in greater numbers on the livestock farm. This potentially suggests that the bat-eared fox has a relatively short foraging range (Mackie 1989, Pauw 2000). A rabies outbreak the previous year (2015) in the wildlife area (F. Reinhard pers. com.) also affected the bat-eared fox abundance (Maas 1993, Nel 1993). Furthermore, it indicates that the combined approach of using aerial images and night-time viewings as a tool to further our knowledge on these mammals is effective.

In contrast though, the high number of burrows recorded on the wildlife reserve were mainly those of the smaller size 8-14 cm in diameter, which provide homes to mammals such as the striped polecat, ground squirrel (*Xerus inauris*) and yellow mongoose (*Cynictis penicillata*). The striped polecat, being the only nocturnal species of the three, was observed only twice during the whole period of recording. This may be attributed to missed sightings as a result of small size, low preference for increased shrub cover (Blaum et al. 2007), or the recent rabies outbreak affecting abundances (Cumming 1982, Larivière 2002).

One possibly confounding effect to the greater presence of burrows on the wildlife reserve, was that shrub presence and area were higher on the livestock farm and in general, shrub presence was negatively related to burrow density. This relates to



**Figure 5:** Productivity (inverse) of vegetation surrounding mammal burrows of different size classifications: small (8-14 cm); medium (15-30 cm); large (31-100 cm). Data collated from aerial drone captured images, using the median MSAVI2 (productivity index) within 5 m distance from identified burrows. Lower values indicate higher productivity. Shown are values for two sites of different management type: Wildlife reserve and Livestock farm, and two sizes of survey areas "Whole survey" and "Less shrubby subset". Bars are standard errors where n equals the number of burrows per category (see Table 4).



**Figure 4:** Negative relationships between mean shrub area and burrow area at both sites of different management type. Data calculated from aerial drone images for “whole survey” area within superimposed cells 50m x 50m (196 cells). Note different scale on x-axis for livestock farm and wildlife reserve, with greater mean shrub area on livestock farm.

burrowing mammals preferring short grass or open areas for their burrows (Skinner & Smithers 1990, Augustine et al. 1995, Apps 2000). However, even taking this into account, by surveying a smaller subset of less shrubby area on the livestock farm, it was still found that there were more burrows present on the wildlife reserve.

### Productivity and shrubs

The aerial images revealed some important differences between the two study sites. In general, the wildlife reserve had higher vegetation productivity than the livestock farm. It is unclear whether this is a cause or consequence of the management type, whereby continual grazing by livestock in this marginal livestock farming area (Mogotsi et al. 2011) is likely to diminish quality fodder. However, grazing is not limited to the farm, and over-grazing was considered a concern on the wildlife reserve. Therefore, finding ways to keep productivity high may lie in helping to improve biodiversity in the area (Sala et al. 2000).

### Potential feedbacks between medium-sized mammals and habitat

The study found that burrow size was related to vegetative productivity. In the 5 m surrounding each burrow we revealed that productivity was higher around the larger burrows (31-100 cm diameter) than the small burrows (8-14 cm in diameter). It is well known that burrowing mammals are likely to provide benefit to habitats in the same ways as do other bioturbators, such as earthworms (Meysman et al. 2006, Eldridge & James 2009, Fleming et al. 2014).

It is often speculated that burrowing animals are likely to impact upon infiltration rates of water in the region (Reichman & Smith 1990, Avenant 2000), something of great importance in dry regions such as Namibia. Further turning of the soil will also cause aeration, which may help decomposition processes, and free up added nutrients (Reichman & Smith 1990, Gabet et al. 2003). Moreover, and particularly the case for the larger medium-sized mammals, defaecation close to or in the burrows will provide further nutrient sources. Therefore, provision of habitat conditions suitable for the mammals may feed back into providing more productive land.

However, despite the link between burrow size and productivity, our study did not reveal a direct link between burrows themselves and productivity. This result may be explained by the factor that burrows themselves cover an area, and cause disturbances to vegetation at their location (Butynski 1984, Augustine et al. 1995). At some intermediate point, these disturbances are likely to improve both productivity and plant species richness (Grime 1973), thereby providing great benefit to the ecosystem through the trophic levels. Further detailed analysis of the aerial images may provide more clues as to the benefit created by the mammals, by identifying and studying productivity indicators at various distances from burrows. Any findings would need to be supported by data collected at ground level, where paired localities can be compared for productivity response. Finding the mechanism for productivity increase, or the impact of the burrowing mammals, may be simpler in either a well-paired observational setting or, even more ideally, by using a controlled



experimental setting, such as exclosures (Ewacha et al. 2016).

Overall, our study comparing two neighbouring sites of different management types has provided some valuable insights into the movements, strategies and potential benefits of these mammals. Further research is needed to extend these findings to other areas of Namibia, and determine the precise mechanisms by which the burrowers may provide benefits to the land. However, our study has revealed and supported evidence that these under-studied mammals have the potential to play an important role in ecosystem functioning.

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**Appendix 1:** Species names and characteristics of all medium-sized mammals observed during night-time surveys on the two different habitat types: Wildlife reserve and livestock farm. Species are listed in order of frequency of sightings (Source of information: Skinner & Smithers 1990, Augustine et al. 1995, Apps 2000).

Species	Social structure	Size of territory (foraging range)	Preferred habitat	Breeding Season	Main Diet	Predators/Hunted
<b>Springhare</b> ( <i>Pedetes capensis</i> )	Social (2-6)	Up to 400 m from burrow	Short, open grasslands	Not seasonal	Grass seeds, stems, leaves, corms, roots & rhizomes. Seeds & seedlings of <i>Acacia</i> spp.	20 mammal species, 7 bird species, 4 reptile species, humans
<b>Aardwolf</b> ( <i>Proteles cristatus</i> )	Solitary	Area with about 3000 termite mounds. Depends on density of termites	Nama Karoo, Succulent Karoo, grassland & savanna biomes	Mating June-July. Breeding October-December	Nasute harvester termites	Black-backed jackal greatest enemy, humans
<b>Bat-eared fox</b> ( <i>Otocyon megalotis</i> )	Social foragers	1.5-2 km <sup>2</sup>	Short, open grasslands	October-December	Insectivorous; harvester termites	African wild dog, hyaena, leopard, cheetah, humans and raptors
<b>African wild cat</b> ( <i>Felis silvestris lybica</i> )	Solitary	About 4.3km <sup>2</sup> (territorial)	Wide habitat tolerance	September-March	Mice, rats, birds up to guinea fowl, scrub hare	Unknown
<b>Small-spotted genet</b> ( <i>Genetta genetta</i> )	Solitary	Unknown	Open, arid habitats, woodland savanna	Summer	Insects, mice, arachnids, birds & reptiles	Unknown
<b>Aardvark</b> ( <i>Orycteropus afer</i> )	Solitary	2.0-4.7 km <sup>2</sup>	Open woodland, scrub and grassland	Not seasonal	Formicid ants and termites	Large predators and humans
<b>Black-backed jackal</b> ( <i>Canis mesomelas</i> )	Solitary or in pairs	18.2 km <sup>2</sup>	Wide habitat tolerance	May-September	Omnivorous, insects, scrub hare, springhare, small antelopes, small livestock	Large predators and humans
<b>Cape fox</b> ( <i>Vulpes chama</i> )	Solitary	1.0-4.6 km <sup>2</sup>	Open grassland, scattered thickets, semi-desert scrub	August-October	Rodents, invertebrates, reptiles, birds, eggs, fruit	Large predators and humans
<b>Striped polecat</b> ( <i>Ictonyx striatus</i> )	Solitary	Unknown	Wide habitat tolerance	Summer	Insects and mice	Large predators

# The influence of two levels of debushing in Namibia's Thornbush Savanna on overall soil fertility, measured through bioassays

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## ABSTRACT

A healthy and productive rangeland depends on well-functioning ecosystem services such as effective cycling of water and nutrients. After rangeland has degraded, bushes may encroach in nature's attempt to restore water and nutrient cycling. When bush encroachment is addressed by debushing, with harvested bush wood sold off the land, then nutrient cycling is disrupted and soil fertility is likely to decline. Former debushing activities on different parts of farms of the Cheetah Conservation Fund in central Namibia provided the opportunity to assess the influence of debushing on overall fertility of soil. Sites were selected on the farm representing nine examples of each of uncleared, partially cleared and totally cleared land. The debushing had taken place at different times, varying between two and 13 years previously. Soil was collected from each of these 27 sites and subjected to bioassay by growing barley (*Hordeum vulgare*) and *Moringa oleifera*. Seedling emergence and height at five weeks for both species were greatest in uncleared soil and lowest in totally cleared soil, indicating the loss of soil fertility as debushing intensifies. There was no evidence of restoration of soil fertility, even 13 years after debushing. Nutritious grass is unlikely to grow well after debushing, and more bush is likely to regrow in nature's attempt at restoring fertility over the long term. If faster restoration is sought, then the full spectrum of minerals removed in harvested wood should be replaced on the land.

**Keywords:** bioassay; bush encroachment; debushing; Namibia; soil fertility; Thornbush Savanna

## INTRODUCTION

Much of the rangeland in Namibia has experienced degradation over past decades, with interference in healthy water and nutrient cycling (MAWF 2012). Only a portion of the nutrients grazed by animals get recycled to the soil through dung and urine, with proportions varying greatly between the different elements. The remainder of the nutrients are removed from the ecosystem if milk or animals are sold off the farm. During the first half of the 20<sup>th</sup> century, Namibia supported a thriving dairy industry, after which live animals took over as the main export through the farm gate. Although the shift from dairy to beef was brought about by imposition of market regulations (Lau & Reiner 1993), it is likely to also reflect decline in quality of grass resulting from both poor grazing management and loss in soil fertility.

In the past, many farmers supplemented their animals' feed with bone meal, which returned largely the elements of Ca and P, but this practice has been banned since 1998 for veterinary reasons (Kaurivi 2013). Currently used supplements return some elements, but unlikely in balance with those removed. The emphasis is usually placed on lick

supplementation of P and N, even though the latter can be freely obtained from the air, particularly if a healthy root system feeds free-living nitrogen-fixing microbes in the soil through exudates (Lovel 2014).

Rock salt may return a portion of the micro-elements, which animals tend to seek out by selectively licking bands within the salt block where the micro-elements presumably settled out during crystallisation. However, in the process, animals also tend to consume large amounts of Na and Cl, which may contaminate the soil when excreted in dung and urine, causing physical, chemical and biological harm to the soil (Andersen 2000). This might be partly responsible for the observations of a general accumulation of Na in overgrazed topsoil (Gröngroft et al. 2010) and higher Na in soil of more recently burned woodland (Nghalipo 2016).

Bush encroachment is one of the symptoms of rangeland degradation, caused by a variety of factors including herbivory, tree harvest, fire, carbon dioxide and soil nutrients (Zimmermann et al. 2008, Van Auken 2009) and may reoccur in cyclical patterns (Cunningham 2014). The interplay between these factors is pointed out by Devine et al. (2017), such as

soil fertility being both a cause and an effect of woody cover. Farmers often respond to the symptom of bush encroachment by debushing. However, this removes even more nutrients from the system than grazing, if the harvested bushes or their products are sold off the farm, yet there is rarely any attempt to return the lost elements to the debushed land. Many farmers leave some of the harvested material, usually the thinner branches with most of the leaves, on the land (De Klerk 2004, Joubert & Zimmermann 2017). However, this is insufficient to compensate for the elements lost in the larger portion of harvested bush that is removed.

Since debushing had taken place between 13 and two years previously on different parts of the Cheetah Conservation Fund (CCF) farms, and at two different levels of partial and total clearing of bushes, it provided the opportunity to examine the effect of these factors on soil fertility. Sites selected for debushing were expected to have adequate harvestable bush biomass and to be easily accessible for manual harvesting.

Overall fertility of soil can be conveniently measured by bioassay (Olsvig-Whittaker & Morris 1982), which is a measurement based on the responses of living organisms planted in that soil. Species of plants grown in bioassays should be easy to grow in a range of soil conditions. Two species were selected for this study, barley (*Hordeum vulgare*) and Indian moringa (*Moringa oleifera*). The reason for selecting a grass and a tree is that differences can be expected in their root exudates and rhizosphere interactions used for obtaining nutrients. For example, dicots and non-graminaceous monocots tend to exude organic acids

that chelate Fe, whereas grass roots tend to exude phytosiderophores with high specificity for certain metals (Marschner & Rengel 2003).

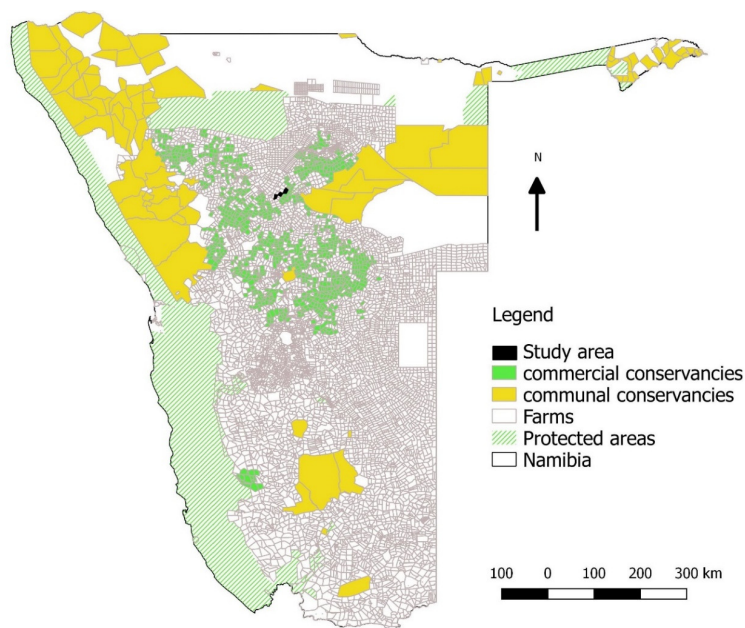
## METHODS

The study area is situated approximately 40 km east of Otjiwarongo in Namibia's Thornbush Savanna (Giess 1971), on four commercial farms (Cheetah View, Boskop, Elandsvregude #367 and Osonanga #368) owned by the CCF, in the Waterberg Conservancy (-20.433 S, 17.078 E; Figure 1). Sites with different post-harvest ages ranging from 2-13 years (mean: 5.6 partially cleared, 4.3 totally cleared) were previously harvested by CCF bush project to promote ideal habitat structure for open-savanna-adapted species, while processing the harvested wood into blocks sold for fuel (CCF Bush Pty. Ltd. 2017). About 89% (n=16) of the harvested sites received aftercare by applying a Picloram active herbicide, commercially known as Access, on the harvested stumps to prevent bush regrowth.

The area receives a mean annual rainfall of 400 mm and experiences three major seasons including the wet-hot (January-April), dry-cold (May-August) and dry-hot (September-December). The vegetation type is classified as semi-arid Thornbush Savanna, dominated by encroaching woody species such as *Acacia (sensu lato) mellifera* subsp. *detinens* and *Dichrostachys cinerea* (Giess 1971). The Waterberg Plateau, a 4,100 km<sup>2</sup> sandstone uplift on the eastern periphery of the study area, is the dominant geological feature of the region. The study area falls within the Damara sequence geologic stratum, the oldest in the Waterberg region. The soils, which are

of sandy loam texture in all study plots, fall into two main associations indicated in Figure 2, the Eutric Regosols and Chromic Cambisols (Mendelsohn et al. 2002). The former is expected to be more fertile, as implied by the name "Eutric", while "Chromic" implies brightly coloured. The topography of the farms is generally flat with slight undulations; consequently, rainfall run-off is slow and there are no permanent river systems on the farms. A number of man-made semi-permanent water reservoirs (earth dams) were developed to provide wildlife and livestock with drinking water.

Sites were selected on the CCF farms (Figure 2) representing nine examples each of uncleared, partially cleared and totally cleared land (Figures 3-5). On uncleared



**Figure 1:** Location of study area farms of Cheetah Conservation Fund

land, the overall density was approximately 1,300 bushes/ha. Grass had not been harvested from cleared sites for hay, as is commonly practiced elsewhere. Although the sites were scattered over two soil associations (Figure 2), an equal number of sites from each debushing treatment occurred in each soil association. The debushing had taken place at different times, varying between two and 13 years previously. A composite soil sample was gathered from each of these 27 sites, by collecting three subsamples using a spade to a depth of approximately 30 cm and mixing together in a bucket. The subsample sites were located randomly in each site using the Hawthstools random extensions run in ArchMap GIS 10.3 (Esri, Redlands, CA). A systematic grid of 100 m<sup>2</sup> was laid over each site and random selection was constrained on these grid points. Each mixed soil sample was divided into 27 growing bags of approximately 500 ml capacity, 15 to serve as replicates for barley and 12 as replicates for moringa, the latter being limited by availability of seed.

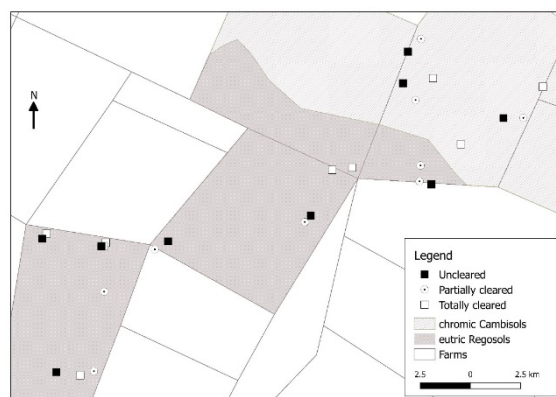
Each growing bag was sown with either two moringa seeds or three barley seeds and kept moist, for later thinning to one seedling per bag. Bags from which seedlings emerged were recorded daily over two weeks, and height of the tallest stem or tiller per bag were recorded weekly until the fifth week (Figure 6).

The first assessment was of the average treatment effect of total clearing and partial clearing on the percentage of bags with seedlings sprouting and surviving over a five-week period, relative to bags with soil from uncleared sites. P-values and confidence intervals were estimated using randomisation inference (Gerber & Green 2012), with clustering at the site level. Nearly equal results were obtained using OLS regression and Chi-Squared tests.

Secondly, we estimated the average treatment effect of total clearing and partial clearing on the height of sprouted seedlings, standardised to the uncleared mean, using ordinary least squares regression. The reported results include non-sprouting seedlings in this estimate, but are robust to their exclusion.

Thirdly, we analysed changes in seedling growth over time since debushing. To test the hypothesis that initial improvements in soil fertility are countered with subsequent declines, we fitted a quadratic polynomial function to allow for a curvilinear effect of years-since-clearing on mean seedling height.

For all of the above, we included controls for soil association and species, and tested for soil and species-treatment interactions.



**Figure 2:** Locations of sites on Cheetah Conservation Fund farms from where soil samples were collected, in relation to soil associations.

## RESULTS

### Seedling Sprouting

Seedlings were significantly ( $p < 0.05$ ) less likely to emerge and survive up to five weeks in soil from both fully de-bushed and partially de-bushed sites relative to seedlings in soil from uncleared land. Analysis of heterogeneous treatment effects revealed no statistically significant difference between treatment effects on barley versus moringa seedlings. If soil association is disregarded, there was a 20% and 31% decrease in seedling survival in fully cleared soil for barley and moringa, respectively, and 13% and 15% decrease in seedling survival in partially cleared soil. When controlling for soil association and plant type, the results shift to 12% decrease for partially cleared and 20% decrease for totally cleared sites. However, soil association also had a significant ( $p < 0.05$ ) but lesser effect than level of debushing on seedling survival. Figure 7 indicates the effect of soil association on number of surviving seedlings of both species combined, with 30% and 22% decrease in seedling survival in fully cleared soil for Ochric Cambisol and Eutric Regosol respectively, compared with uncleared soil, and 16% and 13% decrease in seedling survival in partially cleared soil for Ochric Cambisol and Eutric Regosol respectively.

### Seedling Growth

Decreased fertility is further reflected in the comparison of mean heights of the plants grown in soil samples. The effect of level of debushing was greater than that of soil association for both barley and moringa, which were standardised by height to reflect soil fertility, since moringa grew slower than barley yet exhibited similar patterns. On average, barley seedlings were 18% shorter in cambisol than regosol, and moringa seedlings were 9% shorter ( $p < 0.05$ ). Figure 8 combines barley and moringa seedling heights to indicate soil fertility relative to the





**Figure 3:** *Uncleared land*



**Figure 4:** *Partially cleared land*



**Figure 5:** *Totally cleared land*



**Figure 6:** *Bioassay of sampled soil using barley (left) and moringa (right)*

most fertile soil in uncleared sites. The drop in fertility from uncleared to partially cleared sites was 28% on Cambisol and 17% on Regosol, while the drop in fertility from uncleared to totally cleared sites was 34% on Cambisol and 29% on Regosol. The difference in soil fertility between fully and partially cleared soils is erased when we control for soil association and species in regression analysis. (In OLS regression controlling for species and soil association, these results are 3 cm decrease for barley in partially cleared and 4.2 cm decrease for barley in fully cleared soils, and a 1.3 cm decrease for moringa in partially cleared and a 1.9 cm decrease for moringa in fully cleared soils.) (Point estimates were obtained using OLS regression and p-values using randomisation inference to account for non-normal distribution of the dependent variable.)

Interaction effects between soil association were small and generally not statistically significant at standard levels. However, the interaction between plant type and clearing level were substantially and statistically significant: the negative effect of full clearing increased by 2.2 cm for barley relative to moringa, and the negative effect of partial clearing increased by 1.7 cm ( $p < 0.05$  and  $p < 0.1$ ).

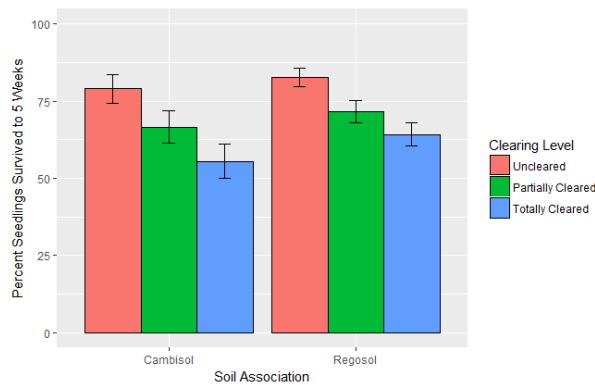
### Change over time since debushing

The data in Figure 9 are noisy and far less clear than those presented above on level of debushing, so no firm conclusions can be drawn from them. The expectation of increase in soil fertility immediately after debushing was not detected by these results that start at two years after debushing. An immediate increase in soil fertility could be expected from root exudates released in response to harvest. From decomposition of the increased dead leaves, the expected increase in soil fertility would be over the next rainy season. Decomposition of larger roots may continue for several years thereafter. From new inputs provided by regrown bushes and trees, the re-establishment of soil fertility is expected to take several decades, and the data in Figure 9 fail to show any hint of this having started yet 13 years after debushing.

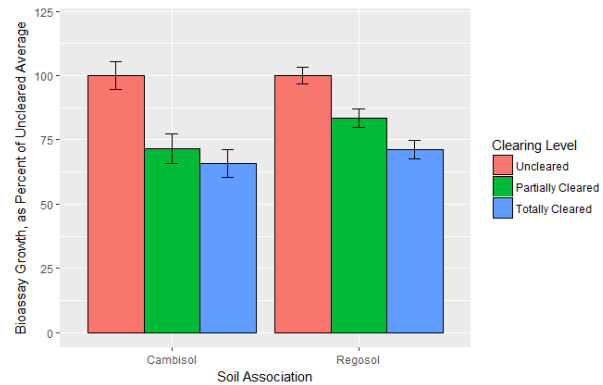
### DISCUSSION

The results indicate a decline in soil fertility from debushing that took place at different times between two and 13 years previously in both soil associations. This is not surprising, given the removal of harvested bushes for sale, without return to the soil of minerals contained in those harvested bushes. There is the possibility that large herbivores may also contribute





**Figure 7:** Percent seedling emergence and survival up to five weeks for barley and moringa combined, as influenced by level of debushing and soil association. Error bars represent standard errors



**Figure 8:** Mean seedling growth at five weeks as percentages of the mean from uncleared sites, to represent relative fertility in soil with different levels of debushing on each soil association. Error bars represent standard errors

to fertility loss and gain, if they feed more in certain areas and spread their excrement more in other areas. However, this was not investigated by the current study.

Andrews (2008) points out that as fertility of the topsoil declines, whether through sale of animals or wood or both, more woody plants grow, which bring up minerals from the subsoil. If the established bushes are given the opportunity to replenish soil fertility over decades, then grasses eventually regain their competitive advantage and bushes decline once more. However, if bushes are harvested, then fertility declines further in a vicious circle as bushes regrow with more vigour.

Various authors offer evidence of woody plants leading to higher soil fertility. For example, Sandhage-Hofmann et al. (2015) found higher soil fertility in the bush encroached zone farther from water points of continuously grazed communal land, which they attribute to the increase in *Acacia* bushes. Mills and Fey (2004) found poorer soil quality where succulent thicket had been transformed by goats to savanna. Rothauge et al. (2007) found that grasses occurring in canopied habitats had a significantly higher nutritive value than those occurring in the open, even if they were of the same species. Hagos and Smit (2005) found a gradient in soil nutrient status, from higher near the stem of free standing *Acacia mellifera* trees, to lower further away from the canopy in open ground. Blaser et al. (2014) found that encroachment by *Dichrostachys cinerea* resulted not only in higher soil levels of C and N that cycle through the atmosphere, but also of P that cycles more directly between biota and soil.

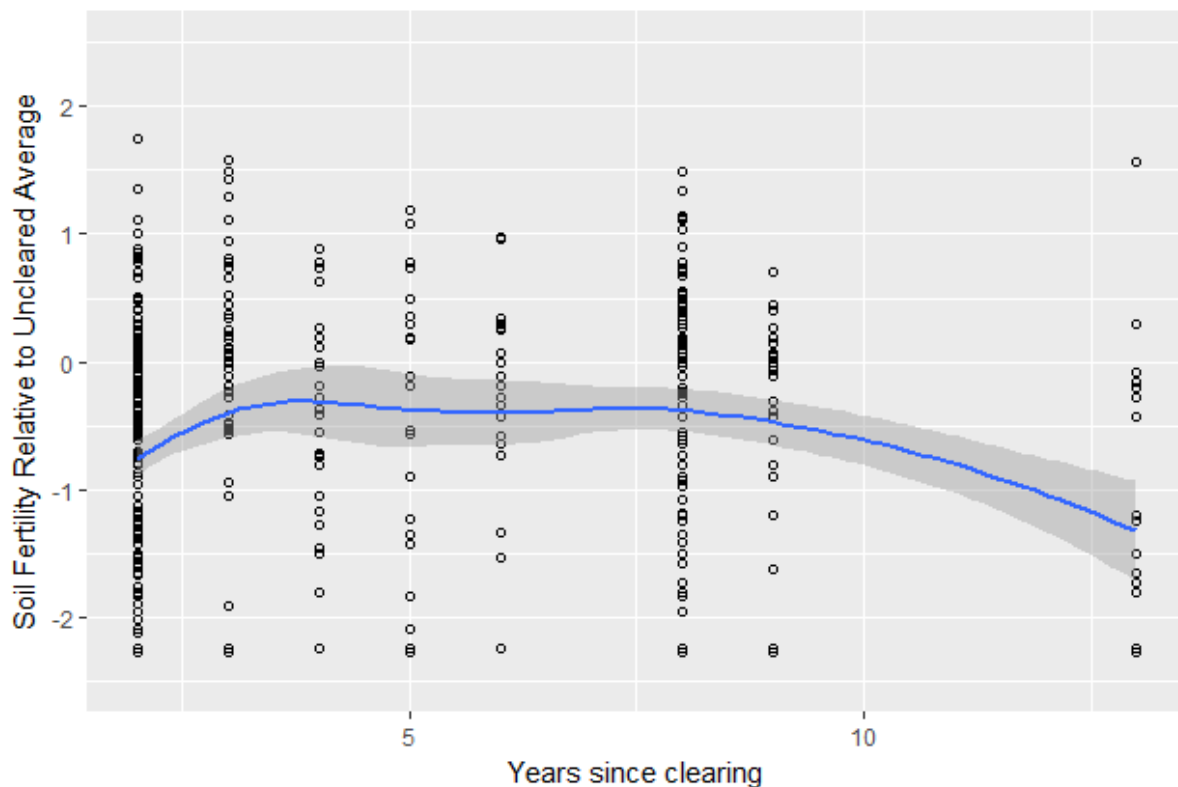
Although credit for the ability of woody plants to raise soil fertility is usually given to annual leaf fall (Nair et al. 1998), biological soil crusts that tend to develop under protection of bushes (Dougill &

Thomas 2004) and termites that feed on fallen twigs and branches (Joquet et al. 2007, Barthès et al. 2015), the less visible role of roots should not be overlooked. Fine roots of trees seem to contribute more to nutrient cycling than leaves, due to their faster release of both N and P (Jose et al. 2000), and root exudates contribute significantly to the pool of soil carbon that feeds the soil food web (Grayston et al. 1997, Clapperton et al. 2003, Bais et al. 2006, Zhang et al. 2017).

The proposition that lower fertility leads to an increase of woody plants is offered by a few authors. The encroachment of leguminous bushes on rangeland in Arizona between 1903 and 1943 is attributed by Albrecht (2005) to the removal of organic fertility by cattle. When analysing data from a wide range of soils over Namibia and South Africa, Mills et al. (2013) found a peak in woody plant cover on soils of intermediate fertility, with lower cover in soils of both low and high fertility, and the latter linked to aridity. They conclude that the abundance of woody plants is partly determined by soil nutrient status, with fertile soil favouring the growth of grasses. After examining tree density in a rangeland subjected to long-term fertilisation trials followed by protection, Mills et al. (2017) found that soil Mg, B, Mn and Cu most likely affected tree abundance, in particular the ratio Mn/Cu, pointing out the “global significance (of these findings) for the practical control of invasive woody plants.”

## CONCLUSION

Combining the information presented in the above discussion, it seems unlikely that the grass growing on debushed land will be very nutritious from the second year onwards, due to the lower soil fertility found even in sites debushed two years previously. The spectacular flush of grass that often follows debushing is usually short-lived, probably declining



**Figure 9:** Seedling growth over time since debushing

as organic material from dead leaves and twigs become depleted. According to Rothauge (2014) “Farmers report much more grass after bush control, but not much better-quality grass”. The expectation would be a rise in soil fertility soon after harvesting, due to release of nutrients from root exudates and decaying leaves, followed by a decline due to lack of further inputs from smaller parts of bushes. However, this study provides a slight hint that the initial decline two years after debushing may be followed by a gentle rise in fertility for a few more years before declining again (Figure 9). If this is the case, it might be explained by slow decomposition of larger bush roots contributing to fertility long after the fast decomposition of increased dead leaves, twigs and root exudates, probably during the first rainy season after the debushing. Many years later, a slow rise in fertility could be expected when regrown or newly established bushes are large enough to contribute significant inputs. However, this natural restoration of fertility through regrowth of bushes seems to be extremely slow, with no evidence of restoration yet after 13 years in this study. The use of arboricide on stumps of harvested bushes is likely to have contributed to this lack of restoration of fertility.

If farmers seek faster restoration of fertility, then the full spectrum of minerals removed in harvested wood should be returned to the land. Animals could only assist with this if they were to do most of their grazing

elsewhere, while depositing much of their dung and urine on the debushed land. This could be achieved over small areas by overnight kraaling (Powell et al. 1996, Sibanda et al. 2016), to effect a transfer of nutrients at the expense of the grazed land. Alternatively, animals could be supplemented with minerals for distributing over the land in excrement, as was achieved by Joseph et al. (2015), where biochar fed to cattle was excreted in their dung and taken into the soil by dung beetles. Ocean products tend to be rich in the full spectrum of minerals (Walters 2005) and, if processed into dry form, could be transported inland from the coast at lower cost than bulkier supplements. The abundance of kelp along Namibia’s coast provides the opportunity, if harvested sustainably, to supplement animals with kelp meal. This not only provides animals with a wide spectrum of micro-elements (McHugh 2003), but also a diversity of bioactive compounds with performance-enhancing benefits for animals (Evans & Critchley 2014) and great potential to reduce methane emissions (Kinley et al. 2016).

If debushing is to take place, then partial debushing will allow slower loss in soil fertility than total debushing. The pattern of partial debushing could influence the outcome. If bushes are cleared in strips, then the strips of uncleared bushes could serve as windbreaks and fertility generators, while the cleared strips could provide alleys for animals to graze and

run, and for farmers to drive through should they have the need. In very sandy soil where rainwater infiltrates straight into the soil, the alignment of strips could be at right angles to the prevailing, or most destructive, wind to reduce evaporation and erosion. On soil that experiences runoff during intense rain, the cleared strips should be aligned on contour (Bruwer 2014) to encourage infiltration and improve water cycling. This could be further facilitated by placing some of the harvested bushes as filter lines on contour, to favour the growth of grass underneath and mimic the natural pattern of banded vegetation (Tongway et al. 2001). If harvested wood is burnt as an energy supply, then ideally the ashes that remain after the burn should be returned to the land from where the bushes were harvested (Pitman 2006), which could effectively be aligned on contour (Bruwer pers comm.).

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# Biodiversity zoning in the Greater Fish River Canyon Landscape in southern Namibia

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## ABSTRACT

An objective process of biodiversity zoning is presented using the Greater Fish River Canyon Landscape in southern Namibia (partially overlapping with the Ai-Ais – Richtersveld Transfrontier Park, which stretches across both sides of the Orange River) as an example. Using satellite imagery, broad habitat units were mapped, and plant species lists were compiled for these units, based on local, national and regional distribution data, published information and targeted field work. Formal protection, red-list status, and an index of rarity based on distribution range and endemism were used to determine a rating for species of conservation importance. These ratings were summed per landscape unit, providing an overall rating for each unit. The topographically heterogeneous landscape in a transitional area between southern Africa's winter and summer rainfall regimes was delineated in 32 landscape units. A total of 835 plant species was included in this study, 265 of which were defined as having particular conservation importance. The systematic incorporation of level of endemism and an index of rarity based on range size, facilitated a fully objective process of biodiversity zoning. The resulting zoning presents a relative measure of prioritisation for conservation, sufficiently flexible to be adapted to the species richness and data range in a particular study area and is therefore site-specific. Although a large part of the study area is formally protected in state-protected areas and adjoining private nature reserves, nearly 50% of the most important area in the western Succulent Karoo section of the landscape is on private farmland and townlands of the mining town Rosh Pinah with no formal protection. Also, exploration and small-scale mining take place along the Orange River. All these factors pose a threat to some of the rarer plants.

**Keywords:** biodiversity, endemism, zonation, Gariep, mountain flora, rarity, range size, Succulent Karoo biome, transfrontier park, Namibia, Fish River Canyon

## INTRODUCTION

Managing biodiversity in areas where inventories are incomplete, fragmented or biased towards certain groups of taxa is a challenge worldwide (Reid 1998, Cooper & du Plessis 1998, Fuller et al. 2010, Franklin et al. 2011). Where biodiversity information and data-collection time are limited, expert opinion is often used to prioritise areas of importance for biodiversity conservation (Cowling et al. 2003). While this approach has its merits in many situations (Reyers et al. 2007), it lacks objectivity, resulting in a call for more consistency in biodiversity assessments (Landi & Chiarucci 2010). This paper outlines an objective process for biodiversity zoning at landscape level, using the Greater Fish River Canyon Landscape (GFRCL) in Namibia as an example.

The GFRCL is positioned in the Gariep Centre of Endemism in southern Africa (van Wyk & Smith 2001), in a transitional area between summer and winter rainfall. On regional maps a biome boundary is depicted to run through the Huns Mountains in the

centre of the GFRCL (Snijman 2013) or pockets of Succulent Karoo are delineated in a Desert- Nama Karoo matrix (Rutherford 1997). While such pockets of 'biome islands' are also found across other southern African biome boundaries (Potts et al. 2015), this transitional position is expected to result in high plant diversity in this area (Burke 2015), as predicted for ecotone areas elsewhere (Kunin 1998, Kark 2013, Potts et al. 2015).

Administratively the GFRCL contains the Namibian section of the Ai-Ais – Richtersveld Transfrontier Park, which is jointly managed by Namibia and South Africa. While a fair amount of information on biodiversity of the Richtersveld is available (e.g. Williamson 2000, Duncan et al. 2006, Hendricks et al. 2007, Young & Desmet 2016), the Namibian part of the park is poorly studied. As part of a park-neighbour initiative in Namibia, inventories of the flora were undertaken to enable biodiversity zoning and to support the development of management guidelines. The Greater Fish River Canyon Landscape was created as part of a Global Environment Facility – Namibian government

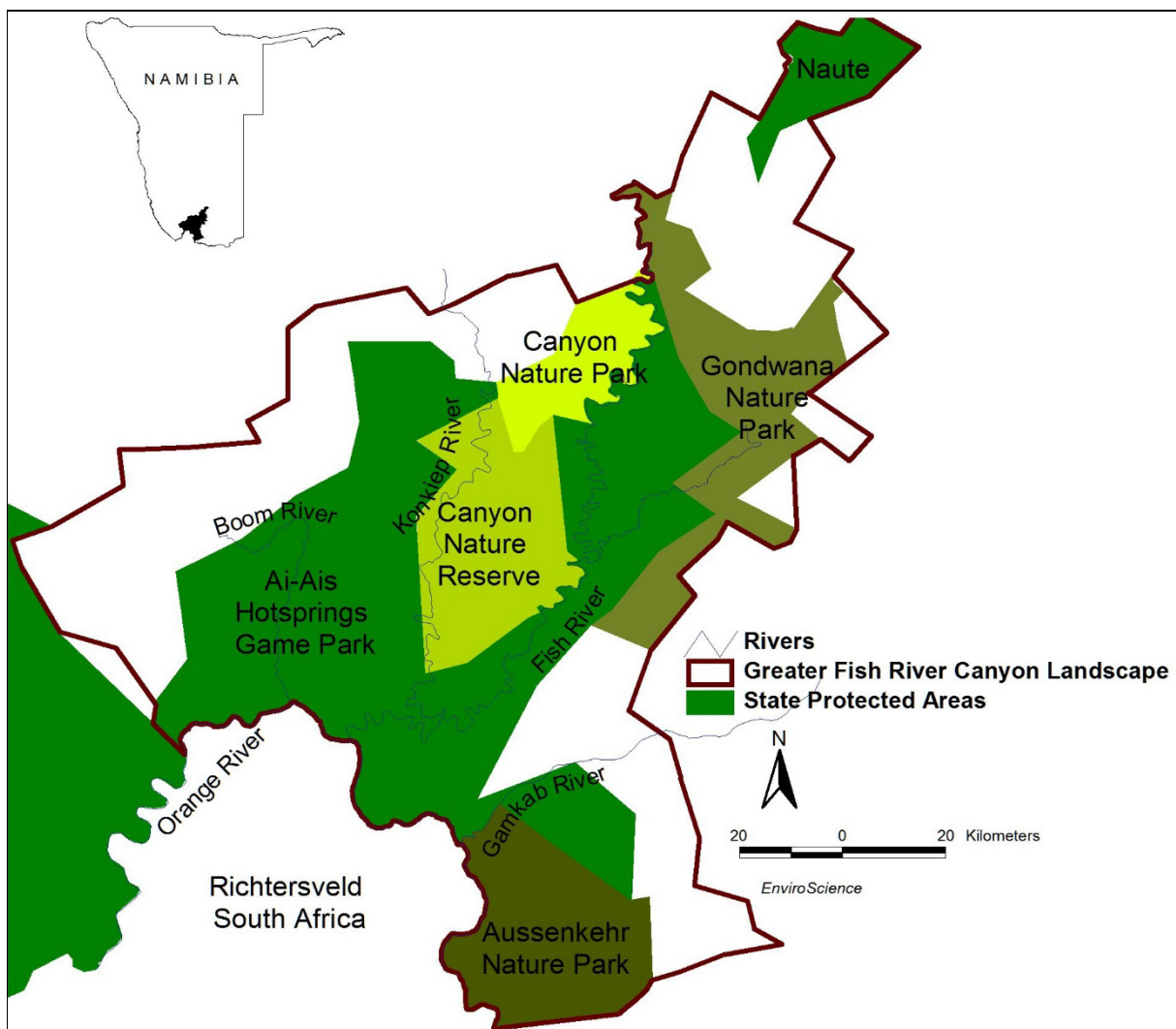
initiative to address integrated land management of protected areas and their surroundings. The landscape was developed around one of Namibia's prime tourism attractions, the Fish River Canyon, and includes the Ai-Ais Hotsprings Game Park, several private nature reserves and farming areas adjoining the park (Ministry of Environment and Tourism 2011) (Figure 1). The purpose of this paper is to present the process and the results of zoning the biodiversity of the Greater Fish River Canyon Landscape. Plants are used as indicators for biodiversity because (1) they are the basis for most food chains, (2) compared to other taxa in Namibia (except for birds), information on distribution is readily available and (3) they are immobile and therefore indicate habitats at a small scale. They can therefore be considered a proxy for other biodiversity. This is in line with many conservation planning initiatives around the world where plants are often used as indicators (e.g. Brudvig et al. 2009; Egoh et al. 2009) and particularly in areas where the flora is known to be of great conservation

importance, such as the Gariep Centre of endemism and the Cape region (van Wyk & Smith 2001; Rebelo et al. 2011).

## METHODS

### Study Area

The study area is positioned just north of the Orange River in Namibia (Figure 1) and is a varied landscape of mountains, gorges, valleys, plateaus and plains. It includes the Huns and Namus Mountains in the west, the lower Fish River, the Gamkab plains and the Orange River valley, with the Fish River Canyon as a popular tourist destination. The southern boundary is the perennial Orange River. At the interface of three major Southern African biomes, the Succulent Karoo, Nama Karoo and Desert biomes (Rutherford 1997), and floristically part of the Gariep ecogeographic unit in the Extra Cape Flora (Snijman 2013), the area is rich in biodiversity.



**Figure 1:** Position of study area and extent of Greater Fish River Canyon Landscape (GFRCL) in Namibia. White areas in the GFRCL are commercial farmland.



The underlying geology ranges from the oldest rocks in Namibia to recent Quaternary sediments. Some 600 to 550 million-year-old sedimentary rocks of the Nama Group are exposed in large parts of the study area. Gneisses, granites and other metasedimentary rocks of the Namaqua Metamorphic Complex (1,800-100 Ma [Million years]) and the Gariep Group (800-500 Ma) are also present, as well as sedimentary rocks and volcanic intrusions of the much younger Karoo Group (345-140 Ma) (Swart 2008). The complex geology and geomorphological forces created a rugged landscape, with altitudes ranging from less than 100 m above mean sea level at the deeply incised Orange River to 1,654 m at the Namuskluft peak. Soils are poorly developed regosols or arenosols.

The climate is arid and mean annual rainfall ranges from 50 to 100 mm, increasing along a south-west to north-east gradient. Summer and winter rains can occur. Mean annual temperatures range between 16°C and 20°C, increasing along a west-east gradient. The south-eastern corner borders one of the hottest areas in Namibia where temperatures over 36°C are measured during the hottest months (Mendelsohn et al. 2002). The temperature range at the Canyon Roadhouse, for example, in the north-eastern part of the study area, was 0-38°C during 2017. These broad bioclimatic gradients are modified locally by mountainous terrain. The higher reaches receive more rainfall and temperatures are lower here. Fog occasionally drifts east from the Atlantic Ocean along the Orange River valley and reaches the western part of the study area. In summary, this is an area of high diversity in landscapes and climatic extremes.

The vegetation is sparse and consists largely of leaf-succulent dwarf shrubs (mainly the genera *Amphibolia*, *Antimima*, *Eberlanzia*, *Lampranthus*, *Leipoldtia* and *Ruschia*, as well as *Zygophyllum*), herbs and grasses in the west and south (Figure 2). Shrubs, grasses and herbs are dominant in the east. On the eastern plains prominent shrubs are *Euphorbia gregaria* and *Rhigozum trichotomum* (Figure 3). Grasses, largely represented by *Stipagrostis* species, are more prominent in the north and east. Trees and tall stem-succulents e.g. *Aloidendron dichotomum*, *A. pillansii* (formerly *Aloe*) and *Pachypodium namaquanum* provide prominent landmarks, mainly in the west of the study area (Figure 2), but are widely dispersed, except along rivers. Here riparian thickets with trees such as *Acacia (sensu lato)* karroo, *Salix capensis*, *Searsia pendulina* and *Tamarix usneoides* grow along the permanently-flowing Orange River. Trees and patches of reeds (*Phragmites australis*) also grow along the Fish River. Mountain slopes support very diverse vegetation with many range-restricted species such as *Arctotis fruticosa*, *Antimima dolomitica*, *Caesalpinia merxmuelieri*, *Conophytum taylorianum*, *Eberlanzia clausa*,

*Moraea thermarum*, *Portulacaria armiana*, *Ruschia ruschiana* and *Schwantesia loeschiana*.

The eastern and north-eastern part of the study area is covered by extensive plains and a section of the Gamkab basin (Figure 3d). The north-central part provides a mosaic of plains and rocky terrain, but unlike in the west and east, these plains are positioned at higher altitude between 1,000 and 1,400 m above mean sea level. The Fish River Canyon incises deeply into the plateau and rocky terrain in the central-eastern part of the study area, resulting in a landscape characterised by deep valleys and steep slopes flanked by plateaus at different elevations.

### Approach to the study

The study built onto biodiversity zoning at landscape level for the Ai-Ais Hotsprings Game Park (Burke 2011) and was undertaken to fill data gaps in a rapid assessment and biodiversity zoning to inform management. Plants were selected as biodiversity indicators because vegetation can readily be used for landscape-level mapping, and information on plant distribution is electronically available in Namibia, although not complete and to some extent at a coarser scale than required for this study.

Available plant distribution data (at quarter degree square resolution, i.e. 15 minute intervals on a latitude-longitude grid) from Namibia's National Botanical Research Institute's (NBRI) specimen database (WIND) were used to guide the fieldwork, in order to fill evident data gaps (De Ornellas et al. 2011). Fieldtrips were undertaken in summer (19 April - 4 May 2013) in the eastern part of the study area and in winter (31 September - 9 October 2013) in the western part. A Google image was used to delineate homogenous mapping units, based largely on habitat (Cowling & Heijns 2001), and mostly at 1:100,000 scale, but high resolution Google images obtained by the Namplace project were used to clarify some mapping boundaries. Field data from the 2011-2013 surveys were combined with the specimen data from WIND, data from the Karios observatory of the BIOTA project (Jürgens et al. 2010), publications (Burke 2004), web-based databases (Craven & Kolberg 2017) and data collected for other biodiversity assessments. Plant species lists were then compiled for each landscape unit, and although not entirely complete, these species lists provide the best approximation and serve as relative indicators. The conservation importance of each plant species was rated based on the criteria endemism, red-list status according to IUCN (Loots 2005, with updates to 2013), narrow-range (as an indicator for rarity) and legal conservation status (Table 1), in this study collectively referred to as "species of conservation importance". The nomenclature follows Klaassen & Kwembeya



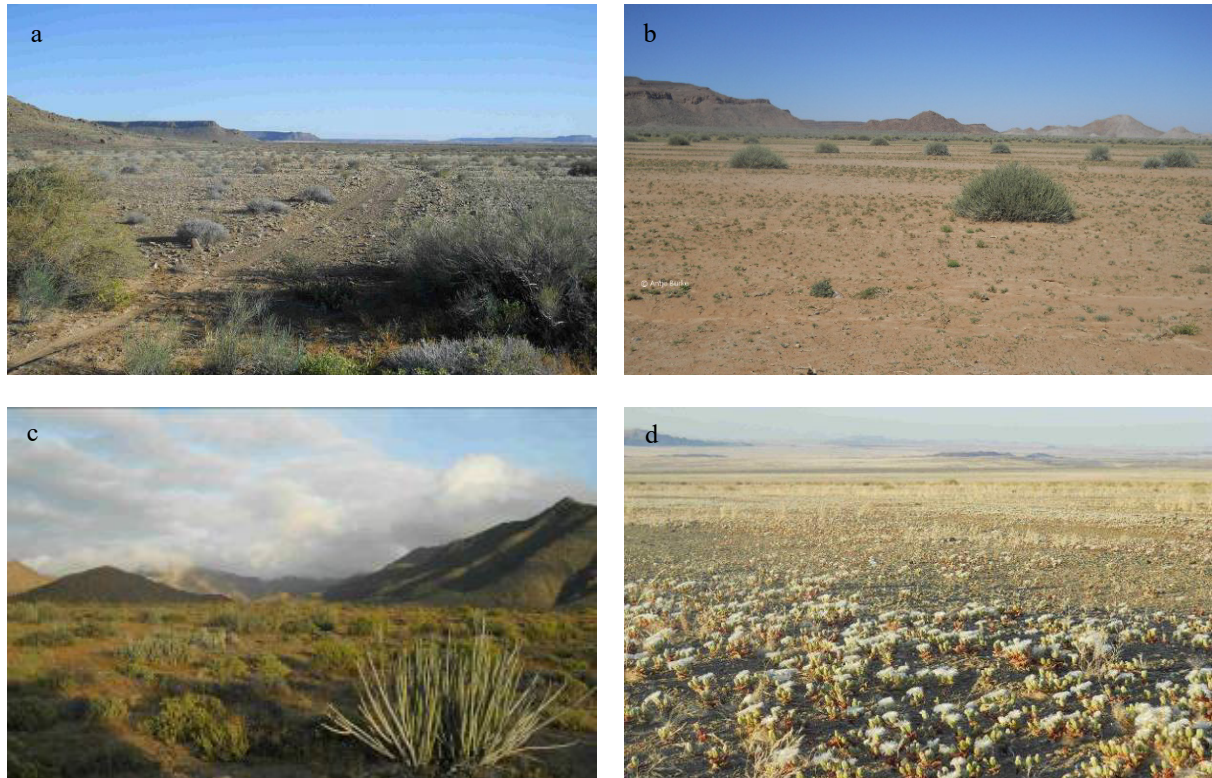
**Figure 2:** Mountain landscape units in the Greater Fish River Canyon Landscape: (a) Granite koppies in the south-east of the Aussekenhr Nature Park support a surprising number of plant species in the driest and hottest part of the study area, (b) steep canyons dissect the limestone and shale layers of the northern Huns Mountains, here with *Pelargonium spinosum* in the foreground (c) the well-vegetated slopes of the western Namus Mountains advertise Succulent Karoo. Inset, left to right: *Aloidendron ramosissimum*, *Tylecodon paniculatus* and *Pachypodium namaquanum*.

(2013), with updates for new species (Steiner 2006, Goldblatt & Manning 2013, Kativu & Bjora 2016).

Endemism and narrow-range were introduced as criteria additional to the IUCN red-list classification, because many endemic species with a very restricted range in Namibia are not legally protected. For the purpose of this study narrow-range (as an indicator of rarity) is based on the number of recorded distributions in quarter degree squares (1 qds = 15 minute intervals on a longitude-latitude grid). One quarter degree square (qds) is approximately 625 to

650 km<sup>2</sup> in Namibia (i.e. three qds amount to 1,875-2,025 km<sup>2</sup>, which is close to the 2,000 km<sup>2</sup> threshold of a plant species' range considered critical by IUCN (2001) based on area of occupancy). The rating used in this assessment is based on these figures, although distribution across quarter degree squares equates more to extent of occurrence. The NBRI's specimen database, published sources (Loots 2005), web-based databases (Craven & Kolberg 2017, Raimondo et al. 2009) and own observations were used to determine the known distribution of plant species.





**Figure 3:** Plains in the Greater Fish River Canyon Landscape: (a) fringing the northern Huns Mountains, dwarf shrubs are dominant on the Moedhou plains; (b) *Euphorbia gregaria* is the dominant plant on the Gawachab plains in the north-east of the study area; (c) the Dreigrat plains in the Succulent Karoo Biome receive occasional fog; (d) the Gamkab plains in the eastern study area are covered in carpets of succulent creepers (here *Mesembryanthemum garusianum*) and grasses (*Stipagrostis* sp.) after good rains, but are barren most other times.

An additional rating of 1 for protected species (Nature Conservation Ordinance 4 of 1975 and 247 of 1977 and Forest regulations No. 170 of 2015) or Cites (Convention on International Trade in Endangered Species of Fauna and Flora) listed species was only added if the species had not received

**Table 1:** Rating of plant species according to level of endemism, red list status, narrow-range and legal conservation status (qds = quarter degree square).

Category	Description	Rating
Endemism	Confined to Southern Namib or southern Karas Region	3
	Confined to Namib Desert or Gariiep Centre of Endemism	2
	confined to Namibia	1
Red list status	Endangered	3
	Vulnerable	2
	Near-threatened	1
Narrow-range	1-3 qds	3
	4-6 qds	2
	7-10 qds	1
	>10 qds	0
Protected	Protected under Nature Conservation Ordinance 4 of 1975 and 247 of 1977 and Forest regulations No. 170 of 2015	1
Cites		1

a score for endemic, red-listed or narrow-range, to avoid double accounting (protected and Cites species are often endemics). The ratings of plant species that received a ranking of conservation importance were added per biotope resulting in one numerical figure for each landscape unit. The mean of all values was determined and used to delimit the three class intervals (Table 2), which in this study was weighted towards higher values, as the data range was large (2-301) and dominated by one outstanding highly rated mapping unit.

Data quality was rated to provide some indication of completeness of the plant species list for each landscape unit. The quality assessment took into account the coverage of the area for each biotope and the nature of the rainy season. This is subjective, but it is not included in the algorithm for determining biodiversity importance and used only to (1) better interpret the resulting zoning and (2) guide future

**Table 2:** Range of values applied for biodiversity importance in the Greater Fish River Canyon Landscape.

Class interval	Value	Biodiversity importance
Lowest value to mean	<57	Fair
Mean + mean	57-114	Medium
> mean + mean + 1	>14	High

research. “Good”, “medium” and “poor” were used in this evaluation. “Poor” data quality usually means a lack of rain in these areas resulting in not being able to include short-lived species and species which are only active for a short period of time, such as such as herbs and bulbs.

To ascertain whether particular functional groups of plants were more likely to be of conservation importance, growth form distribution was compared between the overall flora and species of conservation importance. The growth form categories ferns, geophytes, grasses, herbs, shrubs and trees followed standard definitions (Ellenberg & Mueller-Dombois 1967), while succulents were classified according to von Willert et al. (1990).

## RESULTS

Based on the proposed algorithm, nearly one third (265) out of a total of 835 plant species included in this study were species of conservation importance (see downloadable [Appendix 1](#)). The highest ranked species are shown in Table 3, most of which are succulents. Overall, the majority of the species are shrubs (33.5%), followed by herbs (22%) leaf-succulent dwarf shrubs (15.7%), geophytes (10.3%), grasses (7.9%) and dwarf stem-succulents (3.5%). Species of conservation importance showed a very different growth form distribution: over one third of the species of conservation importance were leaf-succulent dwarf shrubs (37%), followed by shrubs (21%) and dwarf stem succulents (11%) (Figure 6). This included many Namibian endemic leaf-succulent dwarf shrubs such as *Astridia hallii*, *Drosanthemum nordenstamii*, *Eberlanzia clausa*, *E. sedoides*, *Ruschia ruschiana* and *R. sabulicola*. Other functional groups (geophytes, grasses, herbs, trees and tall stem-succulents) each contributed less than 10% to the species of conservation importance, but this includes the charismatic stem-succulents *Aloidendron pillansii* (formerly *Aloe*, Grace et al.

**Table 3:** The highest ranked plant species of conservation importance in the Greater Fish River Canyon Landscape. \* denotes succulent plants.

Plant species	Endemism	Red- list	Narrow -range	Summed rating
<i>Neoluederitzia sericeocarpa</i>	3	3	2	8
<i>Juttadinteria albata</i> *	2	2	3	7
<i>Schwantesia loeschiana</i> *	3	2	2	7
<i>Aloidendron pillansii</i> *	2	3	1	6
<i>Antimima dolomitica</i> *	3	1	2	6
<i>Crassula numaisensis</i> *	3		3	6
<i>Elephantorrhiza rangei</i>	3		3	6
<i>Lebeckia dinteri</i>	3		3	6
<i>Monsonia trilobata</i>	3		3	6
<i>Chlorophytum boomense</i>	3		3	6
<i>Ruschia sabulicola</i> *	3		3	6

2013) and *Pachypodium namaquanum*. Shrubs are represented amongst others by the range-restricted Namibian endemics *Caesalpinia merxmullerana*, *Eriocephalus klinghardtensis*, *Euclea asperima* and *Haematoxylum dinteri*.

The mapping delineated 32 landscape units based on landform, climatic regime and, in some instances, underlying rock types (Figure 4, Table 4). Thirteen landscape units represent plains (Figure 3), three represent rivers and 16 represent mountains and inselbergs (Figure 2). The western and southern part of the study area is mountainous with plains wedged between these mountains.

**Table 4:** Landscape units, their biodiversity conservation rating and data quality in the Greater Fish River Canyon Landscape.

Landscape unit	Biodiversity rating	Data quality*
Altdorn hills and plains	Fair	Good
Arimas plains	Fair	Poor – dry
Aussenkehr hills	Fair	Good
Canyon slopes	Medium	Good
Dreigrat plains	Medium	Good
Eastern sand plains	Fair	Good
Eastern Orange mountains	Medium	Poor – dry
Fish river and tributaries	Fair	Moderate
Gamkab inselbergs	Fair	Good
Gamkab river	Fair	Good
Gamkap plains	Fair	Good
Gawachab plains	Fair	Good
Gondwana hills	Fair	Moderate
Holoogberg	Fair	Good
Klein Karas mountains	Fair	Moderate
Konkiep plains	Fair	Moderate
Lowen river and Naute dam	Fair	Moderate
Moedhou plains	Fair	Poor – dry
North-east canyon lands	Fair	Moderate
North-west canyon lands	Fair	Moderate
NE Namus mountains	High	Moderate
W Namus mountains	High	Moderate
Naute inselbergs	Fair	Moderate
Naute plains	Fair	Moderate
Northern Huns mountains	Fair	Poor – coverage
Nudavib mountains	High	Good
Obib mountains	High	Moderate
Orange River valley	Medium	Good
SE Orange mountains	Medium	Moderate
Southern Huns mountains	Medium	Poor – coverage
Succulent Karoo sand plains	High	Good
Zebrafontein plateau	Medium	Good

\* Dry: only dry season data were available;  
Coverage: the area was only partially sampled.

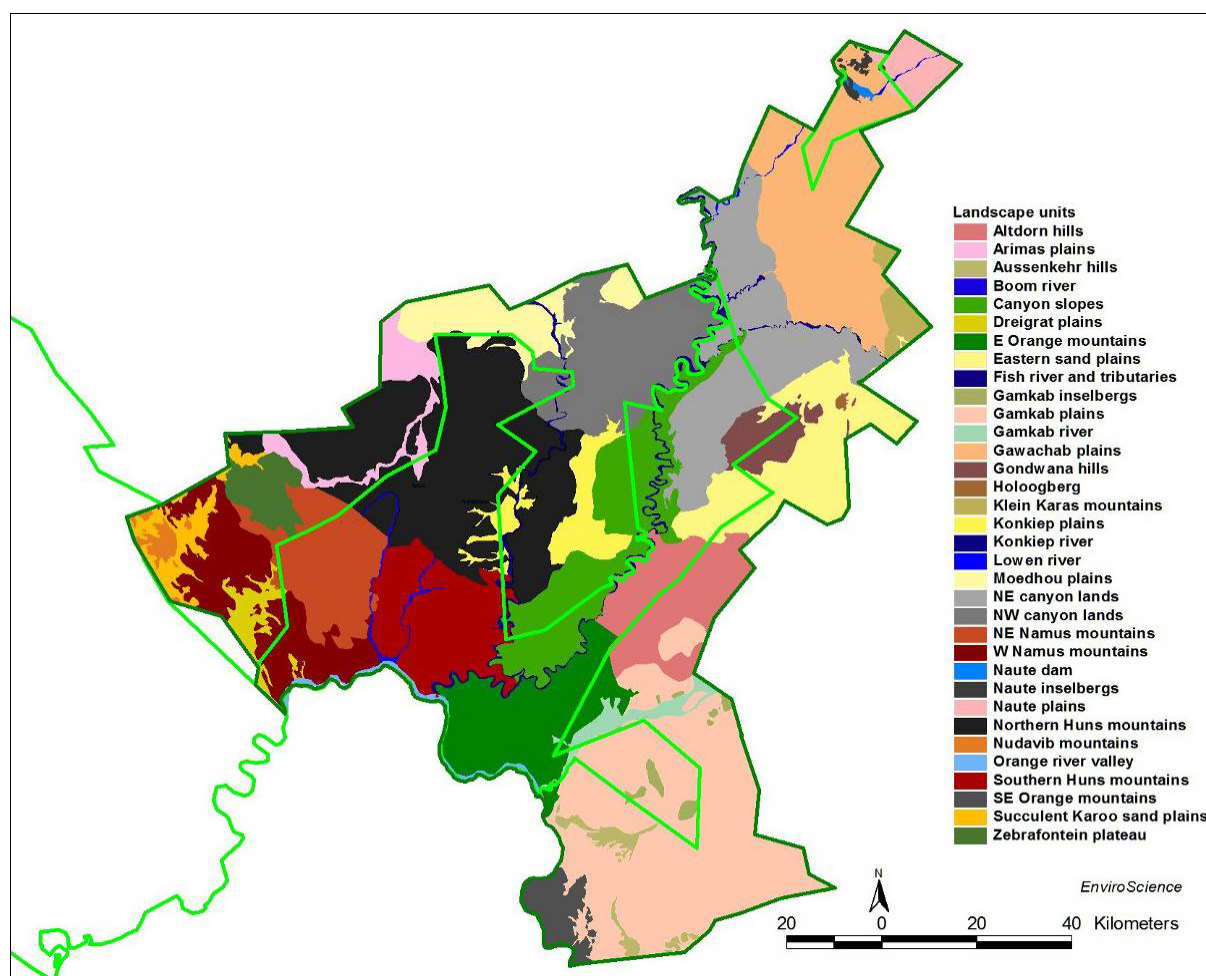
In the context of the GFRCL, most of the western Succulent Karoo section of the study area is classified as being of “high” conservation importance, the remainder of the winter rainfall influence area of “medium” conservation importance (Figure 5). “Medium” conservation importance also extends north-east along the Fish River and east along the Orange River. Data quality is considered good for nearly half of the landscape units and poor only for five landscape units (Table 4).

## DISCUSSION

### Species of conservation importance

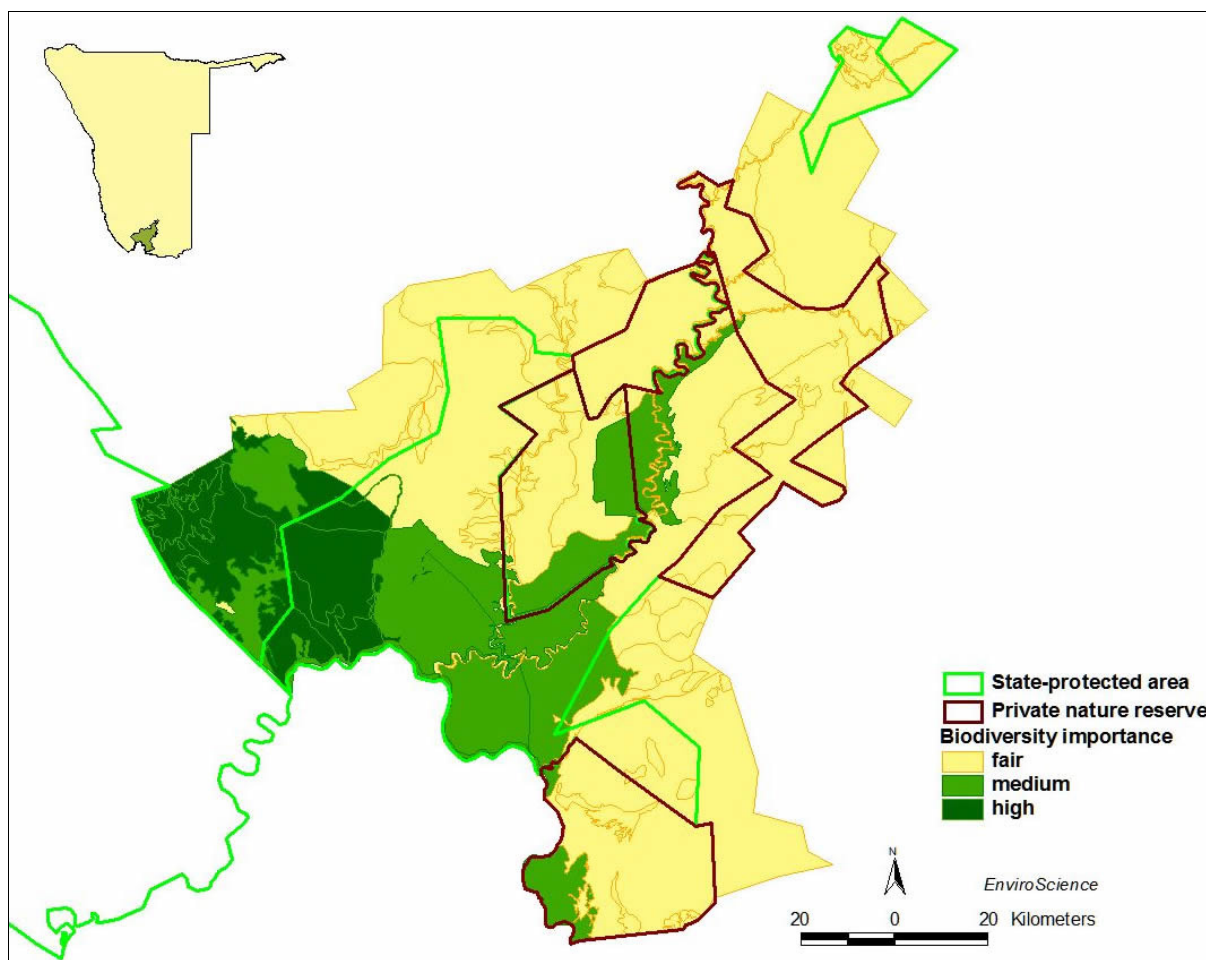
Leaf succulent shrubs of the Aizoaceae (formerly Mesembryanthemaceae) contributed the largest group of species of conservation importance. This is not surprising in a transitional area to the Succulent Karoo Biome and therefore in close proximity to a major centre of diversity of Aizoaceae (Chesselet et al. 2000, Klak et al. 2004). Also, a substantially higher proportion of dwarf stem-succulents were amongst the species of conservation importance than represented in the overall flora, and also, more stem-

succulents and trees than in the overall flora (Figure 6). The dwarf stem-succulents include, in addition to Aizoaceae (e.g. *Conophytum*), many Apocynaceae (tribe Ceropegiae), a group of plants well represented in this area with the genera *Hoodia*, *Larryleachia*, *Quaqua* and *Tromotriche* (Bruyns 2014), numerous *Crassula* species, Crassulaceae, and dwarf succulent *Euphorbia*, due to a variety of niches suitable to the dwarf stem-succulent habit. The contribution of leaf-succulent dwarf shrubs with Succulent Karoo affinities is evident and congruent with growth form distribution in the Extra Cape Flora (Snijman 2013), to which the study area belongs. It is nevertheless surprising that geophytes do not contribute more to species of conservation importance than represented generally in the flora of the study area and the Cape Floristic Region nearby (Proches et al. 2006), particularly as a number of new geophytes have been discovered during the fieldwork for this study (Goldblatt & Manning 2013, Oberlander et al. 2014, Kativu & Bjora 2016, Burke, in press). There could be two reasons for this: either (1) geophytes are under-collected in this area or (2) geophytes are possibly not as range-restricted as the dwarf stem-



**Figure 4:** Landscape units in the Greater Fish River Canyon Landscape and boundaries of state-protected areas (bright-green outline).





**Figure 5:** Biodiversity zoning for conservation importance in the Greater Fish River Canyon Landscape in Namibia (ochre lines indicate the boundaries of broad habitat units).

succulents. This would have to be investigated in future studies.

### Biodiversity zoning

Conservation planning often incorporates expert knowledge where biodiversity inventories are incomplete or time is limited. Although this is justified in many situations (Cowling et al. 2003, Reyers et al. 2007), expert knowledge carries some bias. The approach used in this study requires a reasonably good inventory for the selected target group of taxa, but offers an objective process in determining biodiversity importance. To obtain an inventory, all available distribution data (not only data of the National Herbarium's specimen database), plus targeted field work to fill evident gaps in Namibia were used. Although the data are certainly not complete, they are a good approximation of the status of plant diversity in the study area. A recent study in South Africa (Williams & Crouch 2017) clearly supported this approach and highlighted the pitfalls of only relying on national data. This rating reflects plant diversity, endemism and a proxy for rarity, which are widely used indicators in conservation planning (Burgess et al. 2007, Knapp

2011, Pressey et al. 2003). What is new in this approach is the systematic incorporation of range sizes of species and level of endemism (Table 1). This has proved particularly important since IUCN discouraged the use of the category "rare" in red-listing (IUCN 2001, Gärdenfors 2001), and "rare" is now applied only occasionally in Namibia's red data list (Loots 2005), but not incorporated systematically. On the other hand, rarity has been stressed as an important criterion in national assessments of biodiversity (Cadotte & Davies 2010, Knapp & Salomn 2010), and using only red-listed species as indicators, which is often the focus of biodiversity assessments (Cowling & Lombard 1998), may therefore be insufficient in this area. The low number of red-listed plant species corroborates this point (Table 3). In total, only 4.5% (40 species) of plants of conservation importance included in this study were red-listed in a threatened category, a further 13 species were "rare" in Namibia's red-list, while 265 species (one third of the species included in this study) were used as indicators of conservation importance when the method introduced here was applied. This is, to some extent, a reflection of the number of species that have so far been assessed.



However, the fact that overall only 29% of the assessed species were categorised in one of the threatened (including near-threatened) categories, several of which have a narrow distribution range, highlights the possible shortcomings using only red-list species in a threatened category as indicators. Greater diversity in indicators is expected to better reflect the underlying plant and habitat diversity and concomitant species interactions (Danielson 1991, Zobel 1997, de Bello et al. 2010), and would therefore be more representative of the ecosystems in the study area.

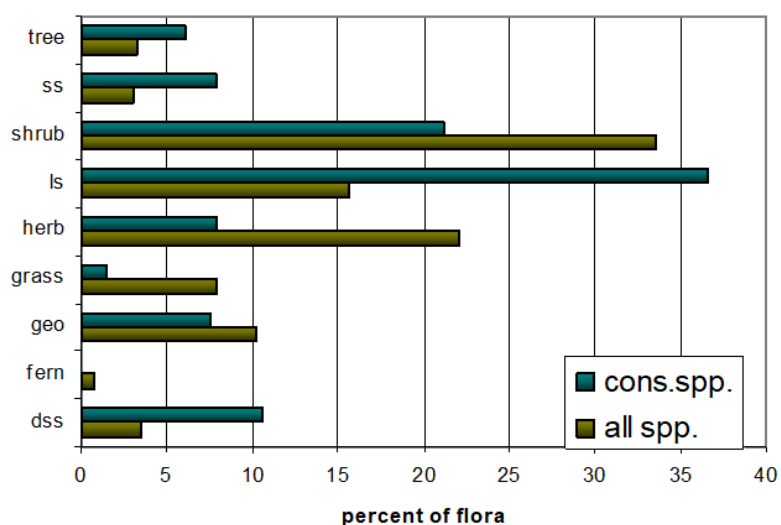
The rating of plant species could be adapted for different purposes (Table 1). For example, this study places legally protected species at the same level as species listed on Cites. If more emphasis should be given to national legislation over international guidelines, ratings could be adopted accordingly. Also, there is, to some extent, an overlap between the categories “endemism” and “narrow-range”, giving a higher weighting overall to the range sizes of species than any of the other categories, which is justified in an area with many species with restricted ranges. For other purposes, the overbearing influence of narrow range sizes may not be appropriate and the categories “endemism” and “narrow-range” could be combined in one rating.

“Rarity”, in a strict sense, is a reflection of the abundance of a species (that is the number of individuals in a defined area). This type of information is hardly ever available for this scale of assessments and range size was therefore used as a proxy for rarity.

Biodiversity importance in the study area is a relative rating in the context of this landscape, providing a

means of prioritisation. It does not mean that areas of “fair” biodiversity importance do not warrant protection or are not important for biodiversity. The biodiversity zoning in this study clearly indicates the ecological importance of river systems and mountain areas in this arid landscape as well as the overriding influence of winter rainfall. Biodiversity importance of the landscape units considered of “poor” data quality, largely due to the lack of rain in these areas (the plains north of the Huns Mountains and the eastern section of the mountains along the Orange River) and difficult access (e.g. the Huns Mountains) could change if additional data are incorporated, particularly for those units presently rated as “fair” (Arimas and Moedhou plains, northern Huns Mountains) (Table 4).

The development of the class intervals to assign biodiversity importance could be considered arbitrary, but the proposed algorithm was developed for this particular area and purpose and also presents a relative, but not absolute measure of “biodiversity importance”. The method is area-specific and class intervals are adapted to the resulting data range and the spread of the data to provide a means of prioritisation. While this provides a structured process, it still maintains some flexibility which is tied to the species richness of the study area and is therefore site-specific. Incorporating species values adapted to a particular situation has also been applied in conservation planning at continental and regional scales (Margules & Usher 1981, Burgess et al. 2006). This approach has proved useful not only in conservation planning, but also in environmental and other biodiversity assessments in Namibia (Burke et al. 2008, Burke 2011) and could be applied elsewhere.



**Figure 6:** Distribution of growth forms amongst the Greater Fish River Canyon Landscape flora (all spp., n=835) and species of conservation importance in this landscape (cons.spp., n=265) (dss=dwarf stem-succulents, ls=leaf-succulents, ss=stem-succulents).

It is recognised that habitat units with plants as indicators do not provide the complete picture of biodiversity patterns, but as the core element of the food chain and indicators of many ecosystem services (Cardinale et al. 2012), plants are considered a reasonable proxy to present biodiversity patterns. Where adequate data are available for selected fauna indicators, a similar approach could be used to rate fauna indicators and incorporate these in biodiversity zoning (Linder et al. 2012). Also, where information is available at the level of detail required for a particular study area, ecosystem services could also be included as indicators to provide a more holistic approach (O’Farrell et al. 2010). Of potential consideration is

the fact that spatial interpolation methods are affected by sampling design, spatial distribution, data quality, correlation between primary and secondary variables and interaction among factors (Li & Heap 2014) and this applies to this study as to many others in conservation planning (Cowling & Lombard 1998).

### Implications for conservation

This biodiversity zoning provides a means of prioritisation and is a relative measure within the study area, which is positioned in the Gariap Centre of Endemism (van Wyk & Smith 2001), and therefore already recognised as an area of high conservation importance. At present, a large part of the most important area for biodiversity in the Greater Fish River Canyon Landscape is formally protected in the Ai-Ais –Richtersveld Transfrontier Park, and the adjoining private nature reserves provide further protection in the form of buffer areas or corridors (Figure 1). Nevertheless, exploration and mining take place along the Orange River, including the section in the Ai-Ais Hotsprings Game Park, and the lack of environmental management associated with these activities threatens some rare plant species (e.g. *Portulacaria pygmaea*). Further, nearly half of the most important area in the west (the western Namus mountains, Succulent Karoo sand plains and Dreigrat plains) are on private farmland, where livestock farming is practiced, and the townlands of the mining town, Rosh Pinah. The expansion of townships around Rosh Pinah comes with concomitant overutilisation of natural resources and these activities may pose a threat to some of the rarer plants. Management guidelines developed on the basis of this zoning attempt to address this aspect (Burke 2013), but these need to go hand-in-hand with addressing the underlying challenge of increasing numbers of people moving from rural to urban areas in search of jobs and better services, and more stringent implementation of environmental assessments and management plans for development activities as prescribed under Namibia's Environmental Management Act.

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# Determining age, growth rate and regrowth for a few tree species causing bush thickening in north-central Namibia

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## ABSTRACT

Age, growth rate and regrowth for some indigenous tree species causing bush thickening in Namibia were determined. The mean age varied between 25.5 years for *Acacia mellifera* subsp. *detinens* and 35.7 years for *Acacia reficiens* subsp. *reficiens*; the mean stem diameter growth rates varied between 2.78 mm/year for *Acacia tortillis* subsp. *heterocantha* and 3.79 mm/year for *Acacia mellifera* subsp. *detinens* and *Terminalia prunioides*. After harvest, the mean rate at which stumps resprouted varied between 45% for *Acacia reficiens* subsp. *reficiens* and 92.8% for *Dichrostachys cinerea* subsp. *africana*. Although our data are limited, they can be viewed as a first step to determining age, growth rate and regrowth for various tree species, although more research is required, including research on variables potentially influencing these parameters for a better understanding of the bush thickening problem in Namibia.

**Keywords:** *Acacia mellifera*; *Acacia reficiens*; *Acacia tortillis*; age; bush encroachment; bush thickening; *Colophospermum mopane*; *Combretum apiculatum*; *Dichrostachys cinerea*; growth rate; Namibia; regrowth; *Terminalia prunioides*

## INTRODUCTION

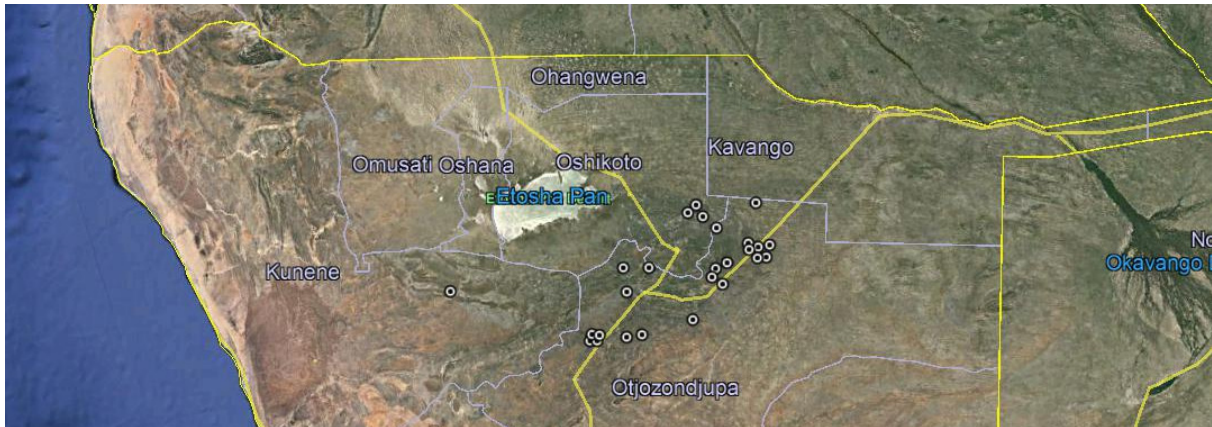
Bush thickening (generally referred to erroneously as encroachment) is a much-discussed topic in Namibia, although with a general paucity of scientific work investigating the issue. Although some papers have been published on bush thickening related issues such as densities (Cunningham 1998, Bester 1999), biomass available for charcoal (Cunningham 1998) and wood production (Zimmermann & Joubert 2002), historic review of bush thickening (Cunningham 2014), fire related issues (Joubert et al. 2012), seed predation (Joubert et al. 2013), vegetation dynamics of a single species (Joubert et al. 2008), identification of bush thickening related causes (Zimmerman et al. 2008), control measures and aftercare issues (e.g. Strobach 1999, Van Eck & Swanepoel 2008, Van Eck & Van Lill 2008) and general overview (De Klerk 2004), little has been published on determining age, growth rate and regrowth of these problem species. According to Smit et al. (2015), this is a priority for research.

Determining the age, growth rate and regrowth for the various species causing bush thickening is important for a better understanding of the problem as well as the successful (sustainable) management of these species. Many indigenous tree species are difficult to age as their rings are obscured by the deposition of heartwood, which is initiated when the

stems are about 50 mm in diameter and susceptible to heart rot e.g. *Colophospermum mopane* (Caughley 1976, Scholes 1990). However, with practice this can be achieved as demonstrated previously (Cunningham 1996).

Age, growth rate and regrowth (%) were determined for *Acacia (sensu lato) fleckii*, *A. mellifera* subsp. *detinens*, *A. reficiens* subsp. *reficiens*, *A. tortillis* subsp. *heterocantha*, *Colophospermum mopane*, *Combretum apiculatum* subsp. *apiculatum*, *Dichrostachys cinerea* subsp. *africana* and *Terminalia prunioides* on 26 farms in north-central Namibia (Figure 1). Soils on these farms (freehold farms focusing mainly on cattle production) were highly variable although dominated by leptosols and cambisols and the mean average annual rainfall varied between 350 and 500 mm. Levels of bush thickening varied roughly between 2,000 to 10,000 trees per hectare and is the reason for initiating charcoal production in an attempt to improve grazing as well as attempt diversification and improve cash-flow. These studies were initiated during mid-2016 in an attempt to ensure sustainable utilisation of this resource, with data collection ongoing.





**Figure 1:** Farms where data were collected in north central Namibia

## METHODS

### Age

Discs of the main stem of trees with a charcoal potential (i.e. bigger specimens between 75 mm and 118 mm as selected by charcoal workers as specimens with a good charcoal potential) were cut at basal height; polished with a belt sander to make the growth rings more visible; photographed and then imported to computer where the picture was enlarged (similar to using a magnifying glass) and the growth rings counted. When ‘heart rot’ was encountered, the average ring increment to the centre of the stem was calculated. An increment borer was initially used but the local hard woods and generally smaller trees made this technique unsuitable while carbon dating is also ineffective at the age range we dealt with. The aim was to determine the growth rate of certain problem species and we thus did not attempt age calculations for a variety of diameter classes and/or the mean diameter classes in each area, but rather focused on ‘charcoal producing’ diameter classes i.e. larger specimens.

Tree growth rings were determined as the light versus darker bands deposited between seasons. However, this is easier said than done as many of the indigenous species have hard woods with faint rings, often with false rings (i.e. rings that are not complete and do not signify a growth year) or are suffering from ‘heart rot’ where the core becomes spoiled. Although accepted as a rough field technique (no known-age trees were available for comparative purposes, while carbon dating is not feasible), with perseverance and practice one can actually count the growth rings and gather valuable data not previously attempted and viewed as the most practical way of determining age (Cunningham 1996) (Figures 2 & 3).

### Growth Rate

Mean growth rate was determined by dividing diameter (mm) by the age (years) for each sample/species and the mean annual growth rate was determined as diameter increase per year (mm/year).

### Regrowth

Percentage regrowth was calculated as the percentage of trees showing coppice after having been felled for



**Figure 2:** *Acacia reficiens subsp. reficiens* with growth rings clearly visible as light vs dark bands (~39 years).



**Figure 3:** *Terminalia prunioides* with distinctive ‘yellow wood’ and some external damage, which affects growth rings (~46 years).

charcoal purposes, for at least 50 tree stumps harvested per farm.

### Soil Texture

The soil texture and approximate clay content (%) were determined by feel using the soil texture decision model – i.e. where a soil sample is wetted, rolled into ball/ribbon/sausage, bent and checked for consistency, etc. This is a simple yet practical way of determining the clay/sand/silt content and is used as a field technique only.

## RESULTS AND DISCUSSION

### Age

The mean age of our samples ranged from 25.5±1.5 years for *Acacia mellifera* (n=36) to 35.7±1.8 years for *Acacia reficiens* (n=23) (Table 1). Strong positive linear correlations were observed between age and diameter for the species analysed: *Acacia mellifera* (r=0.80); *A. fleckii* (r=0.91); *A. reficiens* (r=0.54); *Combretum apiculatum* (r=0.99); *Colophospermum mopane* (r=0.94); *Dichrostachys cinerea* (r=0.72) and *Terminalia prunioides* (r=0.64). *Acacia tortilis* is excluded because of too few samples. Although tree height was not taken into consideration, Cunningham (1996) shows that age is better correlated with circumference (i.e. diameter) than with height for *Colophospermum mopane*.

### Growth rate

The mean growth rates varied between 2.78±0.3 mm/year for *Acacia tortilis* (n=2) and 3.79±0.2 mm/year for *Acacia mellifera* (n=36) and 3.79±0.3 mm/year for *Terminalia prunioides* (n=31) (Table 1).

Growth rates for the eight species we analysed were very similar, ranging between 2.78 and 3.79 mm/year. With a mean annual growth rate of 3.79 mm/year for *Acacia mellifera* and *Terminalia prunioides*, this would imply that it would take ~21.1-26.4 years to reach the diameter of 80-100 mm

viewed as the ‘best’ size for charcoal production. Cunningham (1996) indicated that it would take 23-25 years for various *Colophospermum mopane* woodlands to reach mean circumference classes of between 14.24 cm and 16.64 cm after harvesting in the Limpopo Province in South Africa. Using these growth rates to determine harvesting protocols should take cognisance of variables such as climate (growth rings show annual variation) and edaphic (various soil types) factors which are known to influence growth rate (Cunningham 1996).

### Regrowth

Regrowth of harvested trees has solicited much debate and depends on factors such as species, season harvested, size of tree harvested, method of harvest, rainfall, soil type and browsing impact. Some species coppice prodigiously, such as *Dichrostachys cinerea*, while other species do not, although once again there are a legion of reasons for this that have not been tested for here. It does, however, indicate that not all harvested individuals coppice, which would mean that ‘after care’ (something generally advocated by farmers) – especially by chemicals – may not be required for most species except *Dichrostachys cinerea*. This would have to be determined at farm level due to differences in factors such as species, soils and rainfall. However, regeneration also takes place from seed, although seed sometimes requires exceptional circumstances, such as at least two consecutive seasons of favourable rainfall for *Acacia mellifera*, to become established (Joubert et al. 2013).

Our data show a mean regrowth (%) varying from 45% for *Acacia reficiens* to 92.8% for *Dichrostachys cinerea* on black soils with approximate clay content of between 45-50% and 49.5% for *Acacia reficiens* to 66% for *Acacia mellifera* on red sandy/loamy soils with approximate clay content of between 20-35% (Figure 4). According to Cunningham (1996), growth rates for *Colophospermum mopane* are faster on deep/fertile soils and slowest on shallow/rocky soils, while high clay content can be responsible for stunted growth (Dye & Walker 1980). As we used a practical

**Table 1:** Mean diameter, age and growth rate for eight tree species causing bush thickening in north-central Namibia. (Values given ± SE).

Species	Mean diameter (mm)	Mean age (years)	Mean growth rate (mm/year)	n
<i>Acacia fleckii</i>	96.5±13.5	30±2	3.17±0.3	4
<i>Acacia mellifera</i>	91.38±4.4	25.5±1.5	3.79±0.2	36
<i>Acacia reficiens</i>	117.3±4.2	35.7±1.8	3.40±0.1	23
<i>Acacia tortilis</i>	90.5±10.5	32.5±0.5	2.78±0.3	2
<i>Colophospermum mopane</i>	108.2±14.7	29.8±3.1	3.59±0.2	5
<i>Combretum apiculatum</i>	97.8±11.1	32±1.6	3.02±0.2	5
<i>Dichrostachys cinerea</i>	76±6.7	27±2.5	2.86±0.2	6
<i>Terminalia prunioides</i>	105.2±6.2	30±1.9	3.79±0.3	31



coppices so vigorously that an entirely cleared area regenerates fully to dense forest within 15 years. However, regeneration from seed to a pre-cleared state could take up to 40 years. Recovery period is shortened by higher rainfall and lengthened by drought, according to growth simulations done on *C. mopane* (Scholes 1990). Determining growth rates could be used to plan for practical, sustainable, rotational-harvesting regimes for the charcoal and wood industries or retrospectively to understand events that caused bush thickening in certain areas in the past.

A first attempt to determine the growth rate of coppice (with known harvesting dates) varies between 2.6 mm/year for *Acacia reficiens* and 9 mm/year for *Colophospermum mopane* (Figure 6). There are numerous factors potentially affecting regrowth (e.g. species, initial fast growth, browsing impact, rainfall, frost and soil type) and although the current data are limited, further research could be refined over time to evaluate the effects of these variables.

More research on growth rate and regrowth for the various tree species, especially including variables potentially influencing these, is imperative to better understand the bush thickening problem in Namibia. Although our results are limited to eight species and few data, this note serves as an introduction to attempt raising the awareness of the value of bush thickening tree species as a potential sustainable resource rather than a curse to be eradicated at all costs. Perhaps it is time to change perceptions regarding sustainable 'agroforestry' especially in the trying environmental times in which we live.

## ACKNOWLEDGEMENTS

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# The status of *Faidherbia albida* trees in the Hoanib River, Namibia

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## ABSTRACT

The Hoanib is an ephemeral river flowing from the highlands of Namibia through the hyper-arid Namib Desert to the Atlantic Ocean. Seasonal floods recharge groundwater that supports riparian woodlands, which are vitally important to wildlife and livestock. Previous studies prior to 2001 found that mega-herbivores are having an impact on the main tree species in the river system, *Faidherbia albida*. In 2002 two permanent boreholes were drilled for wildlife in order to reduce competition with livestock. This paper presents the results of a survey carried out in October 2012 to assess the current population structure and damage to *F. albida* in the lower Hoanib River, and to compare it with previous surveys. There was great variability in population structure, growth form, regeneration and elephant damage between the transects surveyed in different sections of the river. Khomarib Schlucht, where there are few elephants, showed a healthy demography, with many juveniles, regeneration, and little damage. Dubis wetland, which is utilised by game and livestock, had no mature trees and one clump of juveniles. Transects between the two boreholes showed low recruitment, little regrowth, and a marked 5m-high browse-line. They were dominated by mature, single-trunked trees with old elephant damage, and healthy canopies. From the “President’s Borehole” downstream there were juvenile trees, fewer mature trees, and thickets. Juvenile trees showed pruning from above and the sides. Near the floodplain, there were only highly pruned juveniles and a few trees in the 20-40 cm DBH size-class. Another visit, in 2014, showed changes to the juveniles at Dubis wetland, but no other changes. Overall lack of *Faidherbia* recruitment along the mega-herbivore frequented section of the river is of concern for the long-term survival of this important linear oasis. Suggestions are made on key interventions that could be implemented to prevent the loss of these woodlands, which would be a conservation and ecological disaster

**Keywords:** ephemeral river, *Faidherbia albida*, Hoanib River, mega-herbivore, Namib Desert, population structure, recruitment, regeneration, riparian woodland

## INTRODUCTION

Located along the western seaboard of southern Africa, Namibia is an arid to hyper-arid country. Rainfall is both erratic and highly variable, increasing from almost nothing at the coast to an annual average of ca. 650 mm in the north east (Mendelsohn et al. 2002). The ancient Namib Desert extends from the coastal plain inland for approximately 150 km to the escarpment, which separates it from higher altitude areas in the interior of the country. Twelve westward-flowing ephemeral rivers drain the central highlands, channelling seasonal floodwaters through this arid landscape (Jacobson et al. 1995). These rivers form ‘linear oases’ in the desert and act as biological corridors between the interior and the coast. Most support riparian woodlands dominated by various *Acacia* (*sensu lato*) and other woody species (Curtis & Mannheimer 1995). These woodlands are of vital importance to wildlife as well as to the local human communities and their livestock that depend on them (Jacobson et al. 1995, Leggett et al. 2003a, Moser

2006, Moser-Nørgaard & Denich 2011). Many of the larger wildlife species, including the African elephant (*Loxodonta africana*), Angolan giraffe (*Giraffa giraffa angolensis*) (Fennessy et al. 2016), gemsbok (*Oryx gazella gazella*), springbok (*Antidorcas marsupialis*) and baboon (*Papio cynocephalus*) are able to extend far into the true desert, using these riparian resources.

The Hoanib River flows from the mountainous interior west of Etosha National Park westwards through communal conservancies for about 270 km, draining into the Atlantic Ocean just south of Möwe Bay in the Skeleton Coast National Park (Fennessy & Fennessy 2004) (Figure 1). The riparian woodland is dominated by *Faidherbia albida* (Del.) A.Chev. (formerly *Acacia albida*). This species can reach heights of 20-30 m, with a trunk diameter of over 2 m (references in Barnes & Fagg 2003). It is generally an erect, single- or multi-stemmed tree, with a spreading canopy. Since it has a strong capacity for regeneration, often sending up new shoots from exposed roots or producing a number of new stems from a fallen trunk, it is often hard to

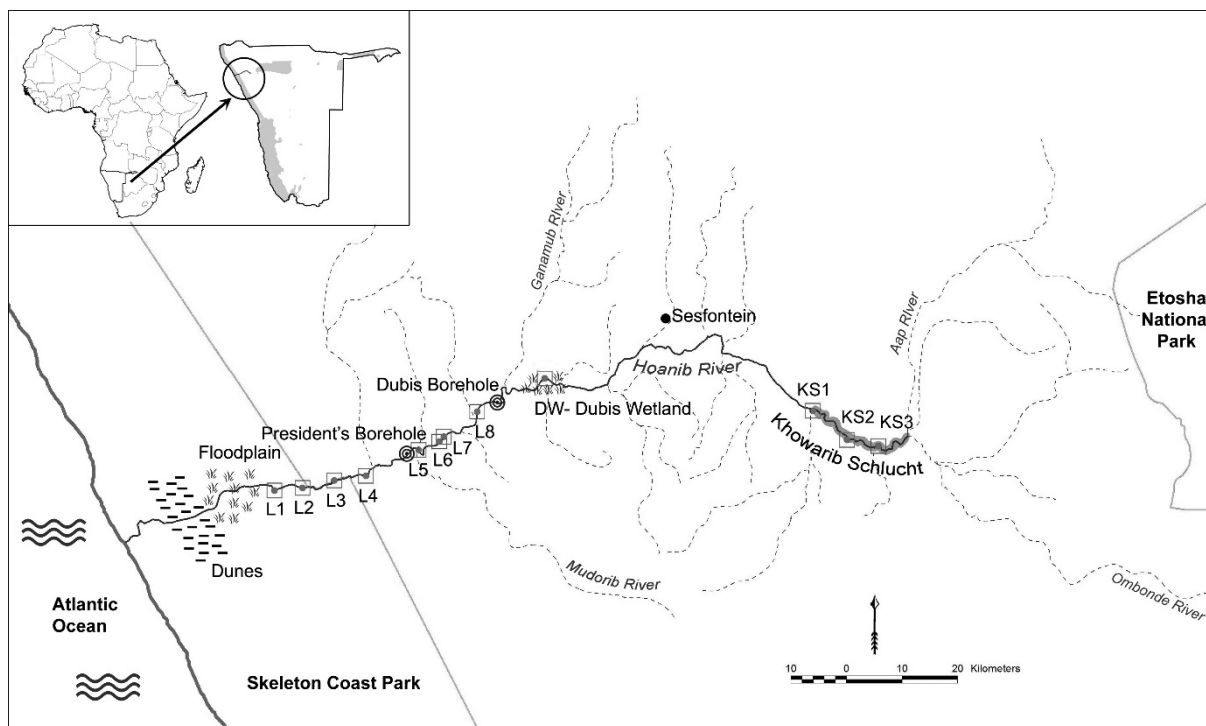


determine whether one is looking at a group of trees growing close together or a single, multi-stemmed tree. Its wide distribution in Africa attests to its ecological adaptability (CTFT 1989, Barnes & Fagg 2003). In areas with low herbivore pressure, branching generally starts near the ground and higher branches often droop to the ground. However, in areas such as the lower Hoanib River, excessive pruning by mega-herbivores results in trees with trunks bare of branches until the base of the canopy at around 5 m (Figure 2). Seedlings and juveniles generally have a single stem, with numerous lateral branches. In Namibia it is commonly known as Anaboom or Ana tree, but elsewhere in Africa it is known as Winter thorn, since it is dormant in summer and produces new leaves and flowers in winter. This tree provides valuable browse and shelter to numerous animals, ranging from invertebrates (Theron 2010), reptiles, birds and small mammals to mega-herbivores such as giraffe (Fennessy 2004) and elephant (Viljoen 1988, 1989, Jacobson & Jacobson 1998, Leggett et al. 2001, 2003b). As a legume, its leaves and pods are high in protein (Barnes & Fagg 2003). Owing to its unusual 'reverse' phenology, producing new growth in the dry season (Curtis & Mannheimer 1995), the tree is vital to wildlife, people and livestock at a time when other resources are limited (Moser-Nørgaard & Denich 2011). Although present along most of the river's length, *F. albida* reach their greatest abundance and size in the lower

reaches (Nott 1987, Jacobson 1995, Fennessy et al. 2001, Fennessy 2004).

Various factors affect the growth and recruitment of *F. albida* (Barnes & Fagg 2003). In riparian woodlands, flooding (Jacobson 1995) and herbivory (Barnes 2001a, Fennessy 2004) have the greatest effects. In ephemeral systems, erratic seasonal floods bring water and nutrient-rich sediments from inland (Jacobson et al. 1995). Flooding occurs during most years in the Hoanib River, but the duration and intensity of floods are highly variable (Department of Water Affairs records, Appendix 1). Since rainfall is very low in the lower Hoanib River (~25 mm/yr), germination and recruitment are dependent on flood water. However, floods are both a blessing and a curse to the riparian vegetation. Apart from recharging the ground water upon which the trees depend, and providing sufficient moisture for germination, heavy discharges uproot and undermine mature trees, and flatten or drown younger trees (Jacobson et al. 1995, Moser 2006, pers. obs.). Lack of floods and declining groundwater can also have a detrimental effect on riparian vegetation (Huntley 1985, Douglas 2014).

Trees have evolved, and continue to evolve, in association with herbivores, and are dependent on them for dispersal and germination of seeds (see Barnes & Fagg 2003, Sebata 2013). Minor browsing probably shapes them into single-stemmed trees from the multi-stemmed juveniles, however, high



**Figure 1.** Map of the lower and mid-sections of the Hoanib River, Namibia, showing survey sites and other features mentioned in the text.



**Figure 2:** *Faidherbia albida* in the Hoanib (and Ugab) River, 2012.

Left to right from top: Elephants under a mature tree near Dubis borehole (note flood debris at the base of the tree); elephant in the Ugab River reaching above the kudu browse-line; old elephant bark damage; row of mature trees along the edge of the main channel, showing distinct browse-line and lack of undergrowth; unbrowsed root coppice in Khowarib Schlucht; thicket at site L2; pruned floodplain tree; stunted leaves on pruned branch; browsed root coppice in section L; giraffe pruning trees at site L1; giraffe eating at browse-line in section L.

herbivore pressure can be detrimental (Bergström 1992, Sebata 2013). Repeated browsing from below often results in a distinct browse-line on trees, while browsing from above by giraffe leads to rounded bushes and thickets (pers. obs.) (Figure 2). In searching for nutrients, elephant often break branches and strip bark off the trees, sometimes entirely ring-barking them or pushing trees over to reach food (see Barnes & Fagg 2003). Trampling of seedlings additionally reduces recruitment (Barnes 2001b, 2001c, Moser 2006). Elephant, giraffe and other herbivores use the river year-round, moving up and down, as well as in and out of the main river bed in response to available water and flushes of new growth and fruit (Viljoen 1988, Jacobson 1995, Leggett et al. 2001, Fennessy 2004). In the 1980s the wet season core activity area of elephant and other wildlife was the Hoanib River floodplain, while this shifted east during the dry season to the Dubis wetlands approximately 50 km upstream (Viljoen 1988).

The increase in mega-herbivore numbers in the 1980s in the lower Hoanib River raised concerns about the impact of these animals on the *F. albida* populations, especially around the Dubis wetland. The latter deteriorated from a 'healthy climax population' with recruitment of juvenile trees at 30% (Viljoen & Bothma 1990) to 0.2% in 1995 (Jacobson 1995). The situation improved to 9% over the next five years — the result of improved rainfall and flooding — combined with a range extension of the elephant population into areas away from the Hoanib River (Fennessy 2004, Leggett 2006).

Elephant damage in the Hoanib River has previously been reported to be highest near the permanent water of the Dubis wetland (Fennessy 2004, Jacobson 1995). In November 2002, two permanent boreholes were established in the lower Hoanib River to provide water for wildlife, which subsequently changed the seasonal movements, use and density of elephant (Leggett 2006). It is feared this may have an adverse impact on the trees near the boreholes. Thus, it is of utmost importance to monitor the trees in the entire river as the maintenance of a healthy *F. albida* population in the western rivers is essential to the survival of desert-dwelling wildlife (Jacobson 1995) that are an integral part of the riparian ecosystem and a major tourist attraction. This paper presents the results of a recent survey to assess the current population structure and damage to *F. albida* in the lower Hoanib River, and to compare it with previous surveys and the situation in the upper Hoanib River.

## METHODS

### Study Area

The ephemeral Hoanib River of northwestern Namibia, with a catchment of 17,200 km<sup>2</sup> (Fennessy & Fennessy 2004), rises in the hills around western Etosha National Park, flows westwards, and cuts through the escarpment in a narrow, meandering canyon known as the Khowarib Schlucht (ca 137 km from the coast) (Figure 1). The canyon length is about 17 km in a direct line over the mountains, or 27 km following the river. The river bed ranges from 70-120 m in width, with silt or rocky banks in most places. The riparian woodland, for the most part, comprises a row of trees and shrubs along one or both sides of the main channel, dominated by *F. albida*, with *Acacia (sensu lato) tortilis*, *A. erioloba*, *Colophospermum mopane*, *Combretum imberbe* and *Salvadora persica*. Towards the western end of the canyon are a series of springs, with variable discharges depending on seasonal floods. At the time of the survey (October 2012), a stream of about 1.5 m width and depth of up to 0.3 m was flowing. Grass, sedges and various small shrubs lined the stream. Elephant and giraffe historically occurred in the upper reaches of the Khowarib Schlucht, although they seldom frequent the area now. The Schlucht falls within the Anabeb Conservancy, an area of 1,570 km<sup>2</sup> with a population of around 1,400 people, three small villages and two tourist camp sites (NACSO 2017). Cattle and goats browse in the river.

Over the next 40 km the river flows westwards through the wide, open Sesfontein valley, with a human population of around 1,500 centred in the village of Sesfontein (NACSO 2017). At about 78 km from the coast, the river narrows to a valley through the mountains. At the start of this mountainous area is another spring, which creates the permanent Dubis wetland of about 8 km in length. This wetland supports reed beds for most of its length, and virtually no large trees. Livestock graze in this area. Dubis borehole (installed by government in 2002 to supply water for wildlife in order to reduce human-wildlife conflict at water-points nearer Sesfontein) is situated about 10 km downstream of this wetland, near the Hoanib-Ganamub confluence. From here to the floodplain (section L, described in the Data Collection section) are the main riparian woodlands, which support healthy numbers of herbivores, including giraffe and elephant. Livestock generally do not come down this far. The main river channel varies in width from 50-200 m, sometimes diverging around islands. In places there are floodplains of varying width. The large trees are mainly found as a single row along the edges of the channel or on the islands, but in some reaches of the river trees occur across the

channel and onto the floodplains. Apart from the species mentioned previously, *Combretum watti* and *Cordia sinensis* are also common shrubs. The 'President's Borehole' is situated at the Hoanib-Mudurib confluence, about 53 km from the coast. At about 40 km from the coast the river enters the Skeleton Coast National Park. Between the Sesfontein valley and the park, the river separates the Sesfontein Conservancy to the north (2,465 km<sup>2</sup>) from the Palmwag Tourism Concession to the south (NACSO 2017). This section of the river is used by tourists hoping to see desert elephants and giraffe. There is an upmarket tourist lodge on the south bank of the river near the park.

Roughly 60 km downstream of Dubis wetland the river widens into an extensive floodplain area located within the Skeleton Coast Park. Floodwaters blocked by the dunes further west spread out across this plain, which extends from about 17-27 km inland and is about 2-4 km wide. Vegetation here varies with flood events. In wet years it is covered in small shrubs, forbs and grasses that are an important food source for wildlife. In 2012 it was densely covered with woody shrubs of 1-2 m high, with very occasional larger trees that were stunted and heavily pruned by browsers. The lowest section of the river from the coast until about 17 km inland is hyper-arid and is blocked by dunes, with no trees. Only in years of exceptionally heavy rainfall inland does floodwater break through to reach the sea (Jacobson 1995).

## Data Collection

*Faidherbia albida* was surveyed in the Hoanib River from the eastern end of the floodplain to the eastern end of the Khowarib Schlucht (between 13° and 14° E; see Figure 1) in late October 2012. This was the hot-dry season, approximately six months after the last flood in early 2012, and at the end of the leafing and flowering season for *F. albida*. For this study, the river was divided into four reaches, based on the marked differences in *F. albida* structure and recruitment. Moving downstream, these were: 1) KS = Khowarib Schlucht, middle of the Hoanib River; 2) DW = Dubis wetland, approximately 80 km from the coast; 3) L = Lower Hoanib from Dubis borehole to the floodplain; and 4) the Hoanib floodplain. The section of river from the western end of KS to DW (Sesfontein valley) was not surveyed. All previous studies concentrated on section L, with the exception of Fennessy et al. (2001) who looked at the entire river.

In total, 12 transects were surveyed over a combined distance of 78 km: three in KS (27 km), one at DW and eight in section L (51 km). Transects in KS and L were selected randomly, at roughly 5 km intervals (see Table 1). At DW, there were only two mature

trees and one patch of juveniles some distance away from the mature trees. Only the juveniles were surveyed. The results from the DW transect therefore do not reflect the entire wetland area, but serve to illustrate what recruitment in this section of the river can be like. Owing to the very low density of trees, no transects were surveyed in the floodplain, but measurements were made on ten individual trees.

The length of each transect was about 0.5 km and the width corresponded to the width of the river along that section (except for DW, where the transect width was the width of the area covered by the trees) (see Table 1). In each transect, the following measurements were made for every *F. albida* individual: (i) height: Trees were classified into height classes of  $\leq 0.5$  m; 0.5-2 m; 2-5 m; 5-10 m; 10-20 m;  $>20$  m. Individuals less than 0.5 m high were regarded as seedlings which were assumed to have germinated after the 2012 floods; (ii) circumference: For trees  $>1.5$  m high, circumference at breast height (1.3 m) was measured and converted to diameter at breast height (DBH) in 20 cm intervals in order to compare with previous studies. (All saplings 0.5-1.5 m tall had diameters far less than 20 cm); (iii) Vitality, or general health of the plant: this was estimated on a scale of 1-5, where 0 = totally dead; 1 = mostly dead; 2 = several live branches; 3 = half tree alive; 4 = a few dead branches; 5 = no dead sections; and additionally, (iv) Growth form, browse-lines, coppicing and regrowth: these were noted along with recent elephant damage – bark stripping (BS), main stem breaking (MSB) and branch breaking (BB).

All individuals  $>0.5$  m high, but with stem diameter of  $<20$  cm, were regarded as juveniles. For purposes of comparison with previous studies, juveniles and mature trees were grouped together and proportions of each diameter class were calculated. For comparisons between sites in 2012, juveniles and mature trees with DBH  $>20$  cm have been presented separately in Table 1 because of the differing importance of juveniles and adults in the population, and the differing growth forms and impacts of herbivores on these two groups.

In December 2014, a brief follow-up survey was undertaken, principally to look at the survival of the seedlings and juvenile trees.

No statistical comparisons could be made between this survey and previous studies as different areas were assessed. Nott (1987) counted all trees greater than 2 m high from Dubis wetland to the floodplain, but did not measure them; Viljoen and Bothma (1990) selected specific trees for size classification in 1982; Jacobson (1995) measured all trees within the first 12 km from Dubis westward, and thereafter



all trees within the first kilometer of six equally spaced sections along the remaining 53 km to the floodplain; Fennessy (2004) measured all trees within the stretch from Dubis to the floodplain. The last three researchers all measured diameter at breast height.

## RESULTS

### Population structure and recruitment in 2012

In total, 812 *F. albida* trees over 0.5 m tall and 232 seedlings under 0.5 m tall were recorded over the length of the surveyed sections of the Hoanib River in 2012. Densities varied considerably, from 2-5 trees/ha in transects L4 – L8 to 41 trees/ha in L1. The overall mean of the 12 transects was  $14 \pm 13$  trees/ha (Table 1). In all transects, the density of mature trees >20 cm in diameter was low. In contrast, the density of juveniles (>0.5 m high but with DBH <20 cm) was indicative of the overall density patterns, with the highest density being in L1, followed by DW. Density of seedlings was low in all transects, except L1 (Table 1).

There was a marked difference in the population structure of *F. albida* between the different sections of the river, and within each section (Figure 3). Transects KS1-3 had a healthy age structure, with

many juveniles and decreasing numbers of older trees, and yet still having old trees with diameters of  $\geq 120$  cm. Only juveniles were observed at DW. In the upper reaches of L (L5-8), there were few juvenile and young trees, with a high proportion of old trees, suggesting low recruitment. The number of juveniles increased again nearer the floodplain, with fewer old trees. Transect L1 had no very big trees, but many juveniles, as well as seedlings.

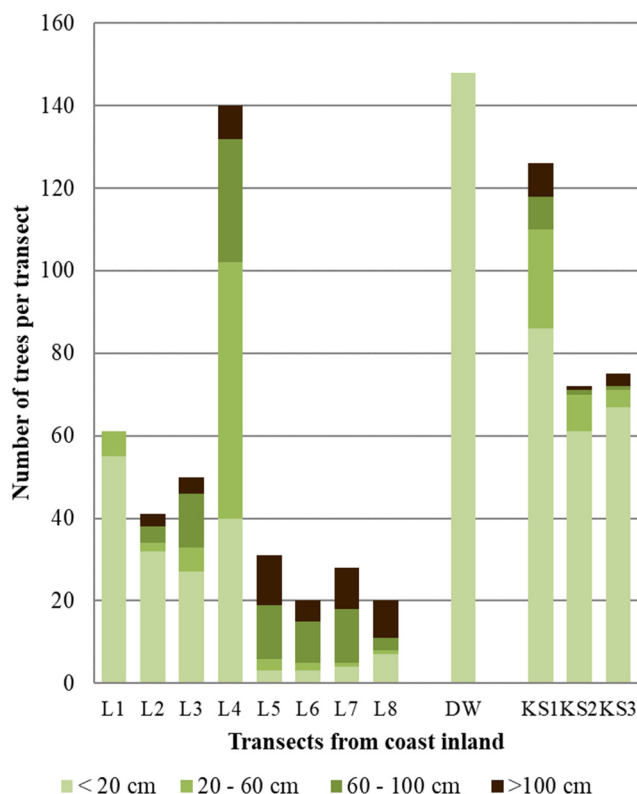
### Growth form, regeneration and vitality in 2012

Growth form and regeneration in the form of coppice growth varied along the length of the river. Most of the trees observed were healthy, with an average vitality score of 4-5 (Table 1). Trees in L4 had the lowest vitality (with a mean score of 3), with many broken and almost dead trees. This transect had a very wide floodplain, and trees furthest from the main river channel had lower vitality scores than those nearer the main channel. This was most likely to be due to water stress, since a lower water table has been shown to adversely affect this species (Ward & Breen 1983). This transect was 9 km west of the President's Borehole where larger numbers of mega-fauna occur.

All tree canopies in the Khowarib Schlucht started near the ground, and young growth had not been substantially modified by browsers, despite some evidence of browsing having taken place. The only browsers encountered were cattle, although old giraffe droppings were observed. The percentage of trees coppicing from the roots in KS was the highest of all (mean of 49%), suggesting that this form of regeneration has not been restricted by browsing and trampling, as it has been lower downstream. There did not appear to be any marked differences in growth form or population structure between the three transects in this section, despite the fact that the two downstream transects (KS1-2) were in areas with a permanently flowing stream.

The juvenile trees at Dubis wetland (DW) exhibited the same growth form as those in KS, branching low down, with long branches, unlike the cropped-off juveniles of further downstream in the L transects.

All the mature trees in section L showed a totally different growth form from those in the KS. Every tree with a diameter >10 cm had one or two unbranching trunks up to a 5m-high browse-line, above which the canopy was spreading and generally healthy, with the exception of individuals within L4. The juvenile trees in this reach of the river were all highly pruned from above,



**Figure 3:** Number of *Faidherbia albida* trees measured per transect in the Hoanib River 2012, showing the relative proportions of trees per DBH category.



**Table 1:** Location and details of 12 transects in the lower and mid Hoanib River, with density and growth characteristics of *Faidherbia albida* in October 2012.

Transect	L1	L2	L3	L4	L5	L6	L7	L8	DW	KS1	KS2	KS3
River reach	Lower Hoanib floodplain to Dubis borehole								Dubis wetland	Khowarib Schlucht		
Co-ordinates S (degrees, mins)	19 23.29	19 23.04	19 22.32	19 21.87	19 19.36	19 18.55	19 18.07	19 15.67	19 12.43	19 15.90	19 18.38	19 18.97
Co-ordinates E (degrees, mins)	13 00.02	13 02.76	13 05.83	13 08.91	13 14.01	13 16.02	13 16.48	13 19.69	13 26.26	13 52.98	13 55.61	13 58.64
Direct distance from coast (km), approx	27	34	37	44	53	56	59	64	77	121	126	132
Section of river downstream of Dubis (km)	45+	40-45	35- 40	25-30	15-20	15	12	5-10	not applicable			
Length (m)	376	544	476	650	625	560	554	510	450	800	450	473
Average width (m)	40	60	90	400	145	195	147	170	100	110	70	72
Area (ha)	1.5	3.3	4.3	26	9.1	10.9	8.1	8.7	4.5	8.8	3.2	3.4
Seedling density/ha	30.7	5.5	0.7	2.1	0.2	2.1	0.6	0.9	4.9	2.5	2.2	6.5
Juveniles (DBH <20cm)												
density/ha	36.7	9.7	6.3	1.5	0.3	0.3	0.5	0.8	30.9	9.8	19.1	19.7
Mature (DBH >20cm)												
density/ha	4.0	2.7	5.3	3.8	3.1	1.6	3.0	1.5	none present	4.5	3.4	2.4
average vitality	5	4	4	3	4	4	4	5		4.0	4.0	4.5
% trees BS	0.0	55.6	21.7	3.0	39.3	5.9	37.5	46.2		old damage only		
mean % BS	0.0	16.0	40.0	10.0	20.0	15.0	10.0	40.0		0.0		
% with BB	0.0	22.2	26.1	30.0	14.3	23.5	29.2	7.7		old damage only		
% with MSB	0.0	0.0	8.7	6.0	3.6	11.8	0.0	7.7		old damage only		
% root coppicing	25.0	33.0	13.0	15.0	7.1	17.6	12.5	46.2		27.5	45.5	75.0
Total # trees >0.5 m	61	41	50	140	31	20	28	20	148	126	72	75
Density per hectare	40.7	12.4	11.6	5.4	3.4	1.8	3.5	2.3	32.9	14.3	22.5	22.1
# thickets	2	12	3	19	0	0	0	0	0	0	0	0
Comments					President's Borehole					running water	running water	

resulting in short, stumpy, very spiny branches, with short leaves clustered tightly among the thorns (Figure 2). Almost all of them had dense, spiny branches from ground level, presumably as a defense against bark stripping. Virtually no broken branches or stems were observed among the juveniles.

A feature of the woodland between L3 and L8, apart from the growth form of the mature trees, was the lack of undergrowth, not only of juvenile and coppicing *F. alibida*, but plants in general. Root coppicing was recorded in  $21 \pm 14\%$  of mature trees, but all new growth had been heavily browsed. Thus there were no new trees developing from this coppice. There was no correlation between coppicing and distance from DW, as had been found by Fennessy (2004), with coppicing highest in L8, nearest to the wetland.

In the four transects nearest to the floodplain (L1-4), there were thickets of *F. alibida* (Figure 2). This phenomenon was most marked at L1, where the main channel was confined between silt banks. These thickets ranged from 1-20 m in length, with a height range of 0.5-5 m, but most were less than 2 m high.

On the floodplain, and just to the west of it, were scattered individual trees that had been stunted by pruning, possibly aided by the wind. These trees ranged from 1.5-6 m in height, with spreading canopies of 1.5-3 m in diameter touching the ground.

The general health of most of the trees was good. Mature trees whose canopies were above the reach of herbivores mostly showed healthy leaf production, many still with flowers (main flowering time is around August and September (Curtis & Mannheimer 1995)), and fresh new pods. Those nearer the floodplain had less healthy leaves and little pod production, compared with those nearer DW.

### Elephant and other damage in 2012

No ring barking was observed on any of the trees examined. Jacobson (1995) noted alteration in cambium growth on many of the damaged trees, which resulted in "numerous deep convolutions.... parallel to the longitudinal axis of the trunk". This phenomenon was noted in the present study on all the mature trees >40 cm DBH downstream of Dubis, as well as on some of the younger ones (Figure 2). Fennessy (2004) classified the age of bark damage into three categories: 'recent' being less than one month old with green gashes; 'current year' damage no longer green gashes but with no cambium regrowth; and 'old' where the cambium was starting

to grow over the scar. No recent (green) damage was noted in the present study, and very little current damage, but there was extensive evidence of old bark damage.

There was also little sign of browse and trampling on the juveniles in the DW. On some of the bigger juveniles a few strips of bark had been removed and a few lateral branches pruned, but overall these individuals resembled those found where there were no mega-herbivores.

Most elephant damage was recorded in section L, the lower reaches (Table 1). About 30% of mature trees showed signs of damage within the last year, with an average of 20% of the bark of these individuals having been stripped. Only 5% of trees had one of their main branches broken by elephant, and only 22% showed broken branches. In most cases new shoots were growing from the damaged branches, but wherever these were within reach of mega-herbivores, the new growth had been severely pruned. Mortality was low, with only ten dead stumps or trees found, compared with 391 living trees in this section. Seven fallen trees that were still living were recorded in this stretch. One of these appeared fairly healthy, the others being almost dead, with a few highly browsed coppice shoots. Three stumps had coppice shoots of up to 1 m, heavily browsed into a thicket. It was not possible to tell whether the fallen trees had been eroded by floods or pushed over by elephant, but we suspect the former, since there was no evidence of trees having been uprooted by elephant. Stumps may have been the remnants of trees broken or ring-barked by elephant in the past. Outside of the transects, there was evidence of trees having fallen due to flood erosion.

### Changes from 2012 to 2014

There were no major changes in population structure between the two survey periods, but the overall health of the trees had deteriorated by almost one category, probably due to the poor rains and lack of flooding in 2012/3 and 2013/4. The greatest changes were noted among the seedlings and juvenile trees, particularly at site DW. Whereas in 2012 there were 22 seedlings, 133 juveniles and no mature trees over 20 cm DBH here, in 2014 there were no seedlings, 121 juveniles and 15 trees in the 20-40 cm category, with two dead juveniles. Thus at least five seedlings must have been recruited into the juvenile category (23%), while the others did not survive. Recruitment of juveniles into the adult category was 11%. Far more browser damage was noted in 2014 than in 2012 (Figure 4), while average vitality decreased from 4 to 3. Coppicing decreased from 21% of trees to 15%. Five trees had been cut with axes, but all

were coppicing. This was the only evidence of human impact encountered along the entire river.

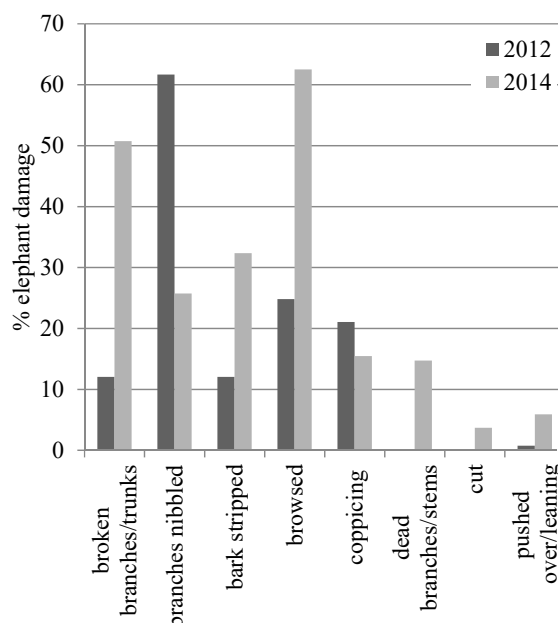
The only place seedlings were found in 2014 was at transects L1-2, nearest the floodplain (12 in total). In L1, six of the 2012 seedlings (13%) had grown into the juvenile category, while three juveniles had grown into adults (5%). Many of the previous seedlings appeared to have become thickets. No recruitment of seedlings into the juvenile category had taken place in any of the other transects, and few juveniles had become adults. There was no new bark stripping since 2012, but at L3 many of the lower branches were broken off. The vitality of most trees was lower, with far less coppice growth, which was in poor health. At a number of sites, the clambering shrub, *Salvadora persica*, was smothering many of the trees.

## DISCUSSION

### Population structure and recruitment

When comparing the results for section L (from Dubis waterhole to the floodplain) with the previous studies, one can see changes in the demography of these trees, with the greatest variability being in the juvenile age class (Figure 5), from almost 0% (Jacobson 1995) to 44% in 2012. Viljoen and Bothma (1990) recorded a healthy population structure in 1982, with relative proportions of each age class decreasing from 30% juveniles, 25% young trees in the 20-40 cm age class to relatively very few old trees with trunk diameter of >120 cm. The years preceding 1982 had very poor flooding. In the summer of 1981/82 there was an above-average flood, which probably gave the surviving seedlings enough moisture to grow into juveniles (Department of Water Affairs records, Appendix 1).

By 1995 the population structure had changed drastically, with almost no recruitment (0.2%) in the juvenile class, a low percentage in the 20-40 cm class, and most of the trees being 40-60 cm in diameter (Jacobson 1995). Because of the low proportion of juveniles, the proportions of the older age classes were higher than that recorded by Viljoen and Bothma (1990), but still showing the same decline in numbers from 40-60 cm to 100-120 cm. It is interesting that there was a higher proportion of >120cm diameter trees compared with 100-120 cm trees. This would suggest growth of some of the latter trees into the next size class. Flood data show generally low flooding from 1984/85 to 1993/94, which would have stressed and weakened the surviving young trees. The big flood of 1994/95 destroyed most of the trees in the juvenile and 20-40 cm class (Jacobson 1995). A similar effect was noticed in the Kuiseb River after an exceptionally big flood (Curtis in prep.). The 40-60 cm trees were



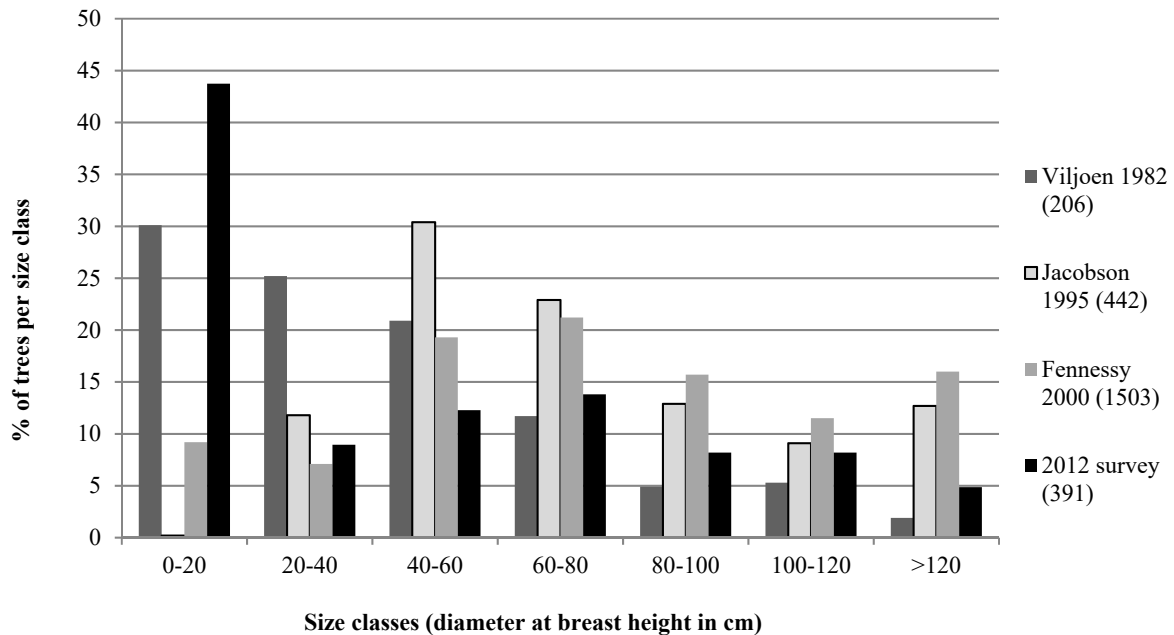
**Figure 4:** Increases in damage to juvenile trees near the Dubis wetland in the Hoanib River from 2012 to 2014.

strong enough to withstand this, and formed the highest proportion that year.

Five years later, Fennessy (2004) recorded a slight increase in juveniles, but this was still much lower than previously recorded. From 1996/97 to 1998/99 there were three years of average flooding, which would have given the juveniles a chance to establish themselves, and thus not as many were destroyed in the extremely big flood of 1999/2000. The low proportion of 20-40 cm trees was a result of the lack of recruitment in 1995, with the 60-80 cm class being the dominant one, following Jacobson's (1995) dominant 40-60 cm class. The larger size classes followed a similar pattern to that shown by Jacobson (1995).

The eight years between Fennessy's (2004) study and this 2012 survey generally had average to above-average flooding, resulting in apparently high recruitment, with the peak of mature trees being in the 60-80 cm class. The lower proportion in all size classes was due to the high proportion of juveniles (44%). The pattern, however, was not the same as shown by the two previous data sets, with a low proportion of really old trees (5%), which was still higher than that shown in 1982.

Transect DW near the Dubis wetland was very different from the other transects, with no mature trees, and the entire subpopulation consisting of juveniles and a few seedlings. This was probably a result of the high water-table and high salinity. Thickets in the lower reaches could have resulted from clumps of seeds germinating together and



**Figure 5:** Changes in population structure of all trees measured from 1982 till 2012 in the lower Hoanib River between Dubis and the floodplain (L1 - L8 of current survey). Numbers in brackets after the author indicate sample size.

growing up as one, or could have sprouted from fallen trees.

The low seedling density between Dubis and the floodplain (with the exception of L1 at the start of the floodplain) was probably due to trampling by elephant and giraffe. In L1, where there are no large trees to attract elephant, seedlings had a better chance to survive. It could also be attributed to the flood waters slowing down here, and depositing more seeds and debris.

The changes in recruitment and age structure of the lower Hoanib *F. albida* trees over the years is of concern, particularly the almost total lack of juveniles in 1995, and low numbers in most of the sites in 2012. One must, however, be careful of comparisons when different sampling methods and sampling sites are compared. Fennessy's (2004) work is the most comprehensive since that study was a total count which measured every single tree, whereas the other studies are sample surveys. The current work underlines this importance, as well as the masking effect of combining data for a long stretch of river. Our survey indicated healthy recruitment in 2012, but only for small sections and not for the river as a whole. Had we included every tree between Dubis and the floodplain, our results would have been different, with a lower proportion of juveniles and higher proportions of the older trees. There is no doubt that recruitment has improved overall, since Jacobson found no young trees in year 1995 (Jacobson 1998), but the low recruitment in the areas where herbivore density is highest is of concern.

The marked increase in juveniles found in our survey would be encouraging, were it not for the fact that most of the recruitment was in the lower portion of the river. The transects from Dubis waterhole to the floodplain showed low recruitment, but not as low as that recorded by Jacobson (1995). There was an average of 18% in the <20 cm age class for these four transects, indicating some recruitment in the last 17 years. There were almost no trees in the 20-40 cm age class. Most of the trees in this stretch of river were mature and had survived from 1995 and before. Moving downstream towards the floodplain, the proportion of juveniles increased, while the proportion of very old trees with a DBH of >120 cm decreased. In the last transect before the floodplain, there were essentially only juveniles, with a few trees in the 20-40 cm age class. The increase in recruitment from Dubis downstream towards the start of the floodplain mimics the increase in total density of tree.

Both stem diameter and height are functions of the age of a tree, to a certain extent. Fennessy (2004) found that the mean height of trees decreased downstream of Dubis, suggesting a decrease in age downstream. This trend was also observed in this study, and is largely attributable to the large number of juvenile trees in the lower sections of this reach. Possible reasons for the greater numbers of trees nearer the floodplain are seasonal floods depositing nutrients and seeds, which would result in the increased recruitment, as observed in 2012 (Fennessy 2004).

The absence of mature trees in the vicinity of Dubis wetland, as noted in all three studies, could possibly be due to the increased salinity recorded by Jacobson et al (2000) as suggested by Fennessy (2004), although this species is generally regarded as tolerant of high salinity (Orwa et al 2009 cited in Feedipedia 2017, WorldAgroForestryCenter 2017). In 2012, there was a section of the river between L8 and Dubis borehole that was currently dry, but had recently had standing water, as evidenced by the sedges, new seedlings and *Tamarix usneiodes* and other water- and salt-tolerant plants growing there. This area was not surveyed, but the vitality of the mature trees was noted. In a stretch of river about 5 km long, we counted 42 mature trees, of which 8 were dead (19%), 29 (69%) were more than half dead and only 5 were healthy.

### Mega-herbivores and other impacts

Elephants have been reported by various authors to affect recruitment. The decrease in recruitment noted in 1995 was attributed to impacts by elephant, as has been observed elsewhere in Africa and for other species of *Acacia* (Ben-Shahar 1993, 1996, Laws 1970, Barnes 2001b, 2001c, Makhabu et al. 2006). In addition to elephant, other herbivores also favour young growth and have impacts on juvenile trees (CTFT 1989, Fennessy et al. 2001). Fennessy (2004) attributes the increase in recruitment between 1995 and 2000 to favourable rainfall and flood events in the intervening years – there were well-above average floods in 1994/95 and 1999/2000 (Department of Water Affairs records, Appendix 1). During this period browsing pressure was reduced due to more reliable seasonal water sources elsewhere, and the re-expansion of elephant ranges away from the Hoanib River (Leggett et al. 2001). There has been a continuation in this trend, with average to above-average floods recorded for most years between 2001/02 and 2009/10. Since 2010/11 floods, have been below average, which explains the low numbers of seedlings and low recruitment in 2012. The current healthy population structure in the Khowarib Schlucht, where there are few elephant, illustrates the effect that elephant are having on this tree species. The suggestion made by Jacobson (1995) to encourage elephant to move from the lower Hoanib River east into the Khowarib Schlucht is not feasible due to increasing development and potential human-wildlife conflict. The river in the Khowarib Schlucht is narrow, and is used extensively by the local inhabitants to feed and water their livestock.

The Khowarib Schlucht has historically been foraged by elephant, as evidenced by some old scars on the trunks of the larger trees. However, the past damage is not as great as in the lower Hoanib River, with no current bark damage observed. Some broken

branches high in the trees were noted, but were more likely due to baboons rather than elephant, especially as the bark had also been stripped from these higher branches. Most of the juvenile trees had a typical young *F. albida* growth form, with one or two main stems, and a few lateral branches from near the ground. No marked herbivore impact on the juvenile trees was observed, whilst a number of trees appeared to have been knocked over by floods, with the majority re-sprouting. The only mortality observed in the Khowarib Schlucht was one tree that had been struck by lightning.

Elephant damage in the lower Hoanib River has historically been high, but has decreased in recent years. In the early 1980s Viljoen and Bothma (1990) recorded bark damage scars on 72% of trees observed, of which 32% (45 of 142) had greater than 20% of their bark removed. By the late 1980s Nott (1987) recorded 74% (124 of 168) trees with more than 20% damage, whilst in 2000 Fennessy (2004) recorded that only 58% of trees were damaged, and of these only 23% (333 of 1466) had damage greater than 25%. Our findings show nearly 100% of trees had old bark damage, presumably inflicted mostly in the late 1980s and 1990s. Current damage was low – at 11.5% (45 of 391 trees), and only 4.3% (17 trees) had more than 20% damage. Thus elephant impact in terms of bark damage has definitely declined over the years and may be linked to the declining elephant numbers in the river (Ramey & Brown 2016).

The current data do not show a strong link between bark usage and the presence of permanent water, unlike previous studies (Nott 1987, Viljoen & Bothma 1990, Fennessy 2004), which observed that the percentage of trees with bark damage was high nearest the Dubis wetland, decreasing westwards, with distance from permanent water. Viljoen and Bothma (1990) showed 100% utilisation at Dubis, with a marked decrease downstream to almost no utilisation at the floodplain. This was in contrast to Nott's (1987) findings round about the same time. He noted 100% of trees utilised as far downstream as 20 km, thereafter decreasing, and increasing again at the floodplain. Fennessy (2004) noted a similar trend to that found by Nott (1987), but with lower percentages of trees affected, and the increase occurring around 25-30 km downstream. Our results showed signs of 90-100% previous utilisation the entire way along that reach, but there was no link between current elephant damage and distance from Dubis. Branch breaking occurred in 84% of 1,678 trees observed by Fennessy (2004), compared with a mean of only 22±8% in 2012.

Giraffe, too, have an impact on *F. albida* in the Hoanib River. Jacobson (1995) regarded elephant as having the greater impact, while Fennessy (2004)

suggested that the vegetation was equally impacted by giraffe. Elephant cause major physical damage by bark stripping, branch and stem breaking, and uprooting trees, whilst giraffe create a distinct browse-line, as well as pruning young trees and coppice growth. The browse-line on all mature trees in the river was around 5 m, which is typical giraffe feeding height. Giraffe also cause the 'topiary or hedging effect' noted on all the smaller trees downstream of Dubis wetland, and absent in the Khovarib Schlucht. This pruning impact has been noted by Fennessy (2004) and other authors cited therein.

Elephant cannot generally reach as high as giraffe. Rather than browsing high parts of intact trees, they will reach up and break off a branch, which does not result in a distinct browse-line (Fennessy 2004). This was clearly noted in Namibia's ephemeral Ugab River in December 2012, where there are no giraffe, and elephant were seen reaching above the cattle/kudu browse-line to pull off small branches (pers. obs.) (Figure 2). Therefore, it is important to look at the combined impact of all mega-fauna on *F. albida*.

The effect of floods on mature trees appears to be greater than that of elephant and giraffe. After the 1995 flood, Jacobson (1995) observed that 2.8% (18 of 638) of trees with diameters ranging from 30-190 cm were undermined by lateral channel erosion. Many of these trees were washed away, but some were merely toppled, and held in place by intact roots. These trees were subsequently impacted by elephant and giraffe, and destroyed within three months, with branches of up to 8 cm diameter being consumed. This is in contrast to the ephemeral Kuiseb River in the central Namib, where there are no elephants or giraffe, and where, when trees topple but remain rooted, they will regenerate from the fallen trunk, producing a line of new trees (pers. obs.); a phenomenon not observed in the lower Hoanib River. Fennessy (2004) noted that 6% of mature trees were uprooted after the 1999/2000 flood and there was a marked increase in mortality compared to pre-flood data. There was little evidence of flood damage in the current study.

Other factors potentially affecting the trees in the Hoanib River are human impacts as well as the smothering of trees, especially younger ones, by clambering plants, notably *Salvadora persica*, which cover a number of other species as well and prevent them from growing properly. The only evidence of tree cutting was noted in 2014, where some of the juveniles at Dubis wetland had been cut.

## Mortality

Tree mortality was lower in this study than previously recorded. Jacobson (1995) noted that 31% of the 638 trees observed were dead, apparently killed by elephant, compared with only 6% dead among 238 observed by Viljoen and Bothma (1990). Most of the dead trees observed in 1995 were in the 20-40 cm category. There was no difference in mortality percentage nearer Dubis wetland and the sections westwards towards the floodplain. Neither the present study nor the previous ones found evidence of trees being uprooted by elephant, contrary to reports in other habitats (Laws 1970). The previous authors attribute this to *F. albida*'s strong tap root (CTFT 1989) since they are regularly subjected to the pressure of floodwater and debris battering their trunks, which they generally survive. Elephant pressure would be minor in comparison.

As previously stated, elephant appear to be responsible for damage to bark and breaking of branches and stems, while giraffe, and possibly other browsers, are responsible for pruning and dense cropping (Fennessy 2004). Mortality through trampling of seedlings and young trees is probably caused by the suite of herbivores in the river, not only elephant and giraffe but also oryx (*Oryx gazella*) and springbok (*Antidorcas marsupialis*). In the Khovarib Schlucht, livestock replace wild herbivores, but do not appear to have much of an impact on mortality.

The effect of mega-herbivores on regeneration poses a serious threat to the Hoanib river *F. albida* population as a whole. In the ephemeral central Kuiseb River, where there has been no elephant activity in the past century, resprouting fallen trees send down new roots and grow into new trees, creating stands of closely packed trunks (Gardiner et al. 2006, pers. obs.). In addition, coppice from damaged roots can grow up into new trees. New recruits formed have the added advantage of the deep root system of the parent tree, and are more likely to survive than are independent juveniles. In the Hoanib River, coppice shoots are severely browsed, forming short, spiny tufts, which will never grow into new trees (Figure 2). No regeneration of fallen trees was noted in the Hoanib River, which could be an issue for long-term regeneration.

## Effect of boreholes

In November 2002 two boreholes were drilled in the lower Hoanib River to attract wildlife away from nearby human settlements. One was at the end of the Dubis wetland, and the other approximately 18 km downstream at the 'President's Borehole' (Leggett 2006). At the time there was concern that the introduction of these permanent water-points may



have a negative impact on the trees of the lower Hoanib River. Since animals, in particular elephant, historically used the Dubis wetland, the introduction of the borehole may not have significantly affected pressure on the trees in this area. However, Leggett (2006) showed that the distribution and density of elephant did change between 1998 and 2004, with a movement away from the Dubis wetland to the area of the President's Borehole. Like Jacobson (1995), this study found very few trees within the first 10 km of Dubis, and almost no recruitment. The President's Borehole appears to have had a slight negative impact on the already low recruitment of the area, with very few juveniles and no seedlings. It does not appear to have affected the mature trees adversely, except that root coppicing was lowest here. Fennessy (2004) predicted a change in population structure in this area, and tree mortalities downstream. With few exceptions, bark damage was highest near Dubis wetland and the President's Borehole.

From 1982-1995 the elephant population remained at around 25 animals (references cited in Leggett 2006) but has declined in recent years to only 18 individuals (Ramey & Brown 2016). As a result of forage and water availability, both elephant and giraffe numbers are, on average, lower near the floodplain than nearer Dubis (Fennessy 2004), thus allowing seedling establishment and juvenile development, which in turn results in higher densities, but decreased mean heights and age structure.

## CONCLUSIONS

Only the Khowarib Schlucht subpopulation had a truly healthy *F. albida* population and physical structure. The small patch of juveniles in the Dubis wetland as well as transects near the floodplain show that germination has taken place within the last five years. However, these young trees cannot adequately provide the recruitment needed for the whole lower Hoanib River. Importantly, the 2014 survey showed that as these trees increase in size they are becoming more impacted by elephants. The fact that recruitment in the areas with highest mega-herbivore activity is so low remains a concern for long-term viability, and ongoing monitoring is key.

These trees, along with the other species comprising the riparian woodlands of the lower Hoanib River, form an essential source of food and shelter for all the inhabitants of the river, and the loss of these woodlands would be a conservation and ecological disaster. The old trees are currently providing abundant shade and food, but they will not last forever, and greater recruitment is required. It is evident that germination does occur, but the young seedlings and saplings are not being allowed to

reach maturity. Since removal of mega-herbivores from the system is not an option, one recourse could be to create exclosures where seedlings can be protected until they are large enough to withstand trampling and browsing. This has been done very effectively in Amboseli National Park in Kenya, where two wires, one live one earth, at elephant eye height have been sufficient to keep out elephants in most plots, allowing the woodland to regenerate (Western & Maitumo 2004, Western in lit.). This would not keep out the smaller herbivores and would allow floodwater to flow past. Another option would be to put up bee-hives to deter elephants (Cook et al. 2018).

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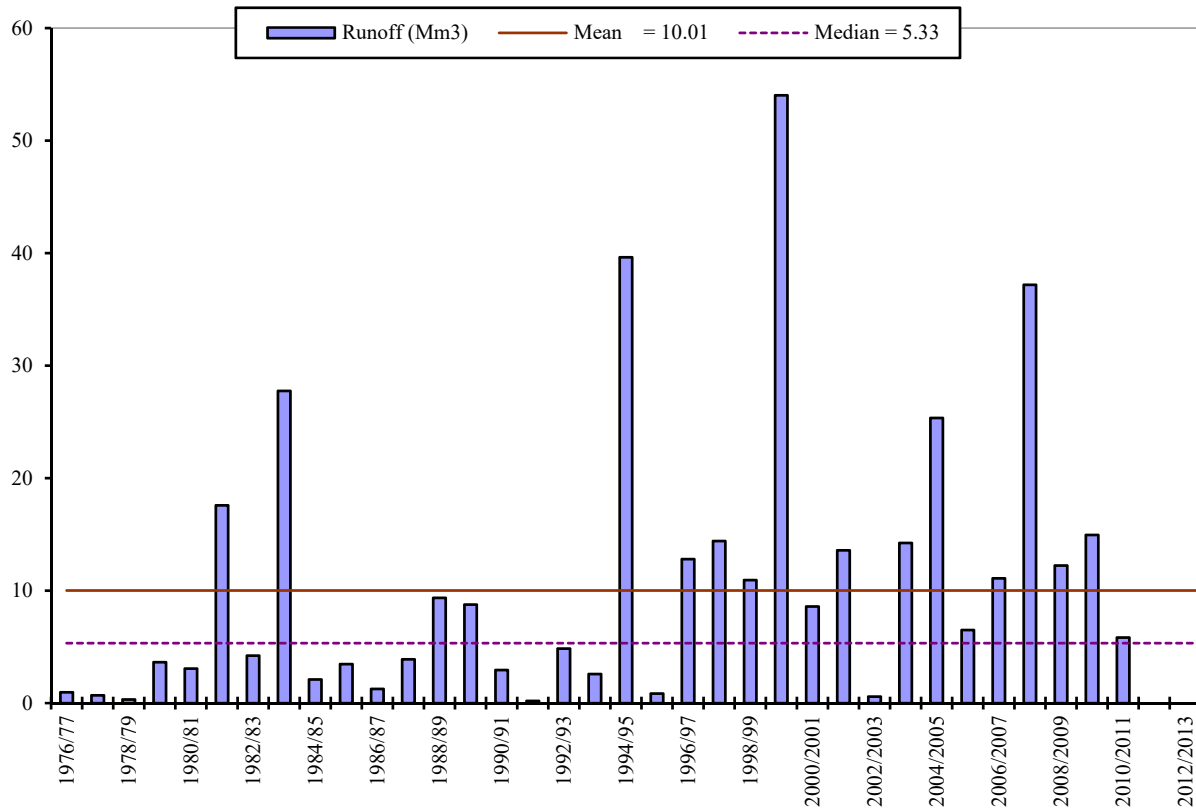
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**Appendix 1:** Flood records for the Hoanib River at Sesfontein. Courtesy of Department of Water Affairs, Windhoek, Namibia.

Hoanib River at Sesfontein.  
Catchment Area = 11,000 km<sup>2</sup>



## Could critical Australian insights illuminate rangeland management in Namibia?

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### Abstract

Climate change is now almost universally accepted as a reality and so too is the “hand of man”. We are causing it. However, do we really understand (accept) what is causing most change or are we focusing huge amounts of money on politically correct (not “wrong”) symptoms? Are greenhouse gases the real problem? Or is how we manage the land the problem and the solution? We put forward some propositions that beg a rethinking of the climate change issue, with a focus on better local land management for better local climate outcomes. We acknowledge that our evidence is based largely on a different way of thinking about climate change and local ecosystem health, but some case studies support this perspective and therefore require close scrutiny with an open mind. The core to our perspective is thermodynamics and the role of plants in that. Plants made our planet suitable for humans. We contend that the ecological malaise is driving climate change at a greater rate than industrial emissions and that the solution lies in land recovery. That is, if we want to address the causes and not simply the symptoms and convenient part truths. Enduring, self-sustaining, ecosystem rejuvenation is the key. We discuss how this can be pursued at a farm scale. We focus on key issues and how they can be addressed by systems thinking, rather than seeing the symptom as the core problem. Bush encroachment is such a symptom. We cannot change how brightly the sun shines, but we can influence how that energy is used, especially if we link it to water management and plant growth.

**Keywords:** bush encroachment, climate change, farming practices, Namibia, rangeland management

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### Introduction

In this paper we explore the potential for ideas developed by Peter Andrews in Australia (Andrews 2006, 2008) to help address key rangeland management issues in Namibia. We spent two weeks as a group exploring these ideas and their applicability in Namibia. While many of the ideas are challenging to our prevailing perspectives and paradigms, we urge an open-minded approach to test them. Surely it is more important to improve our rangeland management than defend it? The ideas we present below are essentially Andrews', but Pringle and Zimmermann see great potential to test these principles to restore natural fertilisation and rehydration processes in sub-Saharan Africa's most arid country. Do these processes also predominate beyond where Andrews has worked? Can they add value to what we are already doing based on decades of field experience? Probably!

#### *The power of plants*

We are all concerned about contemporary issues including global warming, increasing population demands on our planet, declining water supplies and increasing wealth inequality. These are apparently “wicked” problems that will require complex, complicated solutions – or so we are led to believe. What if the primary cause were also recognised as the solution? What if we poured our billions of dollars into correcting nature's own solution to perturbation instead of into disintegrated and disappointing investments whose most impressive outcomes only scratch the surface?

If one accepts that plants and water are both the problem and the solution, then we can bring focus to broad-scale, enduring change. Industrial emissions are minuscule in comparison to what we have done physically to our planet by clearing, overgrazing and “expert” burning. So, should we focus on the sideshow or grasp the problem by the jugular? We need to grow more plants and restore rain-use efficiency. We can do this on both sides of the ledger; by returning the filtering system which prevented our landscape fertility from being lost to the sea; by making it rain more and by cycling more of that rain before it escapes to the currently hostile (reflective) land that typifies most of our planet. As we have more water recycled by plants on the land, we trigger more rain.

It might sound crazily simple; and it is. Plants convert sunlight into carbohydrates, which requires energy; they transpire water in the process and so are landscape air conditioners and they cycle the nutrients for their diverse peers to maintain fertility. Why are we trying to fix everything else? The complexity is human; we have to revive our knowledge of how water and plants can provide very localised and then also continental landscapes that are naturally air conditioned and productive. Surely this is where we should be focusing our efforts to solve the supposedly intractable issues of our time?

#### *Australia's landscape laboratory*

Australia is the “old” continent in that it has experienced least tectonic instability and largely been free of glaciations. Wet (warm) and dry (cold) cycles led to Australian soils being mostly leached and infertile in the Tertiary era. Yet this aridity and infertility was overcome by what can only be understood as some self-organising force that evolved into ecosystems that redistributed water and fertility so that the overall ecosystem was more productive and resilient (Noy-Meir 1981).

The continent of Australia has experienced swings between extremes of harsh and lush conditions over the ages. As plants evolved to occupy land from the surrounding oceans, they gradually improved conditions over the millennia by creating soil, protecting the soil from erosion, moderating extremes of temperature, controlling the cycling of water and providing food for animals that contributed to nutrient cycling. The cooling effect of vegetation during daytime, as transpiration by plants, converted the daily heat to latent heat, and resulted in the land becoming cooler than the ocean. When the resulting vapour shrunk back to water, this caused moist air above the ocean to move inland, joining air above plants already moist from transpiration, causing condensation into rain. The densest vegetation grew in the lowest landscape positions, causing most of the water to flow around the dense vegetation on higher ground where organic debris was deposited on contour at the high-water mark, producing sills that formed steps in the landscape. This created a stepped diffusion hydroponic system that efficiently irrigated the land with slow flow of water through the root zone of plants and abundance of fire-retarding plants. This wonderfully efficient system managed itself through feedback systems in a well-balanced manner, without human intervention.

When humans first settled in Australia 60,000 years ago, they started to upset the balance, largely through burning of vegetation. However, when European settlers arrived around 230 years ago, they brought with them hoofed animals that destroyed vegetation at a much faster rate. This resulted in water flowing rapidly through the centre of valleys, eroding soil and cutting deep channels that facilitated the draining of the surrounding landscape. The bared soil was heated by the sun, resulting in the land often becoming much hotter than the ocean for prolonged periods every year, so blowing air with evapotranspired moisture onto the ocean and reducing rainfall over the land. Since such problems and worse had previously been solved by plants, the Australian landscape provides a laboratory from which the lessons learned can be applied to healing damaged land.

#### ***Applied solutions and potential for Namibia***

Andrews (2008) has recreated the conditions for ecosystem self-management, making it possible to restore efficient water and nutrient cycles on some Australian farms. These successes provide good learning opportunities for application in Namibia, where hoofed animals have been present for a lot longer and land degradation has proceeded at a rapid pace.

When Andrews visited Namibia in September 2015, he toured through some rangelands of Khomas, Omaheke and Otjozondjupa regions, interacting with a few farmers and academics. During the tour, various questions and doubts were raised, needing to be answered and dispelled by practical application. The lessons learned during the tour are shared with readers in this paper. It was evident that as the week-long pre-tour of the Namibian Rangeland Forum progressed, those accompanying Andrews increasingly understood and accepted his perspectives. This was not always easy as it required accepting new, somewhat counter-intuitive ideas.

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### **Core scientific principles of relevance to balanced landscapes**

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#### ***Balance created by diverse functional groups of plants***

Non-edible plants that protect themselves with thorns or poisons perform an important function in rangelands. They return carbon to the soil, modified by diverse microorganisms in the form of a large variety of carbon compounds, many of which get taken up by edible plants. An outbreak of weeds or encroachment by woody plants is a symptom of mismanagement that leads to infertile soil, which they are trying to repair. If given the chance to restore fertility, the soil will once again support more edible plants.

Species of non-edible plants fail to grow well in soils with residue from their own species. This allelopathy prevents any single species from dominating the land, contributing to high biodiversity with each species serving a particular function. Where a single species of edible plant dominates a local area, it is usually maintained by fertility produced by non-edible plants on higher ground draining down to the location dominated by edible plants.

A high biodiversity of plants that forms a natural balance ensures that particular functional groups condition the soil for high production of good quality edible plants that support fast animal growth (Brunetti 2014). One of the means whereby plants of different species interchange resources is through common mycorrhizal networks that link the plant roots (Walder et al. 2012). High biodiversity also provides a "reserves bench" which buffers against perturbations that might adversely affect a small number of species (Walker et al. 1999).

#### ***Salt management***

In a healthy agricultural landscape, salt is kept safely underground, under a layer of fresh water that keeps it there. This is achieved by a high diversity of plants acting as a solar powered pump lifting a balanced amount of minerals and carbon compounds to the surface and facilitating that the replacement water pushes the residual salt below the root zone (Andrews 2006). The removal of perennial vegetation has created salty badlands over vast areas of Australia, yet the focus continues to deal with the symptoms and not the causes. In rangelands, loss of coarse-textured topsoils also enables capillary rise of salt that didn't occur when topsoil pores were large and negated capillary action (Pringle 2002). Fortunately for Namibia, secondary salinity is not yet a major problem, but it does occur locally in lowlands denuded of perennial vegetation by overgrazing.

#### ***Atmospheric carbon cycling***

If humans get to recognise the consequences of our destructive activities such as deforestation, we could use our wetland filters to replace the material we presently get from forest, which would contribute a 20% reduction in current



atmospheric carbon accumulation. In the case of agriculture, if we could mulch farm like successful farmers did in the past but only a few are doing today, it would contribute a further 20% reduction by preventing oxidation of soil carbon and destruction of plant life. Industry (commercial activities that burn up fossil fuels) only contributes 3% to the current increased carbon dioxide level in the atmosphere. It is also reasonable to deduce that we could have some influence on the current natural 60% carbon cycling, possibly as much as 20%. This means that technically-aware humanity can have a 60% greater influence on the current situation. The result would be the moderation of climate, more quality food and certainly more water retained on the land. The difference between these two figures (10%) is the amount of the normal 60% recovery no longer being recovered due to destruction of plants. Oxidation has increased and recovery has declined due to increase in temperature, or failure to use the incoming heat from the sun (transfer from sensible to latent heat). Plants effectively can bring about a pre-industrial era carbon level in less than ten years if 30% of agriculture worldwide changed to mulch farming (the same as a good gardener uses a thick layer of plant residues to control weeds and water by virtue of evaporation loss). Mulch effectively conserves 100% conversion to soil carbon for later use by plants, compared to composting that is only 60% effective. We could test this at any scale by reproducing the process. The above figures are based on estimations at demonstration sites in the Australian landscape by Andrews, when re-instating the landscape's previous efficiencies.

### ***Water cycling in landscapes***

Attention is generally given to the effect of climate change on the water cycle, while insufficient attention is given to the inverse effect, of the water cycle on climate change (Sacks et al. 2014). In a healthy landscape, plants create a self-supporting climate. During the daytime, when plants transpire, they absorb heat. At night, much of the water vapour in the air condenses as dew, releasing heat to provide warmth. Hence the daily temperature fluctuations are greatly reduced (Pokorný et al. 2010). The condensation of dew on plants is facilitated by sharp tips on leaves and thorns that gather dew on their tips to run down stems and water roots. Under such conditions, 70% of the water that condenses as rain and dew originates from plants with only 30% originating from evaporation of seawater. However, currently 86% of the atmospheric moisture has been evaporated from the oceans (Kravčík et al. 2007). If land is wet enough then most of the evapotranspiration is from the land.

### ***Fire control***

Thanks to the natural water management by land and plants, fire was restricted to burn within the mosaic of land and water bodies. Because of the additional amount of atmospheric water, the fires did not achieve the level of heat that currently occurs, so most nights they would subside until becoming extinguished. Within the mix were always plant species that retarded fire, since the water vapour they released reduced heat to the extent that it could no longer sustain oxidation and their tissue lacked highly flammable oils. They lived together with companion plants on atmospheric water balance. However, animals then ate them out, changing landscape functions and resulting in artificial drainage (Andrews 2006).

It is illogical to assess a "desirable" fire regime based on the type of plants present at a site. They are a reflection of fire history, rather than the potential or desired vegetation. As a rule, the plants found in frequently and/or intensely burnt areas are of poor nutritional value to the ecosystem and livestock. Fire is often a short-term stimulant and long-term deficit process in terms of ecosystem productivity and herbivore nutrition. Feed the dung beetles, not the atmosphere!

### ***Nutrient cycling in landscapes***

Organic carbon compounds are the most influential nutrients in the environment, while minerals make up only a small percentage. Nutrients get moved by gravity either quickly if above ground or slowly if below ground, to the low filtering areas from where they used to get recycled back to higher ground by birds and mammals. Since humans have destroyed much of the fauna, both directly and indirectly, the nutrient recycling no longer takes place unless humans carry the fertility from filtering areas to the high ground. The nutrient cycle in a healthy landscape is coupled to the water cycle (Norris & Andrews 2010).

Conventional rangeland science does not include this upslope moving of fertility, but rather focuses on hierarchical patch dynamics, whereby the fertility and productivity are self-organised at multiple scales and levels of organisation in a topographic sequence (Noy-Meir 1973, Tongway et al. 2003, Pringle et al. 2006). Indeed, a clear example of upslope movement of fertility in Australian rangelands is that driven by wind (Gillieson et al. 1994). The upslope movement of fertility may well have been a key part of Australia's landscapes, but what if it wasn't, if it helps address the loss of topsoil and fertility that are well documented (McKeon et al. 2004)?

### ***Stepped diffusion hydroponic system***

This is a particularly contentious and important concept that needs to be well understood. In a healthy landscape, raw material tends to be deposited at high points so that it can be decomposed to soluble compounds moved by water to be combined into living matter. From lower points it needs to be transported back from where gravity moved it, a function that used to be performed by an abundance of birds and mammals. In the process, a natural sub-surface irrigation took place (environmental production line) facilitating an ability for the multiple compounds to be reorganised by many plant species and accompanying micro-organisms that fed the soil food web (Andrews 2008).

Most water and the sediments and nutrients it carried were diverted away from the densely vegetated valley and floodplain floors because water found it easier to progress downslope in slightly higher areas as a result of the thick vegetation and consequent obstruction to flow in the lowest areas.

The “steps” can be at a whole catchment scale, reflecting geological formations such as quartz and banded ironstone ridge gaps as with the Murchison River in Western Australia (Pringle & Tinley 2003), but they can also be at more local scales and based on soft sediments reinforced by vegetation in a positive feedback loop. It is this level of finer scaled water harvesting that is being lost from rangelands globally by poor grazing management (Tinley & Pringle 2013). In Namibia, local changes in the colour of the soil indicate where the “wet spots” used to be: the soil is darker due to retained organic carbon. They can be restored quite easily by helping rebuild the vegetation that allowed the subtle sills to develop. This can be done by placing anchored bush where the sill used to be and thereby regrowing the sill.

### Case studies of sites visited

#### **Mulch gardening**

Vegetable gardens that were visited during the tour received irrigation water onto the soil from above. Since most of the gardens were located on gradients, even if very slight, the irrigation could be more effectively applied through a thick layer of mulch upslope of the vegetables, to allow water to pick up nutrients from the decomposing mulch and flow slowly past vegetable roots, permitting them to drink and feed as required. Long lines of drip irrigation could be applied along heaps of mulch in which creeping plants such as pumpkins could be grown, whose roots would assist in converting the lower layers of mulch to compost more efficiently than done in compost heaps (Figure 1). Key species for processing mulch include pumpkins, potatoes and tomatoes.



**Figure 1:** Drip irrigation through thick mulch strips alongside which vegetables grow.

#### **Mulch for trees**

Rather than placing irrigation basins around trees, a pond could be dug on the upslope side of each tree to be filled with mulch. The soil dug from the pond could be placed on each side as wing bunds to divert rainfall runoff into each side of the pond, cancelling out the energy of the flowing water from each side and allowing the calmed water to seep through the mulch and feed the tree roots slowly over the weeks after rain fell.

#### **Bush mulch on contour**

Along several of the roads driven during the tour were piles of bush that had been chopped in the road verges (Figure 2). Similarly, many of the rangelands driven past were encroached by bushes. If bushes were cut in strips along contour, the cleared bushes could be piled on the upslope side of the cleared strips where they would trap mulch and soil carried in runoff water, thus eventually creating a step in the landscape, below which the strip of grass would slowly receive nutrition from the bushed strip upslope. The bush mulch on contour would be ideal for growing edible bushes and trees in, so that they could be protected from herbivores until large enough to withstand occasional browsing. They could be planted either by seed scattered into the bush line, or seedlings transplanted into it, or cuttings or larger truncheons planted among it. If planted as seedlings, the longstem tubestock method of raising and planting the seedlings would ensure better survival (Australian Plants Society 2010). Cut bush can also be used to construct filters that divert water flow away from gullies.



**Figure 2:** Bush lines provide good opportunities for establishment of edible plants, especially if aligned on contours.

#### **Recreating stepped landscape**

Apart from recreating steps in the landscape by bush mulch placed on contours as described above, ditches can also be dug on contour to speed up the effectiveness of controlling water and nutrient recycling in the landscape. If water that would otherwise flow into pans or dams is instead diverted into contour ditches on either side, the water remains higher in the landscape and is encouraged to seep slowly as in-ground water. This conserves water far better than allowing it to evaporate from exposed pans or dams. Spillways could be constructed where the contours loop around ridges, to spread the spilled water safely onto high ground, maximising its spread over the ground below. If the contour ditches are widened at spillways to form a settling pond, the water is calmed before spilling and can further pick up nutrients if mulch is placed at the outlet of the settling pond. Trees on the higher ground will create the fertility to slowly move downhill to settle in depressions from where plants could be harvested for returning the fertility uphill, if there are insufficient birds and mammals to perform that task.

### Revegetating bunds

Bunds are usually constructed from dug soil that is heaped in lines (Figure 3), sometimes referred to as mounds, banks or berms. Bunds constructed on the downslope side of contour ditches provide the opportunity to grow grass to stabilise the bund. If the bund is built wide, with its downslope side having a low gradient, and if a shallow (less than 30 cm) ditch is dug below it to collect runoff from this bund slope as well as some of the water spilled over spillways, then water will be available for both the grasses grown on the bund and trees grown below the bund. When the ditch fills, the water spills over the length of the ditch, calmly. Mulch pits can be located on the upslope side of the contour bund to process and spread mulch along the bund's length to release in-ground fertilised water.



**Figure 3:** Water from a rain shower of 11 mm is held back in a contour ditch with bund for growing useful trees in a fruitful landscape.

### Diversion ditches on contour rather than on gradient

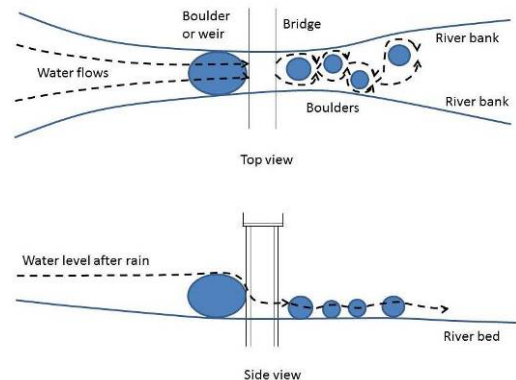
Diversion ditches had been aligned at a gradient of 1:200 to bring runoff water from surrounding land to a 30 ha fruitful landscape (Zimmermann et al. 2015). During the tour, Andrews pointed out that such gradient ditches do not appear in balanced nature. His advice was to construct the ditches on contour instead, with the ditch deeper in the direction of where the extra water needs to be moved. In this way, the spillage during intense rain would occur equally along the contour, rather than at a weak point to which the fast-flowing water would otherwise be directed.

### Pulsing release of water

Where pipes are placed through dam walls to allow water to trickle through slowly (Figure 4), it would be more effective to place a valve at the outlet, so that the stored water could be released in pulses as happens frequently in nature (Middleton 1999). This also avoids unnatural waterlogging of soil that would otherwise become anaerobic and unproductive. In case water is held in a contour ditch and could be syphoned over to lower ground, the opportunity also exists to alternate the locations from where the water is released, as happens in nature by reeds growing where water previously escaped, thereby blocking further releases there.



**Figure 4:** A pipe in a dam wall provides opportunities to pulse the release of water to favour grass growth, if fitted with a valve at its outlet.



**Figure 5:** Diagram of infrastructure above and below bridge to speed flow under bridge and absorb energy of the flow below the bridge.

### Opportunities provided by bridges

Road bridges usually occur in the narrowest sections of valleys (pinch points). The water upstream of a bridge can be made to flow more rapidly under the bridge, such as by constructing a weir or placing a large boulder to cause the water to drop steeply before flowing rapidly under the bridge. By facilitating the speeding of a larger quantity of water than the original channel, a much greater volume of water may pass per unit of time under the bridge. On the downstream side of the bridge, boulders placed at alternating sides cause the water to flow in a figure-of-eight pattern around them, absorbing the energies of the waters flowing on each side, thus calming them (Figure 5). Varying designs can achieve as much as seven times the volume of water under the bridge and it is then possible to use that energy to create a wave below the bridge that automatically exerts back pressure, helping to support bridge installations and preventing the water from gaining enough energy to erode the downstream area for a considerable distance.

## Conclusion and recommendations

The visit by Peter Andrews has generated interest in the potential that Natural Sequence Farming methods have for Namibia. What is now required in Namibia is the building of capacity to test these ideas and opportunities. This might best be achieved through a visit to Australia by interested Namibians and then a return visit from Andrews once some of his core ideas have been tried out in Namibia. All of the ideas need to fit within a more holistic management context regarding how infrastructure and grazing are planned and managed.

It is also recognised that any national, let alone international initiative, needs to respect and acknowledge the work of others who address these issues. Natural Sequence Farming has great potential, but is focused on some key issues that can be complemented by others' work. However, the focus on rangeland restoration is important to Namibia and builds on existing work and understanding.

Finally, can we farm better? Benign neglect as an illusory strategy for global conservation and deforestation, burning of fossils fuels and so forth has created this crisis. To use our planet wisely, we need to manage our fundamental energy source (the sun), the water cycle and nutrients. Plants will be the key. For Namibia, why don't we focus on these key functions for local outcomes. And why don't we find partners in Australia and elsewhere who accept that we can "farm the planet" back from the brink?

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# Options to improve soil fertility with national resources

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## Abstract

Most fertilisers applied to Namibian crop fields are imported, yet Namibia has vast resources that could be used to improve soil fertility. Organic fertilisers such as animal manure and compost could be sourced from farms, or grown on the crop fields, such as green manures and fertiliser trees, or harvested from nature, such as kelp and guano, or sourced from abattoirs or other agricultural processing facilities, such as blood, bone, horn, feather, fish and seed meals. Inorganic fertilisers are usually sourced from quarries or mines on land, while others can be extracted from seawater or salt. Some locally-sourced fertiliser materials could be applied directly to the soil, perhaps after simple processing such as crushing or milling, or after more complex processing such as through chemical reactions. Rock salt and brine solution have been analysed to assess their suitability for extraction of nutrients from which to manufacture chemical fertilisers as by-products of purifying the salt for industrial uses. It is important that harvesting of rock salt and brine from pans north of Cape Cross be done without disrupting the natural regeneration through underground connection to the sea, to ensure sustainability of these valuable resources. Labelling of fertilisers should include a breakdown of the major elements contained, so that farmers who test their soils could determine appropriate application rates to balance minerals that will produce crops of good quality.

**Keywords:** by-products, fertiliser, inorganic fertiliser, manure, organic fertiliser, productivity, soil fertility

## Introduction

The nutritional quality of crops has steadily worsened in the last decades, due to decline in soil fertility resulting from unsustainable farming practices. Studies that show decline in mineral content of agricultural products, such as that by Thomas (2007), usually use data from the 1940s as their baseline, since this is when laboratories started measuring the mineral contents of crops. However, there were warnings of decline in soil fertility taking place well before then (Hensel 1894), so the benchmark mineral contents of crops should actually be higher. The consequences of declining soil fertility are not only weaker crops and worsening human health, but also greater outbreaks of pests (Chabboussou 2004) and weeds (Gilman 2011).

Soil fertility is contributed to by both mineral and non-mineral elements. The mineral elements tend to cycle between rocks, soil, water and living organisms. The non-mineral elements, such as C, H, O and N, tend to cycle between the soil, water, living organisms and the atmosphere. Soil fertility improves when plants grow and their minerals get recycled on the land. With the help of beneficial soil microorganisms and the humus flywheel (Lovel 2014), plants take up mineral elements from the soil, combine them with the non-mineral elements to form organic matter, which then gets broken down to recycle the elements. Conversely, soil fertility declines when plants are harvested and removed from the land or prevented from growing (Andrews 2006). If there are insufficient plants to protect the soil, weathering results in most of the non-mineral elements ending up in the atmosphere and most of the mineral elements being washed into water bodies such as pans and the ocean.

Namibian soils tend to lack fertility, being very old and mostly sandy. Namibia also lacks volcanic activity, which renews mineral soil fertility in other parts of the world. In the past, an abundance of birds and animals used to bring some of the fertility that had been washed into the oceans back inland (Shepard 2012), but their numbers have greatly decreased due to various human influences. Therefore, it is necessary for soil fertility to be restored by other means in order to grow good quality crops.

Most fertilisers applied to Namibian crop fields are imported, yet Namibia has vast resources that could be used to improve soil fertility. Organic fertilisers such as animal manure and compost could be sourced from farms, or grown on the crop fields, such as green manures and fertiliser trees, harvested from nature, such as kelp and guano, or sourced from abattoirs or other agricultural processing facilities, such as blood, bone, horn, feather, fish and seed meals. Inorganic fertilisers are usually sourced from quarries or mines on land, while others can be extracted from seawater or salt. Some locally sourced fertiliser materials could be applied directly to the soil, perhaps after simple processing such as crushing or milling, while others would first need to undergo more complex processing, such as through chemical reactions.

Soil microorganisms play important roles in making nutrients available to crops. Soil amendments that favour beneficial microorganisms therefore enhance the efficiency of fertilisers. Most Namibian soils are low in cation exchange capacity (CEC) because they lack sufficient clay or humus to adsorb sufficient cations. Small quantities of humic substances such as fulvic and humic acids could be used to amend soil and raise its CEC, but this usually implies high cost. Although many forms of organic matter would be cheaper, much larger quantities would be required and their benefits would be short-lived unless reapplied annually. Clay or biochar may offer cheaper alternatives, and combinations may offer synergies.



## Fertilisers from organic sources

### Animal manure

The quality of manure varies tremendously, depending largely on the quality of food eaten by the animals producing the manure. Those that feed on insects or sea fish tend to produce the best-quality manure, such as guano from bats and sea birds. Dairy cows tend to produce the best-quality manure among ruminant animals, because they usually feed on good-quality pastures or fodder. Animals perform the function of spreading manure and urine as they graze on pastures or residues remaining in crop fields after harvest. Mobile night kraals can be moved around crop fields to spread the fertility (Sibanda et al. 2016). However, fresh manure will lose quality if left above ground (Figure 1). Dung beetles perform the function of getting fresh manure underground if they are given the chance to do so (Walters 2008).



**Figure 1:** The quality of manure deteriorates if it is left exposed to the sun.

### Green manure

The greater the diversity of cover crops (Clark 2007) grown for green manure, the more effectively the soil's fertility can be enhanced. Legumes fix nitrogen while grasses grow massive root systems. Dicotyledonous species tend to specialise in particular minerals that their taproots bring from deep underground, while grasses are more generalist, with their roots bringing up a greater diversity of elements (Walters & Fry 2006). Ploughing in of cover crops disturbs the soil and only results in temporary increase in soil carbon before it gets rapidly broken down. No-till or minimum tillage allow for longer-term build-up of soil carbon and fertility. However, if glyphosate herbicide is applied to terminate the cover crop in preparation for growing the production crop, then soil fertility may be harmed by chelation of some mineral elements (Eker et al. 2006). This harm could be avoided by mowing the cover crop, although it may then get blown away by the wind. Use of a roller-crimper (Kornecki & Price 2010) could inactivate and lay down the cover crop, while leaving it firmly rooted in the ground. This also occurs if animals graze down the cover crop, with the added advantages of spreading beneficial microorganisms in their dung and diversifying farm income. Weeds may also be helpful in bringing minerals that a soil is short of from underground to the top soil, if slashed and left to decompose on the surface (Gilman 2011). Smallholder farmers in the tropics, including those in semi-arid regions, often develop cover crop systems that are grown together with their production crops (Bunch 2012).

### Trees

Tree species with deep taproots are also effective green manure plants (Leakey 2012), sourcing minerals from deeper soil horizons than herbaceous plants are able to do. In addition, they provide the extra advantage of breaking the speed of the wind and reducing temperature extremes for crops growing in their vicinity, which becomes more critical with climate change (Nabhan 2013). Farmers usually select multi-purpose tree species for growing in or around their crop fields, to provide various products and perform different functions (Leakey 2012). Leguminous trees that regrow vigorously, preferably without thorns, are useful for "chop and drop" mulching (Thurston 1997). Tall-canopied trees, such as *Faidherbia albida* (Figure 2), which is commonly known as the fertiliser tree, literally cover crops (Umar et al. 2013). If their canopies grow too dense and shade the crops excessively, they can be pollarded by chopping their branches high up, from where new branches will regrow. An added benefit of chopping trees that grow among crops is that the tree roots respond by excreting sugary exudates which feed soil microorganisms, which subsequently feed the nearby crops (Huang et al. 2014). If rows of trees are planted in crop fields, they can either be aligned at right angles to prevailing winds, or along contour ditches where they could also benefit from harvested rain water (Zimmermann et al. 2015).



**Figure 2:** A large canopy of *Faidherbia albida* protects nearby fruit trees from frost at Kaisosi.

### Compost

Compost is usually uneconomical to use on agronomic fields, unless it is diluted through use of aerated compost tea and the microorganisms are fed by roots of cover crops. On the other hand, compost may be economical on horticultural fields if the farmer has insufficient land to restore soil carbon and eliminate soil pests and diseases by rotating with diverse cover crops. The quality of compost depends largely on the quality of ingredients used to make it, assuming that the correct

carbon to nitrogen ratio of roughly 30:1 in the starting materials has been used. Clay or crushed biochar at 1-2% of the starting volume is also an essential ingredient if real humus is to be formed in the compost (Solomon 2013). Composted animal manures from feedlots and poultry houses need to be checked for contamination with chemicals and excessive micro-elements that may have been used in raising the animals.

#### ***Bone meal***

Bone meal is a good source of calcium and phosphorous, while also containing a small amount of slow-release nitrogen (Table 2). However, there needs to be good microbial activity in the soil to break down the bone meal and release its elements. Dry bones can also be added to the organic matter when producing biochar. The charred bones are usually soft enough to crush by hand to produce excellent quality biochar.

#### ***Horn meal, blood meal, fish meal, feather meal, seed meal and seed cake***

If horns and dried blood from abattoirs, fish wastes from fish processing factories, feathers from poultry abattoirs and oilseeds or oilseed cake (Figure 3) are milled into meal, they become valuable fertilisers that release nitrogen and phosphorous slowly. However, some of them may be too costly for direct application to soil, especially for those with competing demand as animal feed. Value addition could take place by using these as animal feed and then applying the resulting enriched animal manure to the soil, or allowing the animals to spread it over the soil for processing by dung beetles.



**Figure 3:** Sunflower seed cake produced at Shadikongoro.



**Figure 4:** Kelp washed up on the beach near Swakopmund

#### ***Kelp***

The mineral-rich Benguela current ensures rapid growth of kelp (Figure 4) in waters along much of Namibia's coast. It used to be harvested, washed, dried and milled in Lüderitz, but sadly that seems to be no longer the case. The high diversity of micro-nutrients in kelp makes it valuable as an animal supplement and fertiliser. The latter also because of its auxins, gibberellins, and cytokinins, which are hormones that promote plant growth (Fry & Simmons 2005). Kelp can be applied to soil as a meal, or as a liquefied emulsion that can also be applied directly to crops by foliar spraying. Alternatively, value could be added by feeding kelp to animals and then using the enriched manure on the soil.

### **Fertilisers harvested from inorganic sources**

#### ***Rock dust***

Wherever rocks are processed, such as being crushed, ground or sawn, the powder or dust that is produced might have fertiliser properties, especially if from igneous rock (Lisle 1994, Moore 2001). Rock dust could be sourced from sediment of mine tailings dams or from rock-processing industries scattered throughout Namibia (Figure 5). It is not only the presence of mineral elements that provide fertility, but also the paramagnetic properties of some rock dusts that enhance plant growth (Callahan 1995). However, there is also a danger that rock dust may contain toxic elements, such as excessive fluorine, heavy metals or radioactivity, which therefore need to be checked for before being used as fertiliser.

#### ***Lime***

Most lime that is mined and crushed in Namibia, such as around Tsumeb, is dolomitic lime (Figure 6) which is high in magnesium. This may be appropriate for loose sandy soils, because it helps them to become firmer. However, this causes the soils with more clay to become too tight and compacted, requiring frequent ripping. For such soils, calcitic lime would be more appropriate (Andersen 2010). The lime deposits east of Henties Bay may be a suitable source of calcitic lime.



**Figure 5:** Granite dust being collected outside a factory in Brakwater where the granite is sawn



**Figure 6:** Spreading of dolomitic lime at Ndonga Linena

### **Gypsum**

Gypsum is comprised of calcium sulphate and therefore supplies both calcium and sulphur, which both contribute to soil fertility when they are present in sufficient quantity and balance. They enhance the efficiency of other elements supplied by more expensive fertilisers, thus saving on costs (Tiedjens 1965).

### **Seawater or rock salt**

Dilute sea water has been used to good effect to improve soil fertility and the nutritional value of the crops grown (Murray, 1976; Walters, 2005). However, extreme care is required to avoid contaminating the soil with excessive sodium, especially in soils experiencing high evaporation rates and low carbon content. There are ways to remove most of the sodium and remain with most of the high diversity of valuable micro-elements. One way is to allow sea water to evaporate in shallow pans, where sodium chloride and a few other salts crystallise on the surface, after which the brine solution underneath is syphoned out for its valuable micronutrients (Amena 2006). Another way is to raise the pH of seawater to between 10.60 and 10.78, whereupon a precipitate containing the valuable micronutrients settles to the bottom, leaving the sodium and chloride ions dissolved in the water above, which may then be syphoned off (Taylor 2007).

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## **Fertiliser by-products of chemical manufacturing from local inorganic and organic sources**

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There are many opportunities for chemical manufacturers to broaden their scope and capacity to serve Namibia's agricultural industry. The first project that we have embarked on is to work with the salt industry to produce nutrients as a by-product in the production of pure salt.

### **The extraction of chemicals and production of fertilisers from sea water and rock salt**

The process involves the removal of valuable nutrients such as potassium and magnesium that are present in seawater and rock salt. The two methods being developed to achieve this are indicated in Figures 7 and 8. Ideally all by-products from the processing should be put to use, so that the refinery has zero or minimal waste to dispose of.

Samples of various types of rock salt were collected north of Cape Cross and analysed to determine chemicals and fertilisers that could be produced and to predict the volumes of minerals that could be derived. Results from the analysis are given in Table 1. One of the research objectives was to see what the possibilities are of adding rock salt to brine solution to speed up the crystallisation process and recover some of the more valuable chemicals contained in the rock salt, such as potassium, magnesium, barium sulphate and calcium carbonate.

Potassium can be economically recovered from potassium-bearing brines by dissolving the rock salt in a brine solution (which also contains potassium) and then precipitating the potassium as KClO by the addition of sodium hypochlorite. Potassium is a very valuable agricultural product. Magnesium can be recovered as magnesium hydroxide by the addition of sodium hydroxide. Magnesium is also valuable as it is used extensively in metal alloys and in addition it has agricultural value. It is also used as magnesium chloride for the extraction of marine phosphates. Magnesium is also the central atom in chlorophyll and is a constituent of several plant oils, among others. Barium sulphate is also recovered by precipitation. It has medicinal value and is also used in oil drilling. Precipitated calcium carbonate can also be recovered from brine solutions and has multiple uses in agriculture and other industries, particularly in refining of metals and production of cement.

The analysis of rock salt and brine solution gives a clear picture of what has to be removed in order to produce a salt suitable for the Chlor/Alkali process. It indicates that very pure salt can be economically produced as a result of the recovery of by-products which have industrial, agricultural, as well as other applications. These analyses have also provided us with data to assess more accurately the extent of these resources.

The rock salt deposits north of Cape Cross regenerate through underground connection with the sea. We estimate that these rock salt deposits could sustainably yield 36,000 tons of gypsum and 2 million tons of salt per year, from which could

**Table 1:** Analysis of rock salt and soils from north of Cape Cross, on a dry mass basis. (The seven samples are: 1. ML. 147 Cape Cross SA 9/1 Rock Salt @ 0-0.1 m depth; 2. ML. 147 Cape Cross SA 13/2 – Black Rock Salt; 3. SA 9/2 147 Rock salt out- crop; 4. SA 5-1 soil impregnated with salt; 5. SA 4-1 Soil impregnated with salt; 6. ML 147 Cape Cross SA 14/2 @ 0.25 – 0.7 m; 7. ML 147 Cape Cross SA 14/1 @ 0-0. 2M)

Sample no:	1	2	3	4	5	6	7
Assay, as NaCl, % m/m [LABS 008]	74.98	95.29	96.16	96.75	92.15	98.17	97.63
Sulphate, as SO <sub>4</sub> , % m/m [LABS 005]	2.3	1.16	1.25	0.96	2.35	0.67	0.28
Calcium, as Ca, ppm [LABS 004]	10 430	4603	4865	3790	10 030	2423	1017
Magnesium, Mg, ppm [LABS 004]	318	101	35	246	346	596	375
Ca:Mg Ratio	33:1	46:1	139:1	15:1	29:1	4.1:1	2.7:1
Insoluble Matter, % m/m [LABS 003]	19.49	2.85	0.89	0.37	3.87	0.12	0.02
Total Nitrogen, as N, ppm [Kjeldahl method]	153	111	99	131	107	124	95
Iron, as Fe, ppm [LABS 001]	2	3.7	0.6	2.9	1.1	1	1.3
Chromium, as Cr, ppb [LABS 001]	33	49	23	43	<20	<20	<20
Bromine, as Br, ppm [LABS 007]	13	25	13	52	11	82	78
Aluminium, as Al, ppm [LABS 001]	1	1.3	0.3	1.7	0.6	0.7	0.6
Silica, as SiO <sub>2</sub> , ppm [Colourimetric]	9.8	9.9	9.9	1	10	1	2
Potassium, as K, ppm [AA]	238	183	150	289	330	508	355
Strontium, as Sr, ppm [ICP]	131	50	101	40	380	17	9.4
Heavy Metals, ppm [Calculation]	3	5.2	0.9	4.7	1.8	1.8	2
Ammonia, as NH <sub>3</sub> , ppm [Kjeldahl method]	1.3	1.2	1.2	1.2	1.2	1.2	1.3

\*For all samples, the concentrations of following were found to be: nickel (Ni), molybdenum (Mo), manganese (Mn), copper (Cu) and lead (Pb) <0.1 ppm; iodine (I) <0.4 ppm; vanadium (V) <20 ppm; titanium (Ti) <1 ppm; fluorine (F) <0.5 ppm; total alkalinity, as Na<sub>2</sub>CO<sub>3</sub> and %carbonates, as Na<sub>2</sub>CO<sub>3</sub> <0.1%/m.

be extracted 1,460 tons of potassium chloride, 6,100 tons of magnesium chloride and 3,800 tons of barium chloride. However, if indiscriminate mining disrupts the natural regeneration process, then these valuable resources will soon become depleted.

#### Methods to extract specific minerals used in agriculture from seawater, brine solutions and rock salt

The Swiss (Krebs) SALEX process is used to produce pure salt (sodium chloride) required by the Chlor/Alkali process by the removal of other unwanted salts that have agricultural use. These include magnesium hydroxide, potassium salts, barium sulphate and calcium carbonate. There is also a fairly recent process developed at the Great Salt Lake in Utah to extract potassium from brine solutions (Orris 2011).

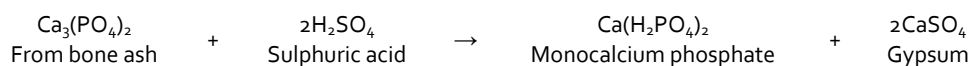
We have now designed an extraction and integrated salt purification and value addition process shown schematically in Figures 7 and 8. It has a dual function to produce a pure sodium chloride salt suitable for producing chemicals from, and to extract potassium, barium, calcium carbonate and magnesium hydroxide from rock salt, brine solutions and seawater.

The process can be implemented in two ways. The first way, indicated in Figure 7, extracts all the salts together. The second way, indicated in Figure 8, extracts each salt separately.

#### The manufacture of super phosphate

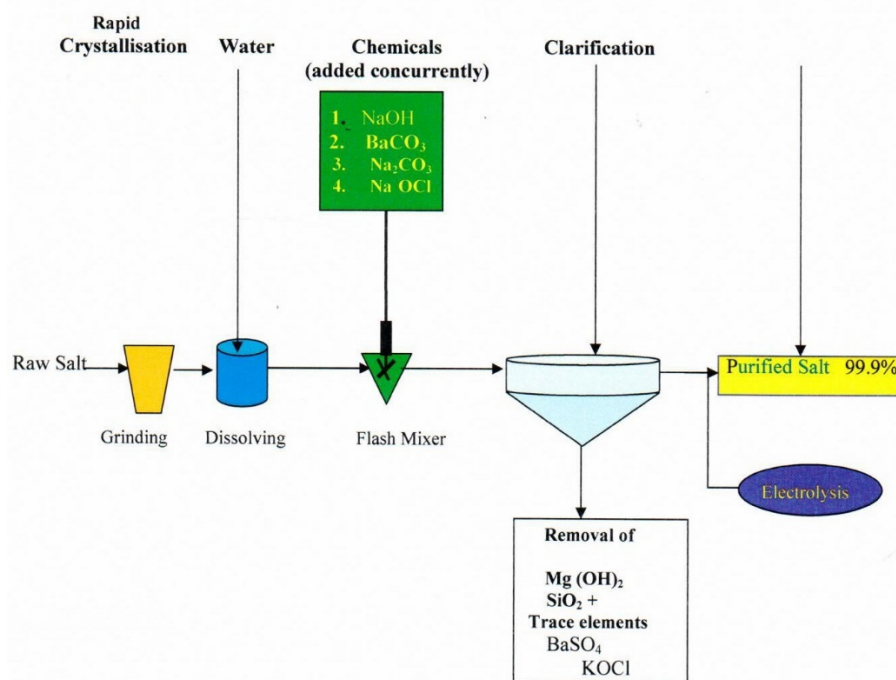
There are no large rock phosphate deposits in Namibia. There are large deposits of marine phosphate and currently the proposed utilisation of these resources has evoked much controversy. The alternative, until some kind of clarity is established, is to use bone meal to produce a super phosphate and to mix it with guano.

Bone ash is produced by burning bone meal (while capturing the generated energy) and reacting it with sulphuric acid, which is currently produced in Namibia by the copper smelter in Tsumeb. The bone ash contains approximately 42% phosphate and 55% calcium.



This form of phosphate is more soluble than that in bone meal. It also contains a number of trace elements such as Al, Ba, Fe, Mg, Mn and Ti. This can be mixed with guano, the N, P and K contents of which appear in Table 2.

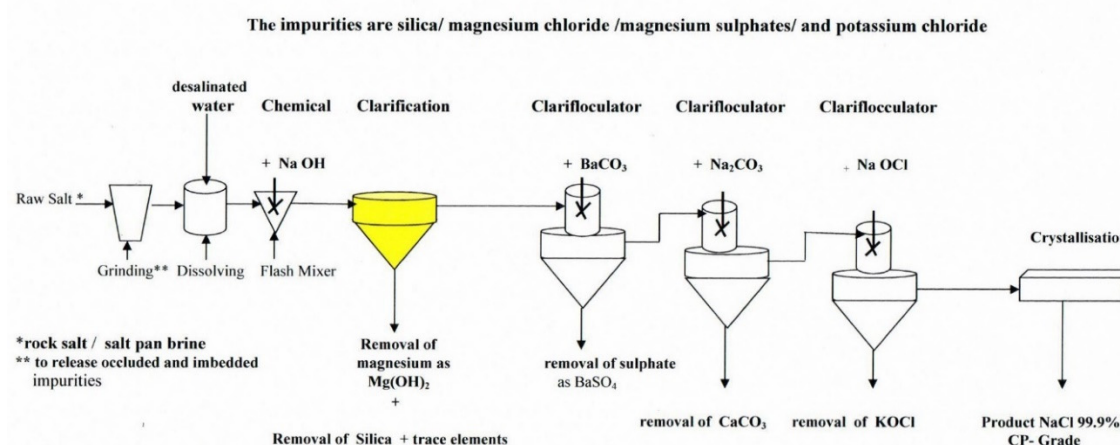




**Figure 7 (left):** The combined precipitation option to process rock salt into purified salts, extracting all the salts together.

**Figure 8 (below):** Second option of processing rock salt into purified salts which can each be extracted each separately and then used to manufacture fertilisers, as indicated in the text

#### The Polex ISPAVAP Process Chain (Integrated Salt Purification And Value Addition Process)



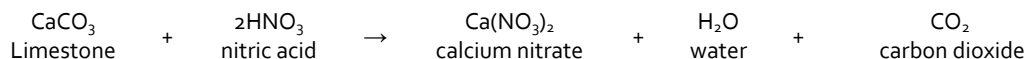
**Table 2:** Percentage of major elements provided by various types of Namibian fertilisers. Blank cells do not necessarily represent zero, but rather data that are unavailable.

Type of fertiliser	Percentage of element in the fertiliser							
	N	Ca	Mg	K	Na	P	S	Fe
Bone meal	4.0	21.0				10.0		
Compost (Gottesgabe Buschprodukte)	0.5	1.1	0.1	1.0				0.6
Vermicompost (Gottesgabe Buschprodukte)	1.0	0.6	0.1					
Guano (Seabird Guano)	15.3			2.0		4.3		
Sunflower seed cake	4.2	0.1	0.1	0.3		0.7		
Dolomitic lime (from near Tsumeb)		20.0	14.0					
Calcitic lime (from east of Henties Bay)		45.2	0.2					
Gypsum (Elspe Minerals)		17.7	0.2	0.5	0.3		13.7	0.4
Rock salt		0.1	0.2	0.1	37.0		0.3	
Kelp	0.8	2.2	0.8	5.7	5.8	0.3		0.2



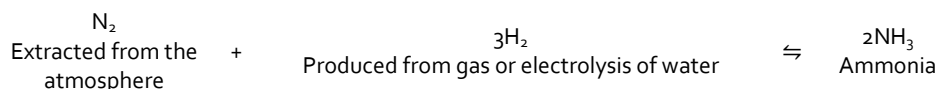
### Manufacture of calcium nitrate

Calcium nitrate can be used as a fertiliser from the reaction:

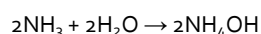


### Manufacture of ammonium nitrate, calcium ammonium nitrate and ammonium sulphate nitrate

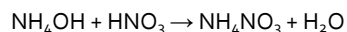
Ammonia gas is dissolved in water to produce ammonium hydroxide. Because ammonia is used extensively in the uranium industry and the explosives are expensive, it is viable to manufacture ammonia and from this to make ammonium nitrate. Ammonia ( $\text{NH}_3$ ) is normally produced using the Haber process based on the reaction:



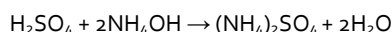
The dissolution of  $\text{NH}_3$  in water results in ammonium hydroxide ( $\text{NH}_4\text{OH}$ ) according to the reaction:



The ammonium hydroxide can be reacted with nitric acid ( $\text{HNO}_3$ ) to form ammonium nitrate ( $\text{NH}_4\text{NO}_3$ ), which is a fertiliser:



Because sulphuric acid ( $\text{H}_2\text{SO}_4$ ) is produced in Namibia, it makes it easier to produce ammonium sulphate. This is done as follows:



Calcium ammonium nitrate is made by adding calcite or dolomite to the ammonium nitrate melt before prilling or granulating. Ammonium sulphate nitrate is made by granulating a solution of ammonium nitrate and ammonium sulphate.

### Manufacture of urea

Urea fertilisers are produced by a reaction of liquid ammonia with carbon dioxide. The process steps include solution synthesis, where ammonia and carbon dioxide react to form ammonium carbamate, which is dehydrated to form urea; solution concentration by vacuum, crystallization, or evaporation to produce a melt; formation of solids by prilling (pelletising liquid droplets) or granulating; cooling and screening of solids; coating of the solids; and bagging or bulk loading. The carbon dioxide for urea manufacture is produced as a by-product from the ammonia plant reformer.

### Potential of producing fertilisers from Kudu gas

Kudu gas would be ideal for production of ammonium fertilisers, such as ammonium sulphate and ammonium nitrate. This is because Kudu gas consists of almost pure methane, unlike most natural gas from other parts of the world that would first need to have its sulphur removed at high cost before its hydrogen could be reacted with nitrogen from the atmosphere through the Haber process to produce ammonia for use in fertiliser production. We believe that it would be more profitable to produce fertilisers and industrial chemicals from Kudu gas than to burn it for energy, additionally more jobs could be created and it would last a lot longer than the predicted 22 years of energy supply (SAIEA 2006). The plentiful sources of renewable energy available in Namibia could rather supply energy needs.

### Multi-nutrient potassium nitrate fertiliser from debushed *Terminalia sericea*

Wood from debushed *Terminalia sericea* is ashed (while capturing the generated energy). The ashes, which are strongly alkaline mainly because of the presence of potassium oxide ( $\text{K}_2\text{O}$ ), are treated with nitric acid, which could also potentially be produced in Namibia. This, in addition to forming potassium nitrate, dissolves other nutrients in the ash. The resulting liquid can be "loaded" onto some inert material such as sepiolite or biochar to make a fertiliser that is not too concentrated.

## Amendments that improve efficiency of fertilisers

### Clay

The low CEC of sandy soils could be raised by mixing in some clay. Small quantities of clay could be sourced from pans and termite mounds. There are a few deposits in Namibia from where larger quantities of clay could be sourced, but at high transport costs. The bentonite clay that is sold for mixing with animal feed is unfortunately sodium bentonite, which would be unsuitable for amending soil due to its sodium content.

### **Sepiolite**

Sepiolite is a type of clay that has many applications including agricultural applications as absorbents and carriers for chemicals and pesticides. Sepiolite improves stability and components suspension of fluid fertilisers in spraying or fertigation applications. Drying at a temperature high enough to remove the zeolitic water in the structural channels improves the sorbent properties. In Namibia, sepiolite deposits are located some 120 km south east of Gobabis in a number of pans.

### **Biochar**

Biochar has the additional benefit over clay of also adsorbing anions and housing beneficial microorganisms. Biochar could be produced on or near crop fields from organic matter such as prunings from trees and crop residues, to use as soil amendment (Taylor 2005), preferably after conditioning by microorganisms (Lehmann et al. 2011). The biochar could be conditioned by composting it (Fischer & Glaser 2012), or throwing it into animal kraals to get mixed with dung and urine, while being crushed by animal hooves (Zimmermann & Amupolo 2013). For further value addition, biochar can be fed to animals, which then spread their manure to be incorporated into the soil by dung beetles (Joseph et al. 2015).

### **Humic acids**

Humic acids are usually extracted from carbon-rich deposits such as leonardite, soft rock phosphate and peat (Lovel 2014). Since these are unavailable in sufficient quantities in Namibia, it may instead be possible to extract humic acids from torrefied wood (Foidl in lit.). The wood for torrefaction could be sourced from abundant encroached bushes, which should be harvested in a regenerative manner such as along contour strips (Bruwer 2014).

### **Microbial inoculants**

In soils that lack microbial activity, such as those that have been abused by chemical inputs, it could be useful to apply microbial inoculants such as effective microorganisms (EM) (Higa 1996, Xu et al. 2000). If the soil also lacks sufficient carbon to feed the microbes, then some dilute molasses could be applied to favour bacteria, or woody material to favour fungi. Inoculating crop seed with species-specific nitrogen-fixing bacteria or mycorrhiza fungi (Medina & Azcón 2010), may result in significant yield increases with minimal costs.

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## **Recommendations**

If entrepreneurs proceed to make use of the many opportunities that exist in Namibia to harvest, process and sell fertilisers and soil amendments from local resources, it is important to label their products with results from analyses of elements contained in their products, such as those appearing in Table 2, although for all major elements rather than just selected ones. This will allow farmers who have analysed their soils to determine the appropriateness of different fertilisers and calculate application rates to achieve optimal balance of the major elements (Astera 2010, McKibben 2012, Solomon 2013).

The mining of rock salt deposits north of Cape Cross needs to be done with care in a sustainable manner that respects the natural regeneration process through underground connection with the sea. Fertilisers should also be produced from the high-quality Kudu gas.

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## **Acknowledgements**

The National Commission on Research, Science and Technology (NCRST) is thanked for funding a research project at the Namibia University of Science and Technology on integrated agricultural technologies, for which this paper provides background information. Uwe Kahl kindly provided results from analyses of gypsum and rock salt, while Mr. Sarnow kindly provided results from analysis of guano. Georg Friedrich kindly provided results from analyses of compost and vermicompost, kelp and seed cake. The other data in Table 2 were found from labels on packaging or sourced from the Internet.

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# Addressing informal settlement growth in Namibia

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## Introduction

Namibia is undergoing a rapid and major transition from a rural-based society to one based largely in urban areas. This transition is most visible in rapid urban growth, especially in informal settlements that accommodate poor families in shacks on the edges of towns. Namibia's urban areas now have some 140,000 informal houses, a number likely to double over the coming seven or eight years if this trend is not addressed urgently. Similar patterns of rapid, unplanned informal settlement growth are to be seen elsewhere in southern Africa, and in developing countries around the world.

The economic, social and environmental costs of informal growth and unplanned urban development are huge for Namibia as a country, and as a society. New forms of poverty and inequality will be entrenched over generations to come if towns fail to develop in ways that facilitate the transition from rural to urban society. By many standards, the continuous rapid growth of informal settlements is one of Namibia's biggest development challenges.

This article is largely based on results from recent research on informal settlements in Namibia, implemented by Development Workshop Namibia (Weber & Mendelsohn 2017). It provides information about the growth and characteristics of informal settlements in Namibia and describes how local authorities deal with the phenomenon. Based on promising approaches used by some local authorities, the research further makes recommendations on how informal settlement growth could be turned into formal urban growth, contributing towards urban development that is more socially just, economically efficient and environmentally sustainable.

## Informal settlements in Namibia

### Urban growth patterns

Namibia's transition from a mainly rural towards a mainly urban-based society is reflected by the growth rates of Namibia's towns. Since 2001, census data indicate a decline of the rural population together with continued rapid growth of the urban population. From 2001 to 2011, for example, the rural population declined by more than 20,000 inhabitants, while the urban population increased by more than 300,000 people, fuelled mainly by urban immigration and Namibia's generally expanding population (Figure 1).

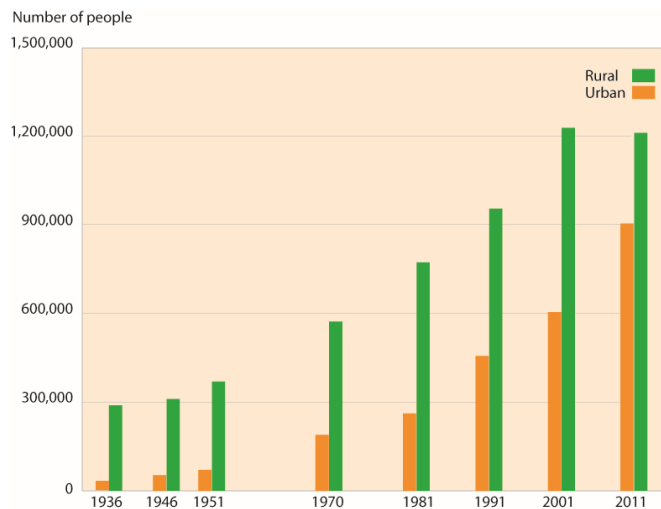
### Informal settlement growth

Informal settlements in Namibia's towns are growing much faster than the formal parts of towns, contributing most to overall urban growth. For example, from 1991 to 2011:

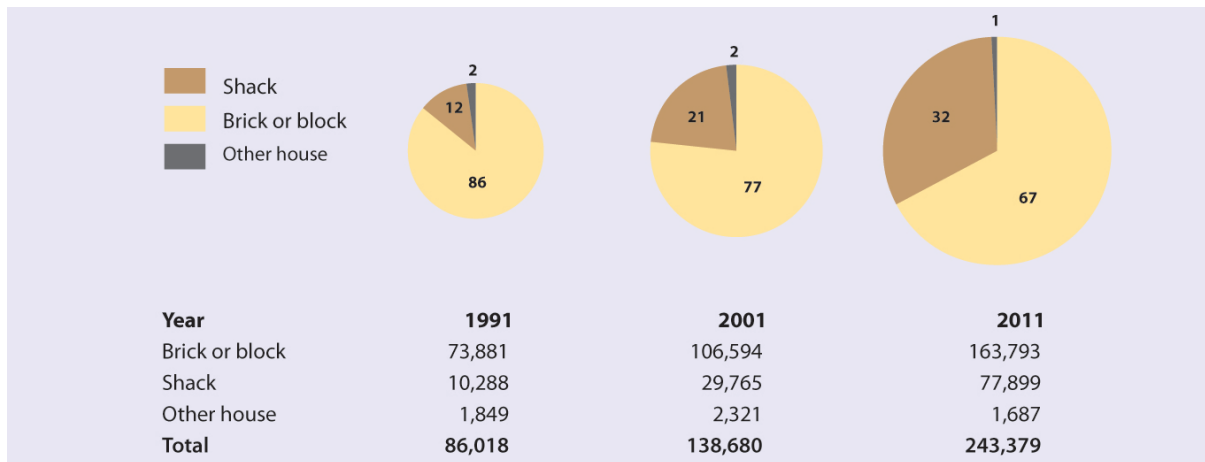
- Brick or block houses roughly doubled from 73,881 to 163,793.
- Informal houses increased more than sevenfold, from 10,288 to 77,899 (Figure 2).

At the time of the last census of 2011, about one third of Namibia's urban population was living in shacks. If these growth rates continue, which is probable if no urgent, decisive and systemic action is taken, then:

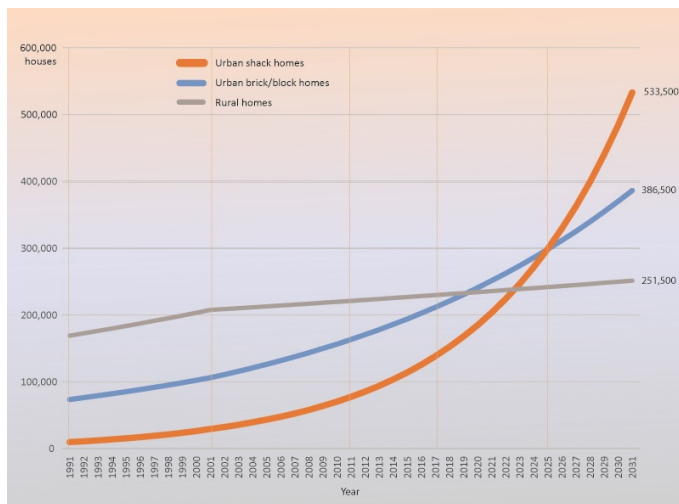
- The number of urban shacks will outnumber formal urban brick/block houses by 2025, and they will outnumber all rural houses by 2023. The predominant form of housing in Namibia will then be urban shacks (Figure 3).
- By 2030, Namibia will have over half a million urban shacks in which about 2 million people will live.



**Figure 1:** The number of people in rural and urban areas recorded during censuses between 1936 and 2011.



**Figure 2:** Pie diagrams of the percentages, and a table of the numbers of different house types in 1991, 2001 and 2011



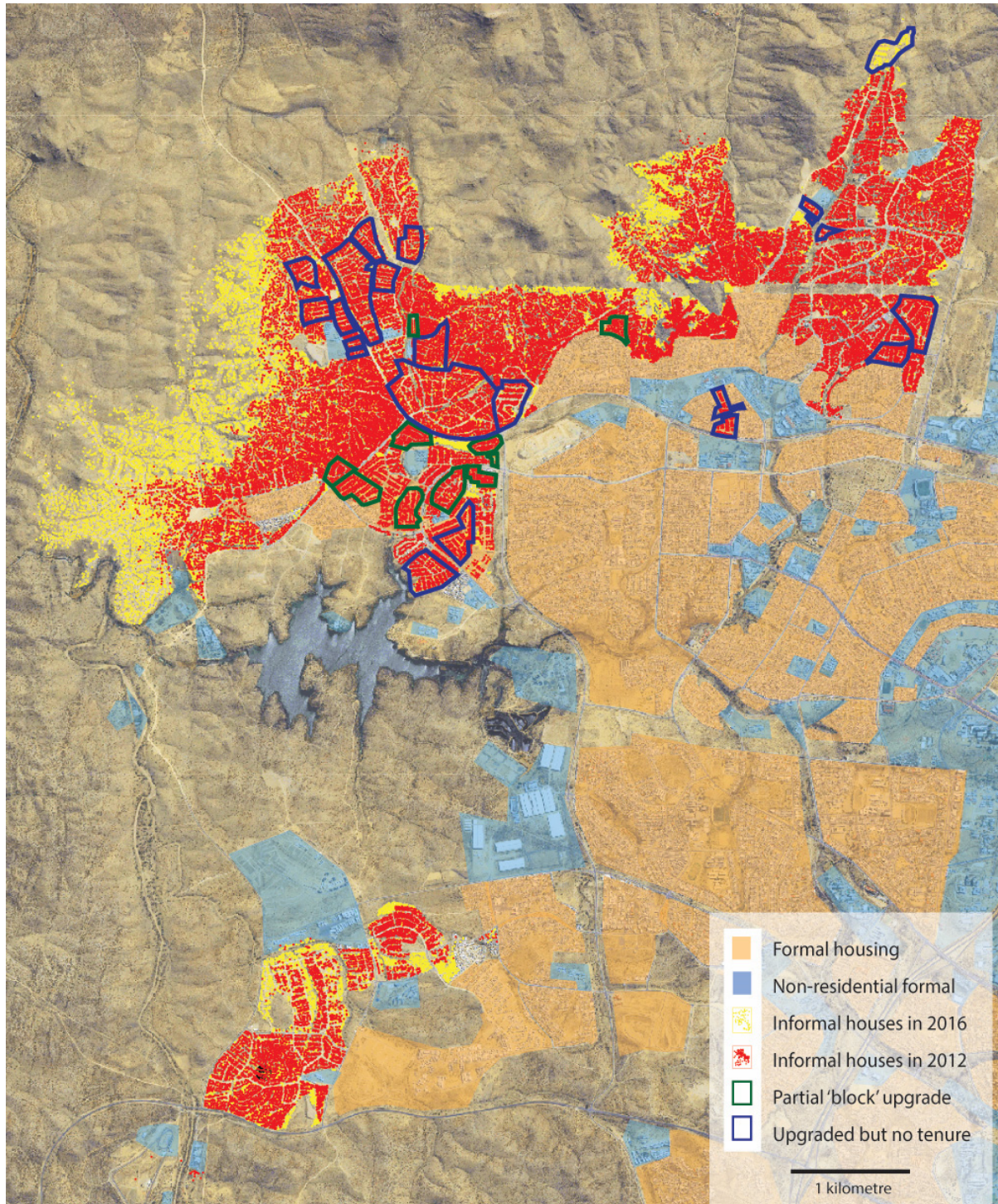
**Figure 3:** Numbers of formal (brick or block) and shack homes in Namibia between 1991 and 2011, and projected forwards to 2031

The geographic expansion of Namibia's informal settlements can be assessed by comparing high resolution satellite imagery from different years. The comparison of satellite images of 2012 and 2016 in the case of Windhoek, for example, show that more than 15,000 shack-like structures have been built during that period of time, meaning that some 3,500 shack-like structures have been erected each year (Figure 4).

The roof top counts based on satellite imagery were made in the context of the above-mentioned research (Weber & Mendelsohn 2017). Probable informal houses were identified and mapped using the following criteria: they were in areas that lacked a formal structure or layout (for example, of roads and plots); the houses were similar in size (to exclude larger shops, stores or tiny shops and toilets); spacing between adjacent houses was roughly similar in any area; and the roofs of these houses lacked the structure, pattern and size normally seen in low income formal housing. The shack count for each town was done twice by different technicians to minimise errors and bias. Nevertheless, the counts are certain to include some structures that were not houses, and, conversely, to exclude some that were houses. Nevertheless, the counts provide measures of the growth, distribution, size and nature of informal housing.

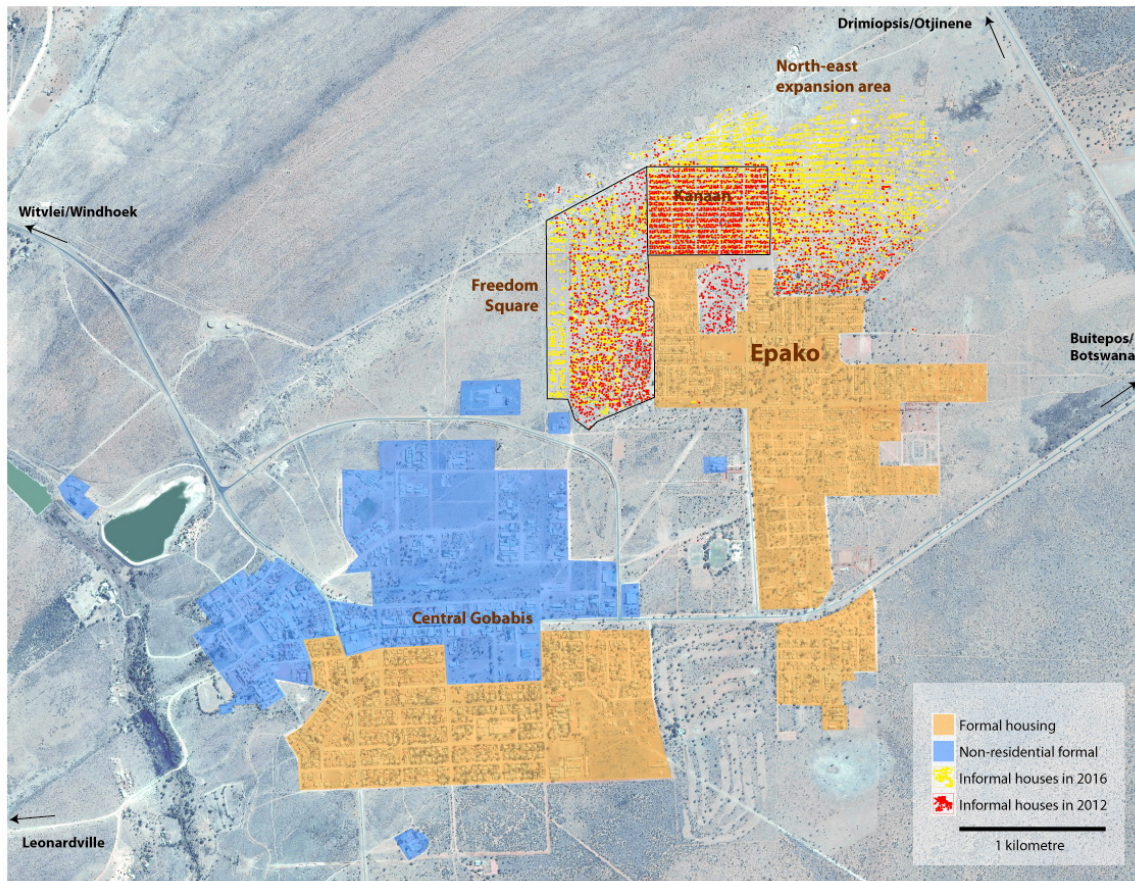
The situation in many of Namibia's smaller towns is similar, as is clearly visible in the case of Gobabis for example, in Figure 5. In Gobabis, some 3,226 shacks were counted on a 2012 satellite image, and 5,297 shacks on an image of 2016. This is equivalent to more than 500 new shacks per year, at a growth rate of more than 13%.





**Figure 4:** The red dots represent shacks that were mapped on the 2012 image of Windhoek, and the yellow dots those shacks that were built from 2012 to 2016. The areas in blue are block erven purchased by groups of residents, and which are registered by the Surveyor General. The internal subdivision of individual erven was done by the residents themselves and these individual erven are not registered. Nevertheless, once a group of residents has a purchase agreement for a block erf with the City of Windhoek (CoW), they are allowed to build with permanent construction materials. The areas upgraded by the CoW are planned and have different levels of services, but the areas are not proclaimed and residents do not have official titles for their erven.





**Figure 5:** Satellite image of Gobabis taken in 2016 showing the location of informal houses mapped in an earlier image from 2012 and then in 2016.

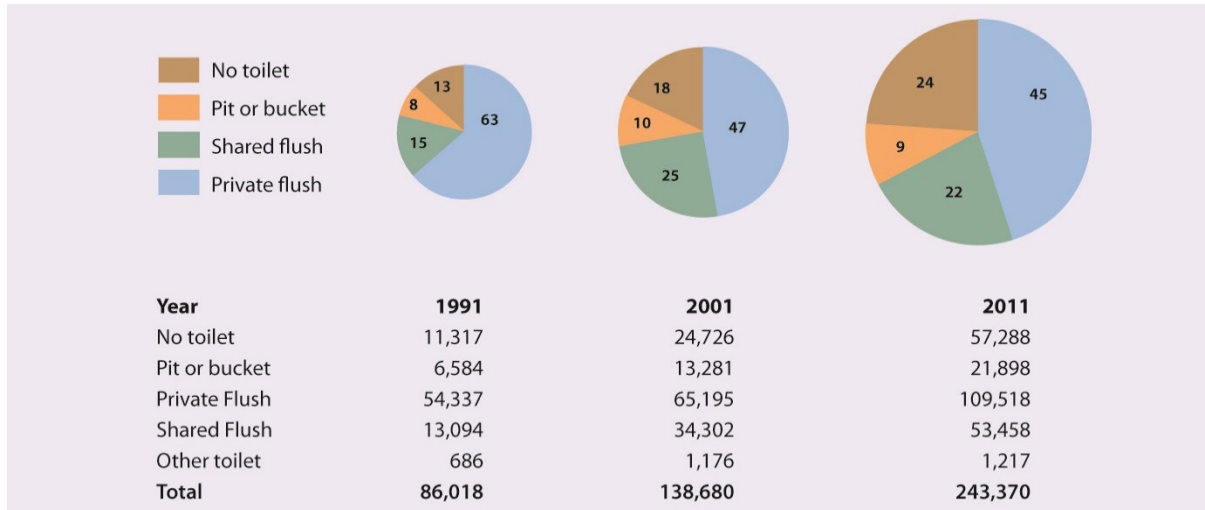
## Characteristics of informal settlements

### Services

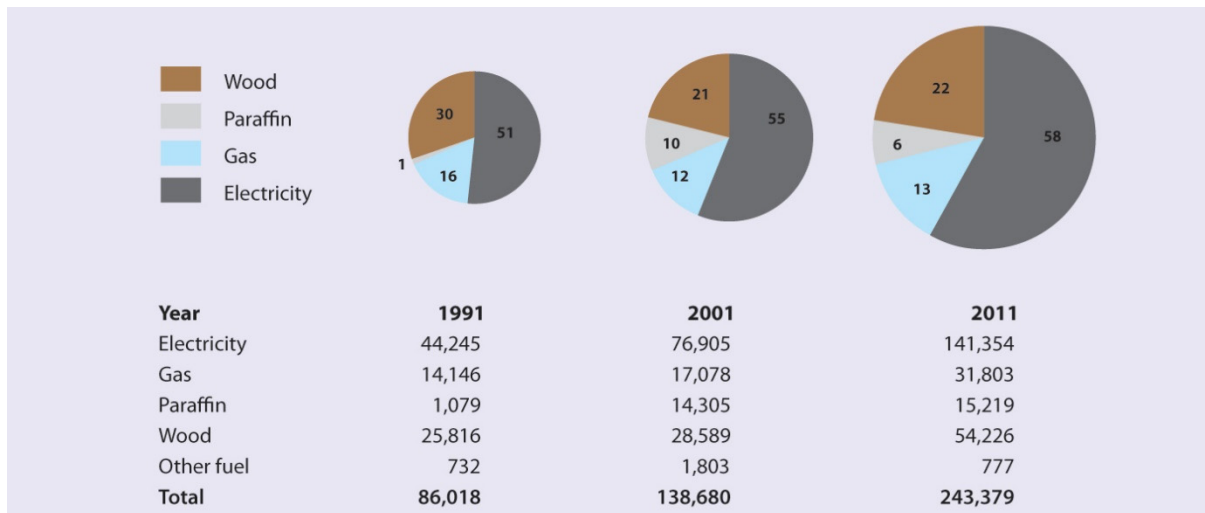
Many of Namibia's informal settlements are characterised by a lack of services. By 2011, for example, almost 60,000 urban households with more than 200,000 people had no access to any kind of toilet (Figure 6). Following a similar pattern, more than 54,000 urban households with more than 200,000 people relied on wood as their main cooking fuel (Figure 7).

### Housing materials

The great majority of houses in urban areas are constructed of either bricks or blocks (formal structures) or corrugated iron (informal structures). Corrugated iron is cheap. A simple shack can be erected within a day or so, and in cases where the shack has to be moved, the material can be re-used to erect a new structure in a different location. Local authorities also tolerate corrugated iron in informal settlements, where the use of permanent construction materials for houses is often prohibited. However, many shack residents have the means to build with bricks if they were allowed to do so. Recent research reveals the broad range of monthly incomes in informal settlements, which in certain areas of Windhoek were found to range between N\$300 and N\$35,000, and up to three-quarters of heads of household had some sort of formal employment (Seliger 2016). With these kinds of incomes most shack owners could invest in permanent housing structures, at least incrementally and over time. Residents of informal settlements in neighbouring countries with generally lower income levels than Namibia normally build much more with permanent building materials (in the case of Angola, see for example: DW & CEHS 2005). Similar conditions hold in certain Namibian towns where residents may build with bricks in some informal settlements, sometimes to such an extent that housing with permanent building materials is the dominant type of housing.



**Figure 6:** Pie diagrams of the percentages of homes having different types of toilets in all urban areas in 1991, 2001 and 2011, while the number of homes with different toilets are in the table.



**Figure 7:** Pie diagrams of the percentages, and a table of the numbers of all urban households using different types of cooking fuel in 1991, 2001 and 2011.

### Physical structure

Many informal settlements have irregular physical layouts that are not formally planned (Figure 8). Furthermore, most informal settlements are located at the peripheries of the formal towns, making commuting times costly and time-consuming. The lack of organised and planned physical structures creates various disadvantages for informal settlement residents, three important ones being:

1. Laying and providing services to unstructured informal settlements is difficult. Infrastructure such as water pipes, electricity grids and sewage systems are usually laid along roads, without which the installation of infrastructure becomes challenging, indeed often impossible.
2. Unstructured and unplanned settlements cannot be legally proclaimed under current planning legislation and policy, effectively condemning them to permanent informal status without tenure rights for their residents.
3. Once an unstructured informal settlement densifies, restructuring and upgrading becomes costly, since it usually involves resettling residents to provide space for the layout of roads and erven.

### Land tenure

In Namibia, residents with houses erected in unproclaimed settlements cannot have freehold title and do not have formal tenure security over the land on which their homes are built. This has many drawbacks since land tenure security is fundamentally important for:





**Figure 8:** Examples of unstructured, unplanned and disorganised informal residential areas in Windhoek's Havana informal settlement.

- Socio-economic household development in terms of generating wealth;
- Use as collateral for commercial loans;
- Protection against eviction;
- Regulating the transfer of rights, such as under conditions of sale and inheritance, and
- Protecting the rights of socially vulnerable people, such as women and the poor.

#### **Environmental Challenges**

Informal settlements create, or are associated with, various environmental problems, most of which stem from inadequate services, infrastructure and planned physical structure. Three challenges are of particular concern in Namibia:

- Removal of vegetation and degradation around informal settlements;
- Open solid waste and pollution of water resources;
- Flooding

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### **Assessing the challenge**

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#### **Urban immigration**

Why do so many Namibians move from their rural homes to live in informal urban housing? The major reason for most movements to towns is the quest for jobs and money (Indongo et al. 2013, Pendleton et al. 2014). Other reasons include rural poverty, family issues and education, with many households having multiple reasons for migrating.

There are important linkages between these factors. For example, people seeking higher education at urban universities are also attracted to the greater availability of jobs in urban areas. Graduates are therefore unlikely to return to rural areas. Likewise, economic conditions in towns are particularly attractive to people who live in places where there are few jobs or opportunities to start enterprises. This is the case in most rural areas of Namibia where farming is the dominant land use. Here it is hard to earn significant incomes from farm produce because environmental conditions – notably low soil fertility and shortages of rain – keep farm production very low, with little chance of having surpluses that could be made available for sale. Access to markets is also not easy in a large country with such a small population as Namibia. Additionally, many farmers prefer to keep any surpluses and livestock as security for the future.

For these reasons most cash in circulation in communal areas comes from pensions, orphans' grants, wages for teachers and other civil servants, remittances and retail trade. The majority of income therefore comes from sources that have nothing to do with farming or rural homes. Migrants are thus attracted by urban economic opportunities, but also spurred away by the poor economic prospects that prevail in rural areas. Since migrants have aspirations similar to people who have spent their entire lives in urban areas, planners may expect immigrants to stay in urban areas, have families and have long-term goals for their urban future.

In summary, rural life is tough in most parts of Namibia, especially for anyone wishing to earn a moderate income. There are exceptions: large commercial farms, big livestock holdings on communal land, irrigated smallholdings, lodges and tourist camps, mines, special plant products to harvest and sell (devil's claw for example) and trophy hunting. All these exceptions generate money, but there are too few enterprises to support substantial numbers of people.

### Land and housing markets

Calculations based on urban growth-rates between 2001 and 2011 suggest that in 2017 alone, some 11,500 formal and 12,500 informal houses were built. Less than 50% of the housing demand during this year was therefore met by the formal market. This is a clear indicator that the current land and housing market does not cater for the majority of Namibia's urban population, obliging more than 50% of urban residents to find access to land and housing in the informal market. Most of these residents are from the poor and lower middle-income segment of the population.

Housing in the informal market is not for free. The building or purchase of a shack may cost between N\$3,000 to N\$10,000, and rental prices range from N\$150 to N\$2,500 per month, depending on shack location, size and provided services such as water and electricity. Informal residents, therefore, do need considerable financial means to satisfy their housing needs. The problem is that the formal housing market does not provide solutions which are affordable for the lower income segment of the population.

Most government housing programmes, for example, the National Housing Programme (NHE) and the Mass Housing Development Programme (MHDP), have focused on the provision of 'low cost' houses that mostly cost N\$250,000 and more, a price that is above the means of a great percentage of informal settlement residents. The size and outputs of these programmes has further been too small to meet demand effectively.

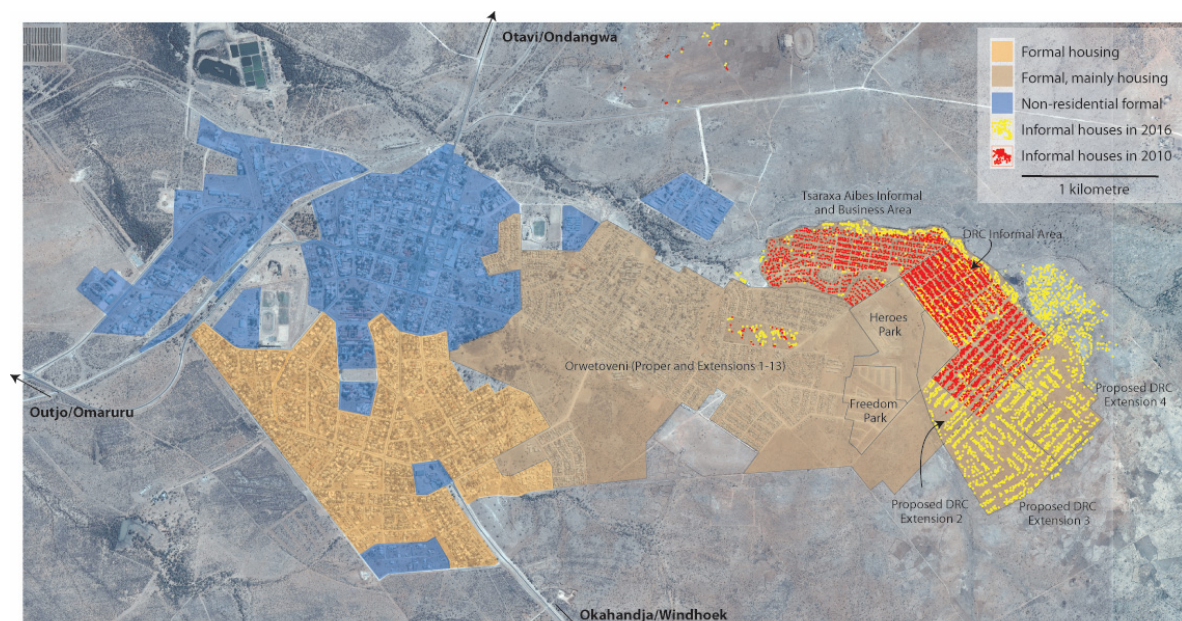
### Ways forward

#### Innovative approaches practiced by some local authorities

Over the last years, some local authorities have experimented with the provision of minimally- or non-serviced erven for low income residents in unproclaimed urban areas. While the details of the approaches differ from town to town, the underlying principles are:

1. Land is planned, surveyed and distributed to low income residents before township establishment;
2. Avoidance of lengthy and complex township establishment allows the local authority to provide low cost land immediately, according to demand;
3. As the new low income informal settlement is planned and surveyed, services can be provided incrementally and over time as funding becomes available, facilitated by the planned road layout;
4. As the land is allocated by local authorities and lies within a planned settlement that will not be changed in its physical structure, residents have 'de facto' tenure security, underlined by the fact that the local authority may allow the construction of houses with permanent building materials;
5. The township can be proclaimed at a later stage for residents to acquire official tenure security;
6. The construction of the houses is left to the residents themselves, to build at their own pace and according to the available financial means.

In towns such as Outapi and Otjiwarongo, this approach allowed control over most informal settlement growth to the



**Figure 9:** Satellite image of Otjiwarongo taken in 2016 showing the location of informal houses mapped in an earlier image from 2010 and then in 2016. The organised structure of most informal settlements in this town is clearly visible.



present date. While many of these settlements are still considered 'informal', they are in fact planned and surveyed settlements (Figures 9 & 10). Many such settlements in Outapi, for example, have been proclaimed in the meantime. Many others, such as in Otjiwarongo, resemble low income formal areas with different levels of services.

Similar approaches can be observed in other towns in Namibia, such as Ruacana, Rundu and Oshakati. However, while towns implement such projects on different scales, they rarely have the necessary dimension to achieve complete control over informal settlement growth. The main challenges faced by local authorities include the availability and technical capacity of their staff to implement such projects on a larger scale, as well as a lack of funds to ensure adequate town planning, surveying and consulting engineering services.

The provision of low cost land is the most effective measure to provide access to housing for low income immigrants and to turn informal into formal urban growth. As the approach is applied by many local authorities across Namibia in some or other way, it provides a basis that can be reinforced and scaled up to meet low cost land and housing demand.

#### ***Town planning, land surveying and servicing costs***

In order to create a properly planned and structured settlement, two basic services are needed: town planning and land surveying. The town planner (ideally in collaboration with local authorities and residents) develops the layout of the new settlement, while the land surveyor transforms the layout into reality on the ground by marking the erven, roads, and any other physical features of the layout, with pegs.

Of all investments that go into the development of a new settlement, town planning and land surveying cost least. Only 3-5% of the cost of a fully serviced erf is for town planning and land surveying costs, with the remaining costs being attributed to the installation of services such as water, sewerage, electricity and roads. In the case of a fully serviced erf costing N\$100,000, only N\$3,000-5,000 is therefore for town planning and land surveying.

Minimal services that ensure access and safe sanitary conditions, as for example communal water points and levelled roads, can be provided at equally low costs. Basic sanitation can initially be provided by giving residents an incentive to build pit latrines as a temporary solution until domestic water connections and a sewerage system are installed.

A minimally serviced erf, within a settlement that is planned and surveyed and can be upgraded with services over time,



**Figure 10:** Road layout in the DRC informal settlement in Otjiwarongo. In many areas, the Municipality allows the construction of houses with permanent building materials, given that the settlement has a planned structure. As a result, while residents may initially put up shacks for temporary housing, investments are soon made into permanent structures that provide much better living conditions and allow residents to invest into their properties.

can therefore be provided at approximately N\$10,000. This is an amount that is affordable by a large percentage of informal settlement residents. Table 1 provides a summary of the costs that constitute a planned, surveyed and minimally serviced erf of 300 square metres. The calculation of land surveying, town planning and servicing costs of a specific layout is a complex process, with costs depending on a variety of variables such as: size of erven, size of layout, topography, soil composition and the availability of bulk services. While costs may vary considerably according to these variables, benchmark numbers can nevertheless be established with the purpose of providing a basic notion of the costs involved in planning, surveying and servicing a new settlement.

Calculations have been done for a sample township with following specifications:

- Size of erven: 15 x 20 meters (300 square meters)
- Road width: 12 meters
- Bulk infrastructure: The calculations are based on the assumption that all bulk infrastructure exists and can be connected to. The costs therefore only reflect distribution within the township and connecting points to bulk at a distance of 100 meters from the township.
- Soil conditions: loose, without special obstacles for excavation (for example, rock)
- Topography: flat

**Table 1: Costs of a minimally serviced erf (N\$ per erf)**

<b>Land surveying</b>	
Topographical survey	278
Cadastral survey	1,478
<b>Town planning</b>	
Layout planning fees	1,000
Statutory Application	867
<b>Services</b>	
Communal water point reticulation system	1,667
5 communal water points (prepaid system)	2,106
Levelled roads	3,095
<b>Total N\$</b>	<b>10,491</b>

Additional costs will accrue as bulk infrastructure has to be upgraded and expanded. However, if the provision of erven is on a cost recovery basis, available local authority resources and government subsidies can be invested to ensure that bulk infrastructure expands as the town is growing.

#### **Costs of providing minimally serviced erven on a national scale**

A new settlement with 300 erven with costs per erf at N\$10,000 costs N\$3-million. The planning, surveying and servicing of a settlement of that size may take approximately 6 months. The proceeds of the sales of the erven can be kept in a revolving fund, in order to initiate the planning, surveying and servicing of the next settlement as soon as the erven of the previous settlement are sold.

With an initial fund of N\$3-million, a town can therefore initiate a financially sustainable process of providing low cost erven. Depending on the efficiency of developing the new settlements, 300-600 low cost erven can be provided per year. Many mid-sized towns in Namibia would cover their housing demand with this number of erven. Smaller towns would satisfy demand with a lower number, such as 50, 100 or 200 erven per year. Such smaller towns would therefore require initial funds of N\$500,000, N\$1-million or N\$2-million respectively.

Projections based on informal housing growth rates between 2001 and 2011 suggest that in 2017 alone, more than 12,000 new informal houses would be built, with the number increasing each year. This number is an indicator of the land and housing shortfall in the low income segment of the market, suggesting that at least the same number of low income residential land or housing units would have to be provided annually to transform informal into formal urban growth. At N\$10,000/erf, this adds up to a required minimal investment of N\$120-million or less than USD10-million, with which an effective and potentially financially sustainable programme for the provision of low cost land on the national level could be initiated. These costs are low if compared to certain government expenditures, such as N\$2,400-million that was budgeted for a new parliament (Shinovene 2016), the initially budgeted N\$7,000-million for the Hosea Kutako airport upgrade or the N\$700-million subsidy for Air Namibia in 2016 alone (Weylandt 2016).

Much of the investment could further be recovered by increasing the basis of payment of rates and taxes, which, in Namibia, are only applied in proclaimed, formal areas of towns. For example, if 30,000 of Windhoek's informal settlement households would pay N\$100 in rates and taxes per month, this would provide the City of Windhoek with an additional income of N\$3-million per month, N\$36-million per year or N\$360-million over the span of 10 years. While the upgrading of existing informal settlements may take time, this tax base could nevertheless be rapidly expanded by ensuring that the future growth of towns is taking place in planned and surveyed settlements that can be proclaimed, and that rates and taxes applied.

## **Conclusions**

Rural to urban immigration and associated urban growth is not a Namibian phenomenon, but a regional and international trend. The world is urbanising fast and Namibia is no exception. People migrate from rural to urban areas in search of formal and informal employment, as well as improving their access to health and educational facilities. As towns are the

motors of economic growth, this transformation from a rural to an urban society provides considerable social and economic opportunities for Namibia. If urban immigrants can be integrated into the towns' economies, they can become productive members and contribute towards generating wealth and development. Access to a piece of land where immigrants can establish their homes provides one of the very basic conditions for households to build secure investments, becoming an integral part of the formal town, and contribute to its economic base and public funds.

#### ***A general recommendation***

All possible proactive steps should be taken to avoid establishing settlements that are disorganised, unstructured and dense shanties of corrugated iron shacks. Conversely, steps taken towards creating ordered settlements where low income residents own their land, can build permanent homes and look forward to the incremental provision of services, should be encouraged. These steps require proactive planning of informal settlements before people settle there.

#### ***Recommendation 1: Focus on the provision of land, not housing***

To address the housing crisis of Namibia's low income urban residents, the focus should shift from the provision of housing towards the provision of affordable land. The construction of houses should be left to the residents, allowing them to build at their own pace, with a minimum of obstacles and a maximum of encouragement. Government and local authorities should supply land with a minimum of cost and at maximum speed.

#### ***Recommendation 2: Gain control over informal settlement expansion***

Gaining control over informal settlement growth should be a priority for any town in Namibia. This requires a supply of properly planned and affordable low cost land.

#### ***Recommendation 3: Support innovative, proactive and pragmatic approaches of local authorities***

Local authorities are the key actors that manage urban development in their areas of jurisdiction. They normally identify local challenges long before anyone else, and they are often the first to produce innovative and pragmatic solutions. These local initiatives and answers should be supported by government, NGOs and the private sector. Specific support should also be provided to help local authorities to:

- Manage the complex, and often lengthy township proclamation process;
- Manage projects to provide low cost land or upgrade existing informal settlements;
- Promote aspects of social inclusion, economic efficiency and environmental sustainability.

#### ***Recommendation 4: Accelerate the provision of tenure in structured or upgraded informal settlements***

Many informal settlements are ready to be proclaimed, having planned physical structures and demarcated erven, for instance. Without any additional funds the settlements could be proclaimed by removing administrative obstacles that stem from current policy. A national inventory of settlements that are ready for proclamation should be assembled, and their proclamation fast-tracked by the Ministry of Urban and Rural Development.

#### ***Recommendation 5: Attract private sector investment***

The provision of low cost urban land can be done on a cost recovery basis, therefore facilitating private sector involvement, while safeguarding real benefits for local authorities. Land can be supplied with minimal use of public funds by government, as is the case with the development of middle and upper income housing.

#### ***Recommendation 6: Attract international donor funding and Corporate Sector Responsibility (CSR) support for upgrading projects***

Many dense and unstructured informal settlements cannot be upgraded without significant resources. Scarce public funds should be assigned to such areas, and used to leverage additional funding from international donor organisations and CSR funds from the Namibian private sector.

#### ***Recommendation 7: Turn rapid urbanisation and the creation of new townships into an economic opportunity for Namibia***

The development of Namibia's rapidly growing towns should be guided by principles of social inclusion, economic efficiency and environmental sustainability. Planning provides opportunities to create new townships that are conducive to the economic and social needs of its residents, and the town and the nation. Migrants from impoverished rural areas need homes that provide them with confidence, services, security and long-term outlooks to be economically productive. The integration of low income residents into the formal land market will also raise public funds from rates and taxes for the betterment of all.

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## Can camera traps count game?

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URL: <http://www.nje.org.na/index.php/nje/article/view/volume1-stratford>

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### Abstract

Game counts provide essential information for the management of wildlife populations. On Ongava Game Reserve, two methods are used annually to count animals: a helicopter aerial count and a 72-hour waterhole count. During the extended waterhole count, observers remain in place for the duration of the count, and are required to record details of all animals seen. In this study we assessed whether camera traps could assist in the count process, firstly by easing the vigilance burden of observers in the nocturnal periods, and secondly by improving the accuracy of the overall count. We found that camera traps not only can substitute for human observers for nocturnal counts, they record more events and hence count more animals. We also found that traps record cryptic and small species that are often missed by observers. Ongava now uses camera traps as the primary counting method from midnight to dawn. In contrast, we found that camera traps cannot substitute for observers for diurnal observations when animal group sizes of more than ten animals are present. Even with extended trap arrays, the field of view and complexity of individual movements in large groups compromises the ability of offline counters to track animals, and they underestimate group sizes. However, there remain a number of advantages of running a camera census in parallel with observers – sightings can be checked and verified, and traps have a consistent mode of operation, minimising inter-observer bias. Images of marked animals can be reviewed offline for further analysis. We believe that the additional information recorded by camera traps provides important population data for both prey and predator species, allowing the development of an integrated ecosystem management strategy.

**Keywords:** camera trap, game count, Namibia, nocturnal animals, waterhole

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### Introduction

One of the essential tools for wildlife managers is the 'game count' and regular, repeatable animal counts provide an important foundation for population management (Bothma 2002). There are several options for undertaking counts, and the selection of the most appropriate depends on the habitat, terrain and species composition of the area in question (Bothma 2002). However, most count methods have associated errors or biases that need to be considered when interpreting the results (Elphick 2008). On Ongava Game Reserve, two annual game counts are undertaken. A new-moon helicopter count is followed by a full-moon 72-hour continuous waterhole count. The data from these counts are analysed across years to assess population trends, and are also compared to provide an estimate of population sizes.

Waterhole counts require observers to remain at the location and to record the details of all visits by animals to the waterhole. When these count periods are extended over multiple days, observer performance can be compromised, especially during nocturnal periods.

In recent years the use of automated camera traps has become ubiquitous in wildlife biology. Researchers are using camera traps to search for rare and endangered species, to compute occupancy, density and structure of extant populations, and even to monitor dynamic parameters such as behaviour and movement (O'Connell et al. 2011). While there is a significant core of literature that discusses the limitations of the use of camera traps in wildlife research (reviewed in Burton et al. 2015), there are few studies that assess the use of camera traps in census work. In this study we explore how camera traps might be used during game counts.

We tested camera traps in two 72-hour waterhole game count scenarios. First, we assessed whether traps would be a sufficient substitute for observers for nocturnal observations. Then we configured arrays of traps to record all animal movement during daylight hours, and compared off-line counts from the image records for selected waterholes with sightings recorded by observers.

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### Methods

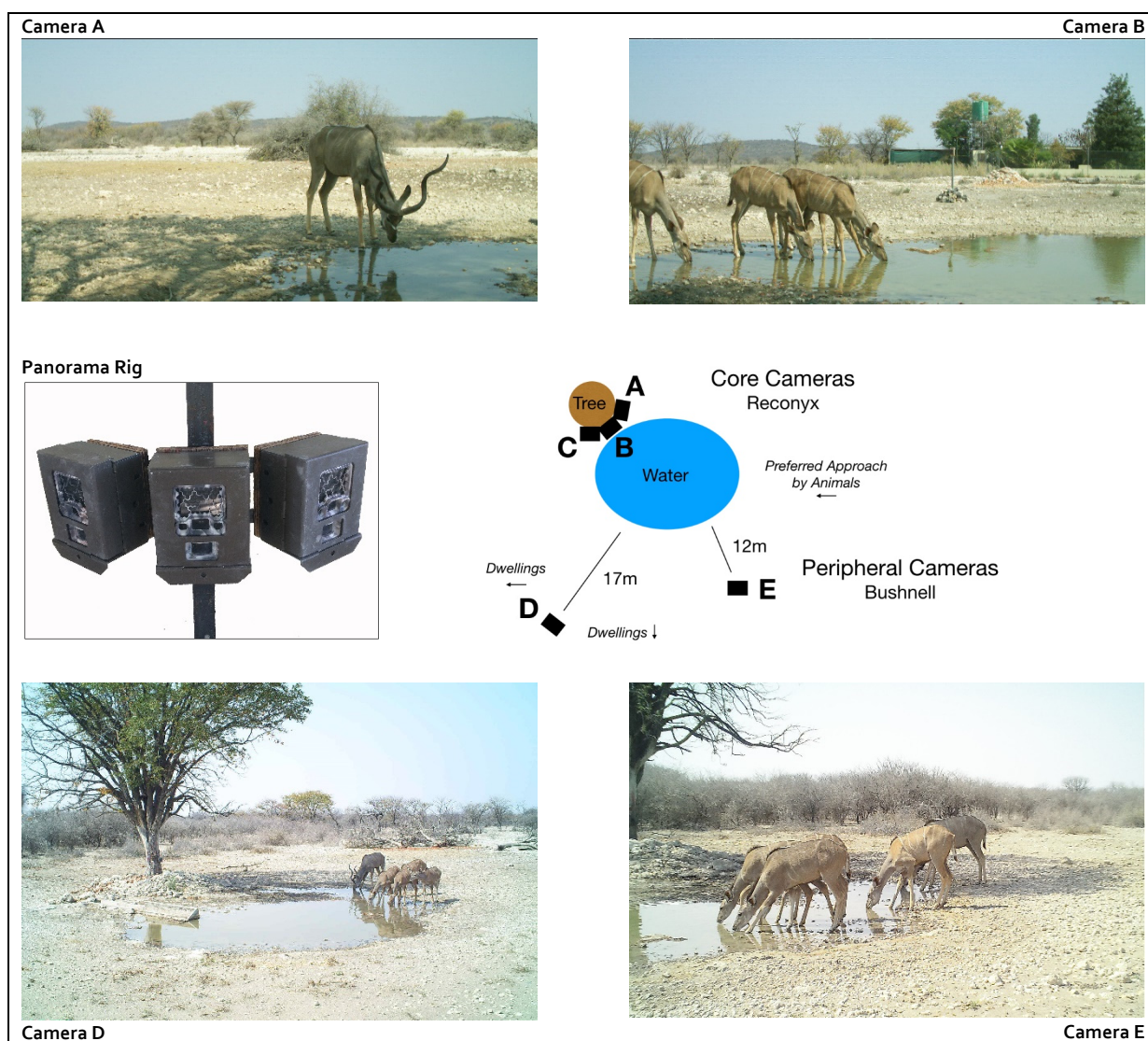
Ongava Game Reserve, a 300 km<sup>2</sup> private reserve bordering Etosha National Park in the north of Namibia, uses a conventional methodology for extended waterhole counts. Two or three people are deployed to each waterhole for the duration of the 72-hour period at full moon. For waterholes close to dwellings or with permanent hides, the observers remain in the same place for the duration of the count. At remote waterholes, hides are erected for diurnal observation, but the observers use vehicles to move closer to the waterhole for nocturnal observation. Observers record the date, time, species, total count and age/sex structure for each wildlife sighting.



For nocturnal trials, camera traps are mounted on nearby supports (typically trees) and are focused on the edges of each waterhole. In most cases, two cameras within 5 m of the waterhole are sufficient to provide complete coverage. When either the area of water is large, or there are no convenient local mounting points, we deploy up to three additional cameras on nearby trees.

For diurnal trials, we supplement the nocturnal placements to achieve a wider view. The trap models we use typically have a 45-degree field of vision, so we designed 'panorama rigs', configured to hold up to four cameras and calibrated to provide a wide view with a 5-degree visual field overlap between adjacent cameras (see Figure 1 Inset). In order to provide additional views, and where there is no convenient natural structure, we mount our rigs on poles (Figure 1). Poles with cement bases are placed close to typical approach trails. To allow animals to habituate to the equipment, we deploy these poles at least two weeks before counting trials, and then attach the rigs without cameras for the final week.

For all trials, we start recording data several days before the observer count. This allows us to confirm that our placements are accurate and that all cameras are configured correctly. Typically, we use Reconyx RC-500 traps for core monitoring at each waterhole, supplemented by Reconyx RC-55, Bushnell HD and Bushnell E-series traps for more peripheral monitoring. All these traps use infrared for nocturnal imaging. For the diurnal trials we also include some Bushnell Aggressor and Cuddeback C-1 traps. All traps are configured to record three or ten images per trigger with either a 1-



**Figure 1:** Examples of views from multiple cameras. Schematic shows the layout of cameras around a waterhole. A, B and C are core cameras used for nocturnal monitoring, D and E are peripheral cameras that are added for diurnal monitoring. These cameras are typically set back from the waterhole to provide a greater field of view. Images are of the same small group of five kudu (*Tragelaphus strepsiceros*) taken at the same time by each of the cameras. Note that the core cameras are placed at a distance that does not provide a field of vision that is wide enough to see the entire group. However, these cameras need to be that close to trigger for cryptic and smaller species. Offline analysis uses a synchronised multi-camera view to assist the operator in recording observations. Inset. An example of a 3-camera 'panorama rig', that provides a calibrated field of vision of 125 degrees with 5 degree overlaps between cameras.

second (core monitoring) or 3-second (peripheral monitoring) delay. The cameras are set to re-prime as fast as possible to ensure we record as many images as possible per observation.

## Results

### Nocturnal

In our first nocturnal trials in 2012 and 2013, up to 26 traps were deployed across 9 (2012) or 8 (2013) of 11 waterholes. When compared to the nocturnal records made by observers, the camera traps recorded additional animals at night (2012:  $n=301$ , 2013:  $n=264$ ). This increase in number of animals recorded was more pronounced for remote waterholes that are not lit ( $n=6$ ). In addition, camera traps recorded more nocturnal observations at waterholes that have associated wallows or secondary reservoirs. Camera traps were particularly effective at recording cryptic and small nocturnal species that were missed by observers (Figure 2). Across the trials, a range of mammalian herbivores was also observed drinking nocturnally (oryx (gemsbok), *Oryx gazella gazella*, springbok, *Antidorcas marsupialis*, Angolan giraffe, *Giraffa giraffa angolensis*, Hartmann's mountain zebra, *Equus zebra hartmannae*, common duiker, *Sylvicapra grimmia*).

On Ongava, typically one-tenth of all animals counted are recorded from nocturnal observations (e.g. 2016: 9.2%, 2017: 10.3%), with about 40% of these occurring in the period from midnight to dawn. After our initial nocturnal trials, management was able to sufficiently trust the data from camera traps to allow observers to stand down and rest, at least between midnight and dawn. These data make a small but significant contribution to the overall count - for example, in total 287 animals were counted from camera trap images in the three midnight to dawn periods across the 72-hour count in October 2017 (32% carnivores), constituting about 4% of the total 72-hour animal count.



**Figure 2:** Examples of cryptic and smaller nocturnal species recorded by camera traps. From left to right, top to bottom: aardwolf, *Proteles cristatus*; brown hyaena, *Parahyaena brunnea*; Cape fox, *Vulpes chama*; caracal, *Caracal caracal*; cheetah, *Acinonyx jubatus*; civet, *Civettictis civetta*; honey badger, *Mellivora capensis*; leopard, *Panthera pardus*; porcupine, *Hystrix africaeaustralis*; serval, *Leptailurus serval*; spotted hyaena, *Crocuta crocuta*; African wild cat, *Felis sylvestris lybica*.

## Diurnal

In 2016 we performed a comprehensive camera trapping exercise in parallel with the 72-hour count, deploying 63 camera traps across 12 waterholes (n=3-7 traps per waterhole, see Figure 1 for a typical configuration). For selected waterholes we performed an offline 'count', using images from the trap arrays. Table 1 shows the results for two of these waterholes, one remote, the other near dwellings and lit at night.

For each waterhole, our offline camera trap counts recorded more observations and counted more animals than the human observers. However, this effect was only pronounced during the nocturnal period, and had greater impact at the non-lit waterhole (see accuracy and number of observations missed). It is important to note that the camera traps also missed some observations at one of the waterholes. This arose due to a core camera failure, emphasising that consistent coverage of the field of view is critical for accurate counts.

The traps were able to replicate the diurnal count in most cases, however group size played an important role in determining trap performance. We found that for group sizes above about 14 animals (12-17 across species) offline counters were unable to track the movements of animals in and out of the field of vision of the camera traps, even when using panorama configured arrays. Their estimates were often less than half of those of the human observers. This effect was exacerbated when animals were disturbed during drinking, such that counters were not able to determine which animals were drinking for the first time, or revisiting. Human observers were much better at counting group sizes for larger groups.

To estimate the impact of this, in Table 2 we show the 2016 observer group size counts for the five herbivore species on Ongava that are typically found in groups or herds.

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Applying a conservative group resolution cut-off of ten animals indicates that camera traps would significantly underestimate population counts for these species in about 25% of all observations. Note that all of the larger group observations for these species were diurnal.

**Table 1:** Comparison of observer and camera trap counts during a 72-hour waterhole count. \*Consolidated count across all traps, most observations seen on multiple traps. \*\* Excludes midnight–dawn period when observers were not active, accuracy is the proportion of correct observations expressed as a percentage.

Period 10:00 14/9/16 to 10:00 17/9/16			
A. Remote Waterhole		Observers	Camera Traps
	#	2	5
Observations*	Total	30	77
	Diurnal	22	23
	Dusk-Midnight	8	19
	Midnight-Dawn	not active	35
	Missed**	16	4 (did not drink)
	Accuracy**	50%	88%
Animals*	Total	100	168
	Diurnal	87	82
	Dusk-Midnight	13	27
	Midnight-Dawn	not active	59
B. Lit Waterhole (near dwellings)		Observers	Camera Traps
	#	2	4
Observations*	Total	76	113
	Diurnal	43	50
	Dusk-Midnight	33	37
	Midnight-Dawn	not active	26
	Missed**	26	18
	Accuracy**	69%	89%
Animals*	Total	205	251
	Diurnal	157	173
	Dusk-Midnight	48	45
	Midnight-Dawn	not active	33

**Table 2:** Number of observations by group size of Black-faced Impala, *Aepyceros melampus petersi*; Blue Wildebeest, *Connochaetes taurinus*; Springbok, *Antidorcas marsupialis*; Waterbuck, *Kobus ellipsiprymnus*; Kudu, *Tragelaphus strepsiceros*.

Species	Group Size					
	1-9	10-19	20-29	30-39	40-49	>50
Black-faced Impala	153	32	14	6	4	5
Blue Wildebeest	5	3	0	1	2	1
Springbok	51	8	2	0	0	0
Waterbuck	41	2	1	0	0	0
Kudu	106	11	1	0	0	0

## Discussion

In this extended study of waterhole counts, we show that camera traps not only can substitute for human observers for nocturnal counts, in fact they record more events and hence count more animals. We also note that traps record cryptic and small species that are often missed by observers. Others have also found camera traps to perform better than more conventional counting methods (e.g. spoor counts, Dupuis-Desormeaux et al. 2016). The management decision to use

camera traps as the primary counting method from midnight to dawn relieves the observers and their subsequent vigilance improves. This methodology is now implemented routinely for most waterholes on Ongava.

In contrast, we find that camera traps cannot substitute for observers for diurnal observations, at least not when animal group sizes of more than ten animals are expected. One reason for this is that camera traps typically trigger in response to movement a maximum of 10-15 m from the trap (closer for smaller animals), hence the first images of larger groups are too close to the cameras and extend beyond the field of view. Conversely, if the camera is placed too far from the waterhole, movements will not trigger the camera. We are currently working on methods to activate the cameras using an external synchronised trigger.

However, there remain a number of advantages of running a camera census in parallel with observers – for example, counts can be checked and verified. Camera traps have a consistent mode of operation, and therefore inter-observer reliability is less of an issue, especially across waterholes (Kaufman & Rosenthal 2009). In addition, images of marked animals (either natural marks, such as spots and stripes, or artificial marks, such as tags) can be reviewed offline. This is important from two points of view. Firstly, we use mark-recapture methods to analyse carnivore populations, and, secondly, identification of marked animals allows us to estimate drinking intervals for each of the herbivore species. We typically run the camera trap census for several days before and after the 72-hour period, allowing us to gather additional information about species counts, and also to assess whether the presence of human observers influences the count statistics. (The effect is nowhere near as significant as the disruption in herbivore drinking patterns created by a pride of lions residing at a waterhole for 72-hours!)

There is a significant overhead to processing the data that come from large camera trap surveys (Scotson et al. 2017), in addition to the substantial initial purchase costs. In the 2016 extended study described above, the 63 camera traps generated more than 1,000,000 images in a 14-day period that incorporated the 72-hour count period. While we have developed in-house software to assist with this processing, an experienced operator will still take 4-5 hours to perform the offline count for just the 72-hour period from a typical waterhole. Researchers are currently developing methodologies and software to standardise data collection, storage and analysis (e.g. Niedballa et al. 2016).

We report that about 10% of all animals recorded are counted at night, and typically 30% of those species are carnivores. Given that the focus of animal census work tends to be on the larger herbivore species, is this part of the count important in the context of managing populations? We believe that it is. In order to manage animals in fenced reserves it is important to have information about both prey and predator species so that the management policies can aim to balance the ecosystem for the primary herbivores and carnivores.

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