



EXPEDITION REPORT

Expedition dates: 1 – 27 October 2017

Report published: July 2020

**Carnivores of the Cape Floral Kingdom:
Surveying Cape leopards, caracals and
other species in the fynbos mountains
of South Africa**





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Abstract

The fynbos biome of South Africa is a biodiversity hotspot renowned for its very high plant species richness, endemic birds and the presence of the Cape mountain leopard, a small race of African leopard *Panthera pardus ssp pardus*. Biodiversity monitoring across a range of faunal and floral taxa was conducted in mountain fynbos habitat. Studies were conducted for two weeks in October 2017, with the help of international citizen scientists recruited by Biosphere Expeditions and based at Blue Hill Nature Reserve (Western Cape).

In this report we provide a status update on the leopards of Blue Hill Nature Reserve for the period 2016-2020 through photo records. Camera-trapping confirmed the presence of male leopard 'Strider', resident since 2015 until early 2019. It also captured a female, suspected of being accompanied by a nearly fully-grown cub. A new male and female have been recorded since mid 2019 (Sugarbird Valley pair). Caracal *Caracal caracal* and African wildcat *Felis lybica* continue to be present, as do most other potential prey species (e.g. klipspringer *Oreotragus oreotragus*, Cape hare *Lepus capensis*). An extended mammal monitoring activity was trialled (mammal mapping).

We also examine the abundance of small mammals in relation to a fire event in the fynbos. Trapping of small mammals using Sherman traps was undertaken, with data compared between 2016 and 2017. Capture rates were highest in the unburnt sections of fynbos.

Honeybush *Cyclopia intermedia* is an important wild harvested plant for the tea industry that is found on Blue Hill Nature Reserve. We set up monitoring plots for 400 plants, which were measured, health status checked and flower buds counted. We examined various correlates of potential yield as baseline monitoring for further studies on this important economic crop.

Opsomming

Die fynbos bioom van Suid Afrika is 'n biodiversiteit 'kern area' bekend vir baie hoë plant spesie diversiteit, endemiese voëls, en die teenwoordigheid van die Kaapse berg luiperd (*Panthera pardus*). Die monitering van biodiversiteit oor 'n reeks van fauna taksa was uitgevoer in berg-fynbos habitat. Studies was uitgevoer oor twee weke in Oktober 2017, met behulp vanaf internasionale burgerlike wetenskaplikes, gewerf deur die 'Biosphere Expeditions' en gebasseer by Blue Hill Natuur Reserwaat (Wes-Kaap).

In hierdie verslag verskaf ons 'n meer onlangse status van die luiperde in en rondom Blue Hill Natuur Reserwaat vir die periode van 2016 tot 2020 deur gebruik van foto rekords. Met die gebruik van lokval kameras kon ons die aanwesigheid van 'n mannetjie, genaamd 'Strider', vasstel vanaf 2015 tot vroeg 2019. 'n Wyfie, wat vermoedelik deur 'n redelik volgroeide welpie vergesel was, was ook afgeneem. 'n Nuwe mannetjie en wyfie was aangeteken sedert middel 2019 (Sugarbird Valley paar). Rooikat *Caracal caracal* en vaalboskat *Felis lybica* is steeds teenwoordig, en so ook ander potensiële prooi spesies (bv. Klipspringer *Oreotragus oreotragus*, vlakhaas *Lepus capensis*). 'n Uitgebreide soogdier moniteringsaktiviteit was ook bewerkstellig (soogdier kartering).

Ons ondersoek ook die getalle van klein soogdier spesies in verband met 'n vuur wat plaasgevind het in die fynbos. Klein soogdiere was gevang deur die gebruik van Sherman lokvalle, en data was vergelyk tussen 2016 en 2017. Vangsyfers was die hoogste in die ongebrande gedeeltes van die fynbos.

Heuningbos (*Cyclopia intermedia*), wat 'n belangrike wild geoeste plant vir die tee industrie is, word gevind op Blue Hill Natuur Reserwaat. Moniteringsplotte vir 400 plante was opgestel. Afmetings was geneem, gesondheidstatus geskat, en blomknoppe getel. Ons het verskeie korrelate van potensiële produksie ondersoek as basislyn monitoring vir verdere studies op hierdie ekonomies belangrike gewas.

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1. Expedition review

M. Hammer (editor)
Biosphere Expeditions

1.1. Background

The Cape Floral Kingdom is one of the world's biodiversity hotspots and as such is a UNESCO World Heritage Site. It is dominated by a fire-driven ecosystem – the [fynbos](#) biome, which has unsurpassed botanical richness: 7,000 of the 9,000 plant species that are found here are endemic. It is in the flower-filled Cape Fold Mountains of South Africa that the Cape mountain leopard (a race of African leopard *Panthera pardus ssp pardus*) is found – a leopard half the size of the leopards found in the savannahs of Africa, but with home ranges twice the size. In 2019, the International Union for Conservation of Nature (IUCN) classified leopards as Vulnerable (Stein et al. 2020) due to hunting, habitat loss and fragmentation. Indeed, they are becoming increasingly rare outside protected areas. From 175 to 588 leopards are predicted to occur in the Western Cape province of South Africa (Devens et al. 2020) and a similar number in the Eastern Cape (Devens et al 2018), the major population stronghold for this type of leopard.

Study area

The [fynbos biome](#) (fynbos, also known as the Cape Floral Kingdom/Cape Floristic Region) comprises one of only six floral kingdoms in the world and is contained entirely within the political boundaries of South Africa, where it is mostly restricted to the Western and Eastern Cape provinces of the [Cape Fold Belt](#). Owing to its exceptional plant species richness and high level of endemism, as well as high levels of animal diversity and endemism, it is recognised as one of the world's 25 biodiversity 'hotspots' (Myers et al. 2000). The biome takes its name from 'fynbos' (Afrikaans for 'fine bush'), the dominant vegetation type in the biome. The other two main vegetation types are 'renosterveld' (Afrikaans for 'rhino bush') and 'strandveld' (Afrikaans for 'beach bush'). Vegetation is dominated by three characteristic families: Proteaceae, Ericaceae and Restionaceae. All of the fynbos experiences at least some winter rainfall. In the western fynbos rainfall is mostly restricted to the winter months, while in the east summer rainfall becomes more important. Conversion to agriculture, urbanisation, climate change and the invasion of a variety of alien plant types all pose major conservation threats to the area (Rebelo 1992).

Small mammal trapping took place at the [Blue Hill Nature Reserve](#) (BHNR) on the western border of the Baviaanskloof Nature Reserve, Western Cape, South Africa. The 2230 hectare BHNR lies between 1,000 and 1,530 metres above sea level (m asl) in fynbos. The reserve falls into an aseasonal rainfall region, with annual informal records from the closest town, Uniondale (40 km distant, 730 m asl), for the 1965 – 1997 period of 344 ± 102 mm (Lee and Barnard 2013). The property that forms the reserve was acquired by the Lee Family Trust in 2009 and they initiated a partnership with [CapeNature](#), the Western Cape provincial agency in charge of protected areas, to convert the land to a stewardship nature reserve. This process was completed in 2013 with the land now managed for biodiversity. Prior to 2009, use of the land had been for agricultural purposes, mostly stock farming of cattle and sheep.

More background information, location conditions and the research area are as per [Lee et al. \(2016\)](#)

The expedition focused on biodiversity monitoring of the fynbos and karoo habitat types occupied by two of Africa's iconic cats: the threatened Cape mountain leopard and the caracal *Caracal caracal*. The surveys feed into efforts by the Eastern Cape Parks and Tourism Agency to mitigate conflict with landowners, by working with them to highlight conservation issues and thereby contribute significantly to cat survival and conservation. In this report we focus on which leopards have been seen, and the density of a potential prey item: the steenbok *Raphicerus campestris*. The expedition also conducted a survey of harvesting impacts on the honeybush tea plant *Cyclopia intermedia*.

The ultimate goal of research activities at the study site is to feed into ongoing research projects in and around the fynbos and its wildlife. The project is interested in developing a monitoring technique that will better inform landowners of the status of their wildlife and predatory cats, identify potential conflict areas, and use the knowledge gained to mitigate conflicts. To this end, the project's overall aim is to develop monitoring techniques that will enable landowners to determine predator and prey densities on their land. The project also wishes to contribute to the biodiversity monitoring of the area through contribution to citizen science projects (specifically [iNaturalist](#)). To achieve all this, the expedition monitored the density, abundance, spatial distribution, home range size and habitat preferences of a population of wildlife on a nature reserve using transect, mammal mapping and camera trap techniques.

Almost all of Africa experiences some sort of human impact and the fynbos is no exception. Much of the wildlife roams on understaffed, underfinanced and remote mountainous nature reserves where monitoring is difficult; or on private farmland where landowners have mixed attitudes to perceived problem animals such as leopard, caracal, jackal *Canis mesomelas*, baboon *Papio ursinus* and bushpig *Potamochoerus larvatus*. The Cape mountain leopard is one of South Africa's '[Threatened Or Protected Species \(TOPS\)](#)', which restricts legal hunting, but the laws are almost impossible to enforce. [There is a strong farming and hunting lobby](#) pushing for greater control of 'pest' species and [anecdotal evidence suggests that lethal control](#) by legal and illegal methods is widespread across the country.

Wherever humans and wildlife come together, conflicts tend to appear, and human-wildlife conflict has been identified as one of the biggest threats to biodiversity worldwide. Sound scientific knowledge is key to mitigating this conflict and to making wise management decisions that balance the need of humans, wildlife and the environment. Biodiversity monitoring on the Blue Hill Nature Reserve not just informs management on the reserve, but also allows interfacing with landowners on sensitive issues (such as wildlife conflict) through engaging with them via more neutral biodiversity monitoring topics (such as bird and vegetation monitoring).

1.2. Dates & team

The project ran over a period of one month divided into two 13-day slots, each composed of a team of national and international citizen scientists, a professional scientist and an expedition leader. Group dates were as shown in the team list below. Dates were chosen to coincide with spring and the period associated with the mildest climate, best plant flowering, mammal and bird breeding, as well as fewest mosquitoes.

The expedition team was recruited by Biosphere Expeditions and consisted of a mixture of ages, nationalities and backgrounds. They were (in alphabetical order and with country of residence):

1 – 13 October 2017: Judith Bird (Canada), Peter Bird (Canada), Neil Goodall (UK), Libby Kerr (Australia), Sandra Kerr (Australia), Karin Kindbom (Sweden), John Munthe (Sweden).

15 – 27 October 2017: Elena Boulet (Germany), Stephen East (UK), Gopalakrishna Jaishankar (USA), Valerie Mills (USA), Tobias Rudolph (Germany), Shobhana Srikant (USA), Brigitte Ziegler (Germany).

Dr Alan Lee, the expedition's field scientist, grew up in South Africa. He graduated from the University of Witwatersrand with an Honours Bachelor's Degree in Botany and Zoology in 1996. While working and travelling from London he obtained a Diploma in Computing in 2001. He then commenced a period of seven years in Peru, first working for a volunteer project investigating impacts of tourism on Amazonian wildlife, and then from 2005 to 2010 undertaking a Ph.D. on the parrots of the Peruvian Amazon. Biosphere Expeditions part-financed and contributed data to the Ph.D. resulting in three peer-reviewed publications. In 2011 Alan set up the Blue Hill Escape guest establishment on the Blue Hill Nature Reserve with his wife, Anja, and parents Chris and Elaine Lee. In 2012 he was accepted as a postdoctoral research fellow at the FitzPatrick Institute of African Ornithology at the University of Cape Town to undertake an assessment of the status of the endemic birds of the fynbos, and maintains a position there currently as Research Associate. From 2017 – 2018 he worked for BirdLife South Africa in conjunction with the South African National Biodiversity Institute (SANBI) Karoo BioGaps project, which undertook a comprehensive biodiversity study of the arid Karoo biome of South Africa. He is Editor-in-chief of Ostrich: Journal of African Ornithology. He is currently home-schooling his two children in the era of Covid-19.

Craig Turner, the expedition leader, was born in Oxford, England. He studied biology, ecology and environmental management at Southampton, Aberdeen and London universities. Soon after graduating from his first degree, he left the UK for expedition life in Tanzania. Since then, he has continued to combine his interest in travel and passion for conservation, working with a wide range of organisations on projects and expedition sites in the Americas, Africa, Asia and the Pacific. He has managed expedition grant programmes for the Zoological Society of London, and is a frequent contributor to the 'Explore' conference held by the Royal Geographical Society (RGS). He is a Fellow of the RGS and the Linnean Society. Having visited and/or worked in more countries than years have passed, he now runs a small environmental consultancy with his partner, based in Scotland, where he splits his wildlife interests and work between the UK and overseas. He is ever keen to share his exploits, writing for several magazines, and is a published photographer.

All medical umbrella, safety and evacuation procedures were in place. There was one medical incident (laceration), which required stitching in the local hospital in Uniondale, after which the patient could continue with the expedition.

1.3. Partners

Leopard monitoring was conducted in partnership with the [Eastern Capes Parks and Tourism Agency](#). Monitoring of Karoo birds and mammals was conducted in conjunction with [BirdLife South Africa](#) – South Africa’s leading bird conservation organisation. These surveys were also conducted in partnership with [SANBI’s Karoo BioGaps project](#).

1.4. Acknowledgements

We are grateful to the citizen scientists, who not only dedicated their spare time to helping but also, through their expedition contributions, funded the research. Brian Reeves was helpful in assisting with permits and project design to partner the monitoring at Blue Hill Nature Reserve with the Eastern Cape Parks and Tourism Agency, responsible for monitoring the neighbouring Baviaanskloof Nature Reserve, as well as supplying catalogues of leopard records from that protected area. Thanks to Dale Wright, Daniel Marnewick and Mark Anderson for supporting the Karoo surveys through funding and provision of a field vehicle. Expedition cooks Melda and Gurli also provided valuable botanical insights through their affiliations with SANBI’s Custodians of Rare and Endangered Wildflowers (CREW) programme, submitting botanical records to the iSpot citizen science project (which has since migrated to iNaturalist). Biosphere Expeditions would also like to thank members of the Friends of Biosphere Expeditions and donors for their sponsorship. Thank you to Chris, Elaine and Anja Lee for being such excellent hosts and making us feel at home at the expedition base. Thanks to Kougaview Game Farm for the entertaining game drive. Thanks to two reviewers for their helpful comments on drafts of this report. Thanks to Stefan Goets for translation of the abstract into Afrikaans.

1.5. Further information & enquiries

More background information on Biosphere Expeditions in general and on this expedition in particular, including pictures, diary excerpts and a copy of this report, can be found on the Biosphere Expeditions website www.biosphere-expeditions.org.

1.6. Expedition budget

Each team member paid a contribution of £1,840 per person per two-week slot towards expedition costs. The contribution covered accommodation and meals, supervision and induction, special research equipment and all transport from and to the team assembly point. It did not cover excess luggage charges, travel insurance, personal expenses such as telephone bills, souvenirs etc., or visa and other travel expenses to and from the assembly point (e.g. international flights). Details on how this contribution was spent are given below.

Income	£
Expedition contributions	25,810
 Expenditure	
Expedition base includes all board & lodging	2,372
Transport includes transfers, car hire, fuel	1,459
Equipment and hardware includes research materials & gear etc. purchased in South Africa & elsewhere	766
Staff includes local and Biosphere Expeditions staff salaries and travel expenses	7,044
Administration includes miscellaneous fees & sundries	201
Team recruitment South Africa as estimated % of annual PR costs for Biosphere Expeditions	6,773
 Income – Expenditure	 7,195
 Total percentage spent directly on project	 72%

1.7. Literature cited

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2. Leopard monitoring at Blue Hill Nature Reserve 2017 – 2020: the changing of the guard

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During 2017-2020 leopard and medium to large mammals continued to be monitored at Blue Hill Nature Reserve using motion sensor camera traps. This brief chapter serves as an update of activity since [Lee et al. \(2017\)](#) and presents key photographs. We focus on activities from 2017, based on the year Biosphere Expeditions groups were last involved at Blue Hill Nature Reserve, and use the opportunity to update the catalogue of leopards *Panthera pardus* recorded on the Nature Reserve. No efforts were made to live capture leopards during this study period and there are no plans to do so in the future.

Most camera trap data have yet to be entered into a database and we thus report only on highlight photos and key species for this period. With no Biosphere Expeditions groups for 2018 and 2019, the absence of volunteers able to assist with this task has been distinctly felt, underscoring the value of these expeditions for this project.

The male leopard named 'Strider' (Figure 2a), first identified in the study site in 2015, continued to be recorded regularly throughout 2017 (almost monthly), but was last seen on camera at the start of 2019. He was also absent for many months at a time, presumably visiting the vast majority of his territory that lies outside of the boundaries of Blue Hill Nature Reserve. His activities were captured on camera mostly at night, with one exception in the late afternoon. His presence was recorded at all camera trap locations (East Road, South Road, close to Blue Hill Escape, but most regularly at the Baviaans gate bordering the Baviaanskloof Nature Reserve). Given his absence on cameras for over a year and the appearance of a new male leopard in 2019, his fate is likely death. However, given his reign of five years and a lack of any reports of leopard persecution in the area, it is encouraging that his disappearance or death was likely a natural event, rather than associated with human-wildlife conflict.

On 11 July 2017, an unidentified male leopard was photographed in Sugarbird Valley hiking trail of Blue Hill Nature Reserve during the day. This male did not match any previously captured on camera (Figure 2b). However, given this was his only appearance at the height of Strider's reign, it is likely he was ousted from the area: he was never photographed again.

During 2017, the scruffy female 'Trampalina' was identified on only a few occasions, but possibly also as recently as May 2018. We suspect that she used the roads where cameras are placed less frequently than the male. Her presumed cub, Lina, was observed on one occasion, and a photo from November shows two leopards together, but identity could not be confirmed (Figure 2c). Not all camera trap photos of leopards allowed identification to individual level because of blurred markings etc. However, there have been no confirmed records since 2018 of either of these leopards. We suspect Blue Hill Nature Reserve was marginal to her core territory, and that she has also 'moved on'.

During 2019, two new leopards were recorded at Blue Hill Nature Reserve. Given that most records are associated with an area of the reserve known as Sugarbird Valley, this pair, a slender female, and a hefty male, have been named the Sugarbird male and female (Figures 2d and 2e). Records at several camera trap positions during late 2019 and early 2020 suggest that Blue Hill Nature Reserve is now part of the core territory of the large male. It is pertinent to remind readers here that, given the large ranges of leopards in the fynbos (20,000 – 60,000 ha), the majority of a leopard's territory in this area will fall across a mix of land use types and that the 20 km² Blue Hill Nature Reserve thus forms only 3-5% of a leopard's territory.

In April 2018 a knowledge exchange meeting was held with Eastern Capes and Parks Tourism Agency (ECPTA) representatives Brian Reeves and Sandile Mdoko, in charge of ecological monitoring of the leopards of the Baviaanskloof Nature Reserve. Monitoring of leopards at the study site complements those of the ECPTA. Photos of the leopards of Blue Hill Nature Reserve were provided to ECPTA, who were able to compare the leopards to the leopards in their database, obtained from camera traps in the eastern sections of the Baviaanskloof Nature Reserve. Interestingly, none of the current leopards that use Blue Hill Nature Reserve as part of their territory were present in the ECPTA database.

As part of their monitoring at 20 camera trap stations, Brian Reeves reported that only 14 of the 40 cameras worked for the full duration of their deployment (ECPTA unpublished report). Six of the cameras did not work at all and 10 of them worked less than half the time they were deployed. Two of the cameras (placed near Smitskraal) were stolen.

ECPTA monitoring takes place only in the 'core' Baviaanskloof Nature Reserve (70 km east of Blue Hill Nature Reserve), and as such permission was obtained for this study to complement the ECPTA study by placing cameras in the Welbedacht, Nuwekloof and Kromkloof western sections of that nature reserve.

Two of these camera locations were retrieved in March 2018, with the Kromkloof location collected in late 2018. However, the old Cuddeback Capture cameras in the Nuwekloof failed, recording images for less than a week, with only one male Cape grysbok *Raphicerus melanotis* recorded. The Cuddeback Attack camera location fared slightly better, however, one camera was destroyed by ants, which made a nest inside the camera. The opposing camera at this location recorded no leopard, but did record baboon *Papio ursinus*, kudu *Tragelaphus strepsiceros* and klipspringer *Oreotragus oreotragus*. The Kromkloof cameras also recorded no leopard, and few wildlife species (baboon, klipspringer, rock hyrax *Procavia capensis*) as most photos were of moving vegetation, which quickly drained the batteries.

At Blue Hill Nature Reserve, caracal *Caracal caracal* were recorded on only two occasions during 2017 (Figure 2f), although intermittently over the following years (2-5 photos per year). There was one report of caracal captured and killed on a farm in the district, although since no formal reporting system exists in this conservancy area, there are likely to be additional unreported incidents. Nonetheless, given the persistence of the species, the current offtake does not appear to be resulting in any local extinction. A lack of distinguishing features on these cats makes individual identification a matter of speculation.

We have previously speculated that increased leopard activity, as recorded for 2017, may result in depressed numbers of competing predators. At least anecdotally this indeed appears to be the case here for the caracal and wildcat (highest caracal recording rate was in 2014, which corresponded with the lowest leopard activity), but on the other hand a record number of black-backed jackal photos were recorded during 2017 (Figure 2j). However, these were all recorded only from camera trap locations on the East Road after a bushfire event (Figure 2j) along this road in April 2017, which could have resulted in habitat changes suitable for black-backed jackal, as well as provided scavenging opportunities on dead or injured animals. It is notable that of the previous four black-backed jackal events recorded since 2010 in or around Blue Hill Nature Reserve, two of these were in locations that had recently experienced bushfire events (South Road 2012, Naartjieskloofberg 2016). The only exception was a black-backed jackal entering Blue Hill Nature Reserve from a neighbouring farm and leaving again after about 10 minutes.

For other species, African wildcat *Felis lybica* was recorded in at least four photos (Figure 2g) during 2017, and in similar numbers from camera trap locations across Blue Hill Nature Reserve. Klipspringer (Figure 2h) and Cape grysbok *Raphicerus melanotis* continued to be regularly-recorded small antelope species, while mountain reedbuck *Redunca fulvorufula* have probably replaced grey duiker *Sylvicapra grimmia* as the most frequently recorded antelope species, with numbers definitely higher than when camera trapping activity initiated in 2010. The similar-sized and similar in appearance grey rhebok *Pelea capreolus* appears to have maintained similar levels, as have baboon and bushpig *Potamochoerus larvatus* (Figure 2i-n). Thus, available medium sized prey species for the leopard appear to be stable.

The camera traps also recorded interesting bird species (Figures 2n, 2o), with blue crane *Anthropoides paradiseus* recorded on the ground (rather than flying overhead) for the first time. A selection of other wildlife from the camera traps at Blue Hill Nature Reserve can be seen at the recently created (2020) [Blue Hill Nature Reserve project on iNaturalist](#), where a variety of biodiversity is recorded. Past expedition members are encouraged to upload georeferenced photos at that site. Additional photos can also be seen at www.facebook.com/bluehillnaturereserve/.



Figure 2a. The male leopard Strider continued to be a constant presence throughout 2017. Usually active at night, here we also show the only picture of Strider captured during the day, captured just prior to the expedition dates at Blue Hill Nature Reserve.



Figure 2b. Strider near the campsite in December 2017 (top), and an unidentified young male leopard (below)



Bushnell

11-28-2016 23:35:08



Figure 2c. Female leopard activity: a pair likely to be a female with older cub (above) and 'Lina' at the camera trap location closest to Blue Hill Escape (below). Last seen 2018.



Figure 2d. Sugarbird Valley female, left flank (top) and right flank (bottom)



Figure 2e. Sugarbird Valley male leopard: left flank (top) and right flank (bottom).



Figure 2f. Caracal: the numbers of photos records varies from year to year, low in 2017 and up in 2018



Figure 2g. African wildcat.



Figure 2h. Klipspringer: here featuring what appears to be a leucistic individual (top) and a regular male (bottom). Leucistic animals show partial loss of pigmentation, which causes a white, pale, or patchy colouration.



Figure 2i. Grey rhebok female (top) and male (bottom).



Figure 2j. A scene from the April 2017 bushfire (top) and one of several black-backed jackal photos from the burnt area (bottom).



Figure 2k. Spot the citizen scientists: Biosphere Expeditions team 2 servicing a camera trap.



Figure 21. Baboon continue to be the most frequently recorded mammals at Blue Hill Nature Reserve.



Figure 2m. Camera disruptions due to baboon activity continue to occur, but we tolerate this because they are also cute (when young, bottom).



Figure 2n. Bushpig (top) and blue crane (below).

This is the only record we have to date of blue crane, South Africa's national bird, from a camera trap at Blue Hill Nature Reserve. The site is from the East Road following a fire event that year.



Bushnell

10-07-2016 12:11:24



Figure 2o. Other novel bird species were also recorded for the first time on camera: Sentinel rock-thrush (top) and Cape crow (bottom), with the latter featuring in several photo events also from the burnt East Road areas.

3. Mammal mapping at Blue Hill Nature Reserve

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3.1. Introduction

Camera trapping can provide useful information on species richness across a range of medium- to large-bodied taxonomic groups, including birds and mammals. Camera trap studies are especially useful for the monitoring of cryptic, shy, rare and nocturnal species. However, the static nature of cameras in the landscape means that it is hard to build up a complete spatial picture of landscape use without a very large number of cameras (currently not available to this study). In addition, most camera trap locations for this study are along routes known to be used by leopard, creating a spatial sampling bias to these locations. Previously we have also used flush transects to determine relative abundance of selected prey species ([Lee et al. 2017](#)). However, again, these were on predetermined routes. As can be seen in this report, expedition citizen scientists participated in a wide variety of monitoring programmes, including small mammal, bat, bird and plant monitoring, which took place over a wide area of the study site. As such, in 2017, to augment mammal sightings and to determine spatial distributions of select (larger) species at Blue Hill Nature Reserve, we implemented the 'Mammal Mapping' activity to maximise the spatial coverage of reporting associated with the various monitoring activities.

3.2. Materials and methods

During October 2017, Biosphere Expeditions citizen scientists were asked to record all mammal species that they saw, including the date, time, and number of individuals. To determine the location of the animals, the GPS position of the observer and compass bearing towards the animals were noted, and then the distance to the centre of the group of animals at their initial location was measured using a laser rangefinder, or the distance was estimated. The position of the animals was then calculated using trigonometry using these equations in Excel:

$$\text{Longitude: Longitude of Observer} + (\text{Distance} * \text{COS}(\text{RADIANS}(\text{Bearing}))) * \text{SIN}(\text{RADIANS}(\text{Bearing})) / (40075017 * \text{COS}(\text{RADIANS}(\text{Longitude of Observer}))/360)$$

$$\text{Latitude: Latitude of Observer} + (-(\text{Distance} * \text{COS}(\text{RADIANS}(\text{Bearing}))) * \text{COS}(\text{RADIANS}(\text{Bearing}))) / (40007860)$$

Group sightings were independent events wherever possible: assigned to the first person in a group that spotted the mammals. Likely resightings on the same day were discouraged, but repeat encounters were allowed on different days.

3.3. Results

72 mammal sightings were recorded, with group 1 providing 41 sightings, and group 2 contributing 31 sightings. Ten species were recorded, of which the most frequently encountered groups were klipspringer, grysbok, rock hyrax and baboon, although baboon dominated the numbers of individuals observed with 55, followed by klipspringer (51) and grey rhebok (47). Numbers of groups and individuals are shown in Table 3.3a. The area covered is best seen in Figure 3.3a, which includes the locations of all 11 species. The locations of the smaller antelope observed is depicted in Figure 3.3b, while mountain reedbuck and grey rhebok are shown in Figure 3.3c.

Table 3.3a. Mammal species observed at Blue Hill Nature Reserve during mammal mapping encounters conducted by Biosphere Expeditions during October 2017.

Species	Count of groups	Sum of individuals
Chacma baboon <i>Papio ursinus</i>	8	55
Cape grysbok <i>Raphicerus melanotis</i>	8	14
Cape grey mongoose <i>Galerella pulverulenta</i>	2	2
Grey duiker <i>Sylvicapra grimmia</i>	1	1
Grey rhebok <i>Pelea capreolus</i>	7	47
Hewitt's red rock hare <i>Pronolagus saundersiae</i>	1	1
Klipspringer <i>Oreotragus oreotragus</i>	23	51
Mountain reedbuck <i>Redunca fulvorufula</i>	1	3
Rock hyrax <i>Procavia capensis</i>	15	36
Scrub hare <i>Lepus saxatilis</i>	6	6
Grand total	72	216

3.4. Discussion

This exercise highlighted the abundance of klipspringer at Blue Hill Nature Reserve. Klipspringer are mostly diurnal, small-bodied antelope occurring in pairs or small family groups, generally no larger than four individuals ([Dunbar and Dunbar 1980](#)). Cape grysbok was commonly recorded in the vicinity of the lodging at Blue Hill Escape due to the presence of a young habituated female (killed by baboon in 2018). Grey duiker was rarely encountered, which correlates with lower numbers of this species currently recorded on camera traps. Cape grysbok is predominantly nocturnal, as is grey duiker, which would also lead to lower sightings by citizen scientists since activities were predominantly conducted during daylight hours.

Unlike the klipspringer, which was encountered throughout the reserve, the grey rhebok and mountain reedbuck were encountered mostly in the old agricultural areas (along the south road, and towards the end of the east road). Most other species were encountered too infrequently to make descriptions of their spatial distribution at this stage, since this is the first time this activity was undertaken.

Conclusions and future research

This activity was easy and productive, and provided interesting preliminary results. It will be continued with future expeditions. Future expeditions should aim to fill gaps in area coverage (see Figure 3.4a). Data can be collected straight into smartphones using CyberTracker software. It is recommended that future participants attempt to install this software prior to arrival at the study site.

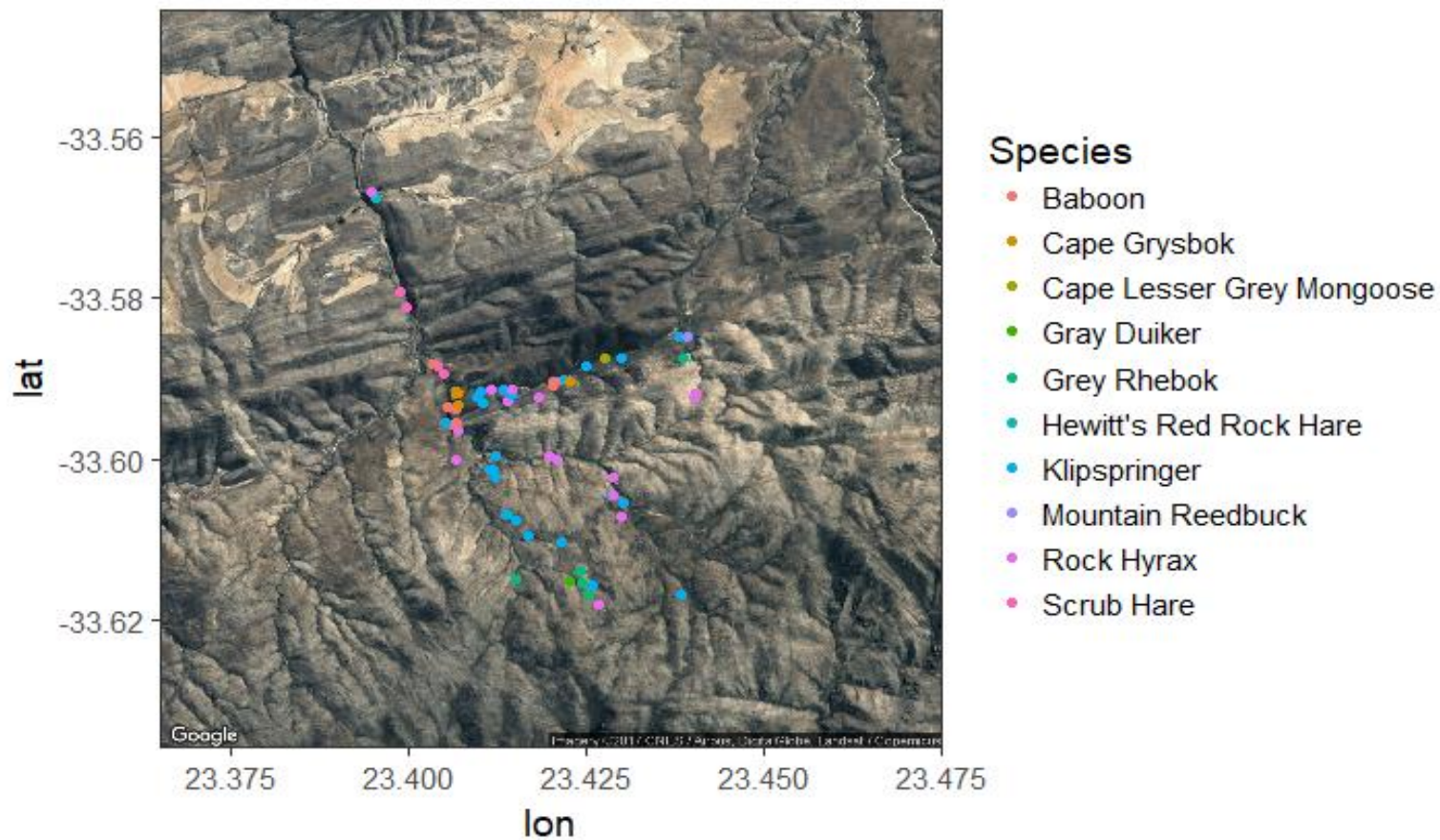


Figure 3.4a. Locations of all species registered at Blue Hill Nature Reserve during mammal mapping, October 2017. This figure best indicates areas that were covered during activities.

Small antelope at Blue Hill Nature Reserve

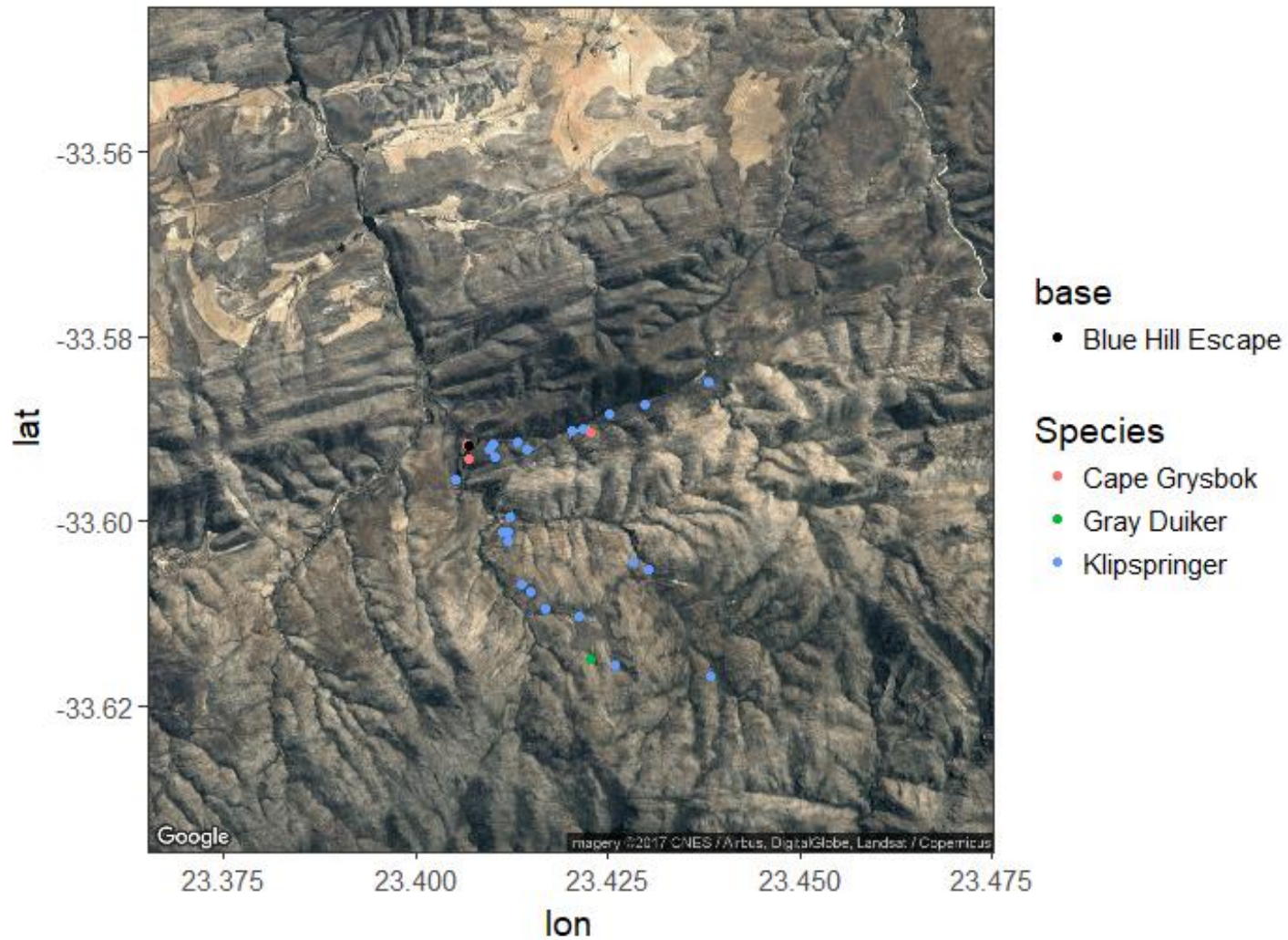


Figure 3.4b. The locations of small antelope species registered at Blue Hill Nature Reserve during mammal mapping, October 2017.

Larger antelope at Blue Hill Nature Reserve

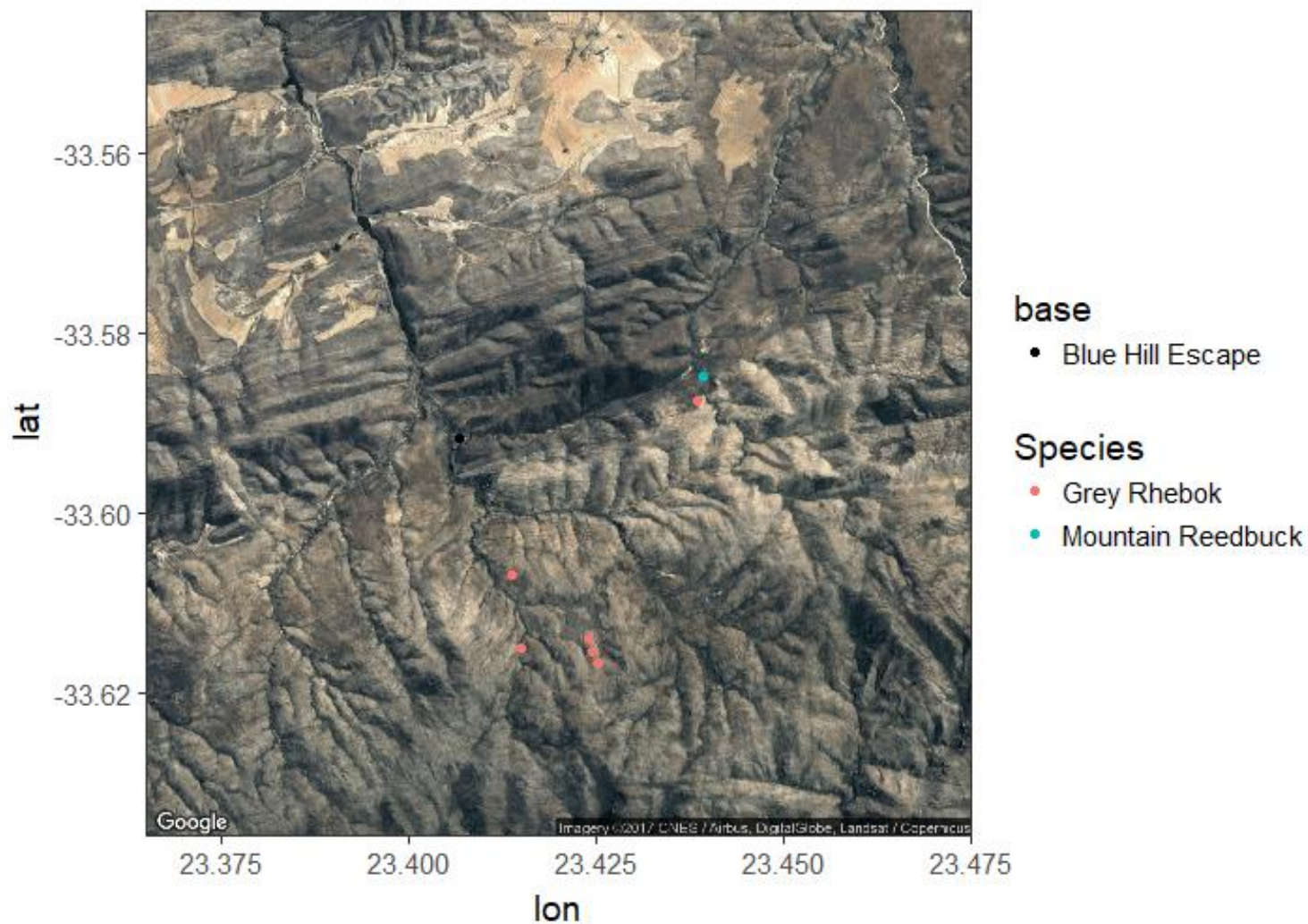


Figure 3.4c. The locations of medium-large antelope species registered at Blue Hill Nature Reserve during mammal mapping, October 2017.

4. Impact of a fire event on small mammal capture rates at a mountain fynbos site

Alan Tristram Kenneth Lee
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4.1. Introduction

Rodents play an important role in fynbos plant ecology, ranging from flower pollinators to seed predators and seed dispersers (Biccard and Midgley 2009, Midgley et al. 2002, Vlok 1995). They are also an important food source for a variety of animals, from snakes, to raptors, to cats (Byrom et al. 2014). A previous study of rodent diversity and capture rates from the Baviaanskloof, South Africa (Bond et al. 1980) concluded that species composition varied according to altitude and aspect with a total catch of only six rodent and two shrew species, and found correlations between abundance of rodent species and vegetation structure and the cover of rock and bare soil. Likewise, in Sherman trap captures we found an increased probability of small mammal capture as a function of decreasing altitude at Blue Hill Nature Reserve (Lee 2016b).

However, the fynbos is a fire-driven ecosystem, with fire return intervals for the most part between 8 and 15 years, depending on local temperature and rainfall patterns (Cowling et al. 1997). As such, most of the plants in the fynbos have life cycles that are adapted to fire: some plants flourish shortly after fire events, such as the geophytes, while others maintain seed banks on the plant, which are only released during a fire. Likewise, bird species show preferences related to the fire cycle, with some birds preferring young fynbos and others reliant on the resources available only in mature, older fynbos (e.g. the Cape Sugarbird is reliant on the flowers of protea species, which only flower several years after a fire (Lee and Barnard 2015)). With respect to medium-sized mammals, the flush of grasses that occurs after a fire attracts grazing species, which will struggle to find suitable food in older fynbos. Conversely, small generalist species such as grey duiker may rely on the cover provided by mature fynbos to escape detection from predators. However, how small mammals (rodents, shrews etc) respond to fire cycles in the fynbos has yet to be explored.

In this study, we were provided with the opportunity to explore small mammal capture rates in relation to a fire event after previous monitoring locations (Lee 2016a) were burnt during April 2017. Two sites were completely burnt, while one site was partially burnt. Here we compare the probability of small mammal capture using insights gained from previous monitoring.

4.2. Material and methods

Small mammal abundance from Sherman traps

In order to sample small mammal abundance, we created three Sherman trapping arrays at BHNr based on guidelines by Manley et al. (2006, Chapter 5: Small Mammal Monitoring). Traps were baited with peanut butter and a commercial mixed bird food or bread. Traps were placed in sheltered locations near rocks or under bushes and checked twice a day (morning and evening).

Trapped animals were photographed, if necessary, for identification, weighed and then released. Trapping did not take place or was abandoned in adverse weather conditions (too hot or cold).

Sampling was conducted from 8 – 14 October 2015, 8 – 14 October 2016, and between 3 and 21 October 2017 (15 nights). On selected nights, 50 traps were set along one of three routes, each of which was split between opposing slopes of a valley with aspect north or south. Traps were spaced at approx. 25 metre intervals. During 2016, sampling was conducted over three nights and then moved to the next route for a further three nights. During 2016 and 2017 routes were only surveyed for one night each. Routes were positioned along an altitudinal gradient at each array. In 2015 the fynbos was aged at approximately 16 years since the last fire.

On 7 April 2017 a large, natural fire burnt two of the routes completely, and roughly half of the third (see Figures 4.2a & b). Thus, sampling in October for burnt locations occurred approximately 6.5 months after this event.

Due to low capture rates within species, we pooled all captures to examine the influence of aspect, altitude and burnt status on probability of small mammal capture rate. We used binomial logistic modelling for capture vs non-capture from each trap check, keeping the individual trap location and sampling year as random effects. We used the lme4 package (Bates et al. 2015) and obtained p values using the lmerTest package (Kuznetsova et al. 2017) in R. We included only results of overnight trapping (n = 700), excluding the mostly unsuccessful daytime efforts, as this resulted in fewer warnings when running models due to zero inflation. Our full model was:

$$\text{rodentpa} \sim \text{scale}(\text{Altitude}) : \text{Aspect} * \text{Burnt} + (1|\text{Year}) + (1|\text{ID}), \text{family}=\text{binomial}$$

We then used the 'dredge' function from MuMIn package (Barton 2011) to select the best subset models by AIC (Akaike's Information Criteria). We present the two resulting models within 4 AIC of the top model. To illustrate the effect of burnt vs unburnt treatments, we calculated least mean squares using the 'lsmeans' function from the lsmeans package (Lenth 2016).



Figure 4.2a. A photo of the capture site during the fire event of April 2017. The mature fynbos (>15 years since last fire) dominated by protea species can be seen in the foreground.



Figure 4.2b. A photo of the capture site after the fire event of April 2017.

4.3. Results

Small mammal abundance from Sherman traps

Capture success along the Sherman trap arrays was low: 8% (108 captures of 1350 checks across 150 trap positions), with capture rate significantly higher in the morning checks after traps had been out overnight (13.8% compared to afternoon checks 1.2%; $t = -6.2$, $p < 0.01$), suggesting strong nocturnal behaviour among this community of small mammals.

Most frequently captured were: Namaqua rock mouse *Aethomys namaquensis* (50); Cape elephant shrew *Elephantulus edwardii* (21); striped field mouse *Rhabdomys pumilio* ($n = 14$); and African pygmy mouse *Mus minutoides* (10) (Table 4.3a). However, there was low confidence in the identification of some species, which most closely resembled Namaqua rock mouse and could have been Cape spiny mouse *Acomys subspinosus* (Table 4.3a). It is possible that species identified as Namaqua rock mouse also included other species.

Table 4.3a. Rodent species capture totals, with mean and standard deviation (SD) of mass in grams.

Species	n	Mean	SD
Namaqua rock mouse <i>Aethomys namaquensis</i>	50	51.7	7.26
Cape elephant shrew <i>Elephantulus edwardii</i>	21	53.4	8.38
Striped field mouse <i>Rhabdomys pumilio</i>	14	39.7	13.29
African pygmy mouse <i>Mus minutoides</i>	10	19.4	3.32
Unidentified	9	41.31	15.93
Fat mouse <i>Steatomys krebsii</i>	2	36.5	3.54
Climbing mouse <i>Dendromus melanotis</i>	1	13.2	
Vlei rat <i>Otomys irroratus</i>	1	89.9	

The best model explaining capture rate as a function of altitude, aspect and burnt status included only burnt status, retaining year and trap location as random effects (Table 4.3b), with capture rates highest at unburnt trap locations (Figure 4.3a, Table 4.3c). The second-best model included burnt status as the only significant predictor, but also included the interaction between altitude and aspect as non-significant variables. Capture rate decreased with increasing altitude in unburnt fynbos, but the opposite pattern was observed for burnt sites, as illustrated in Figure 4.3b. The probability of capture rates may differ slightly between slopes.

Table 4.3b. Model output of variables influencing the probability of finding a small mammal in a Sherman trap at Blue Hill Nature Reserve. Two model results are presented, the 'best' by AIC criteria includes only the burnt status of the location, while the second-best model includes all parameters, but only burnt status as significant effect.

	Estimate	Std. Error	z value	Pr(> z)
Best model AIC 521.3				
(Intercept)	-3.642	0.560	-6.504	0.000
Unburnt	1.859	0.398	4.662	0.000
Second best model AIC 524.4				
(Intercept)	-3.683	0.608	-6.058	0.000
Unburnt	1.917	0.518	3.702	0.000
Aspect Flat : Altitude	0.394	0.488	0.806	0.42
Aspect North : Altitude	-0.183	0.28	-0.653	0.514
Aspect South : Altitude	0.275	0.273	1.008	0.314

Table 4.3c. Summary table of captures per year, standardised by capture effort to indicate overall capture success: although capture totals were highest for 2017, this was mostly due to captures only on the unburnt route.

Year	2015	2016	2017
Captures	21	27	58
Captures per night	0.04	0.14	0.09

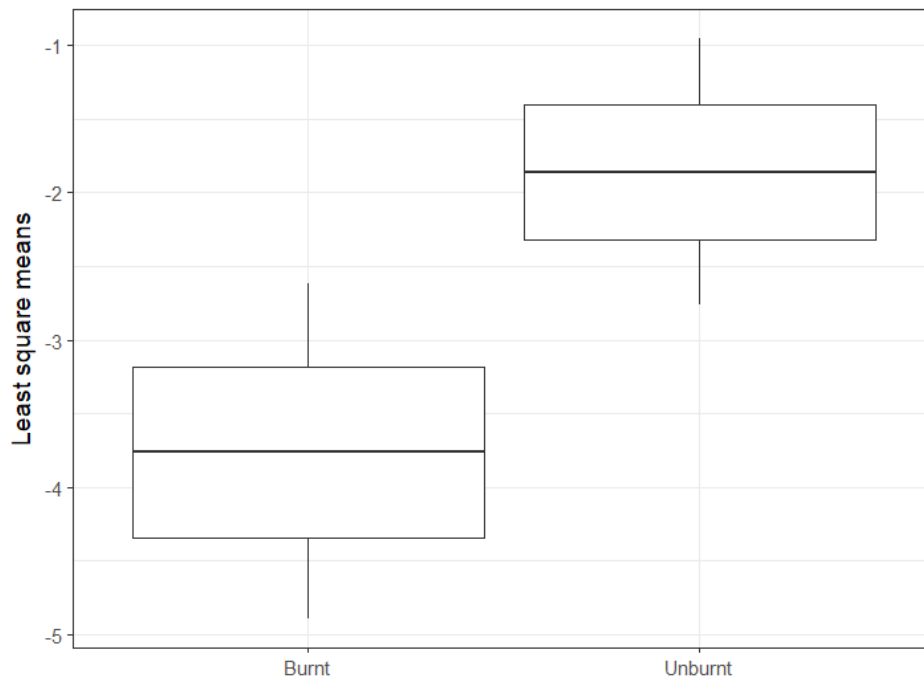


Figure 4.3a. Small mammal capture probability at Blue Hill Nature Reserve over three years suggested that probability of capture was lowest at burnt sites during 2017, as illustrated here with this box and whiskers plot (with median the middle line, upper and lower quartile, with the ends of the 'whiskers' the minimum and maximum) .

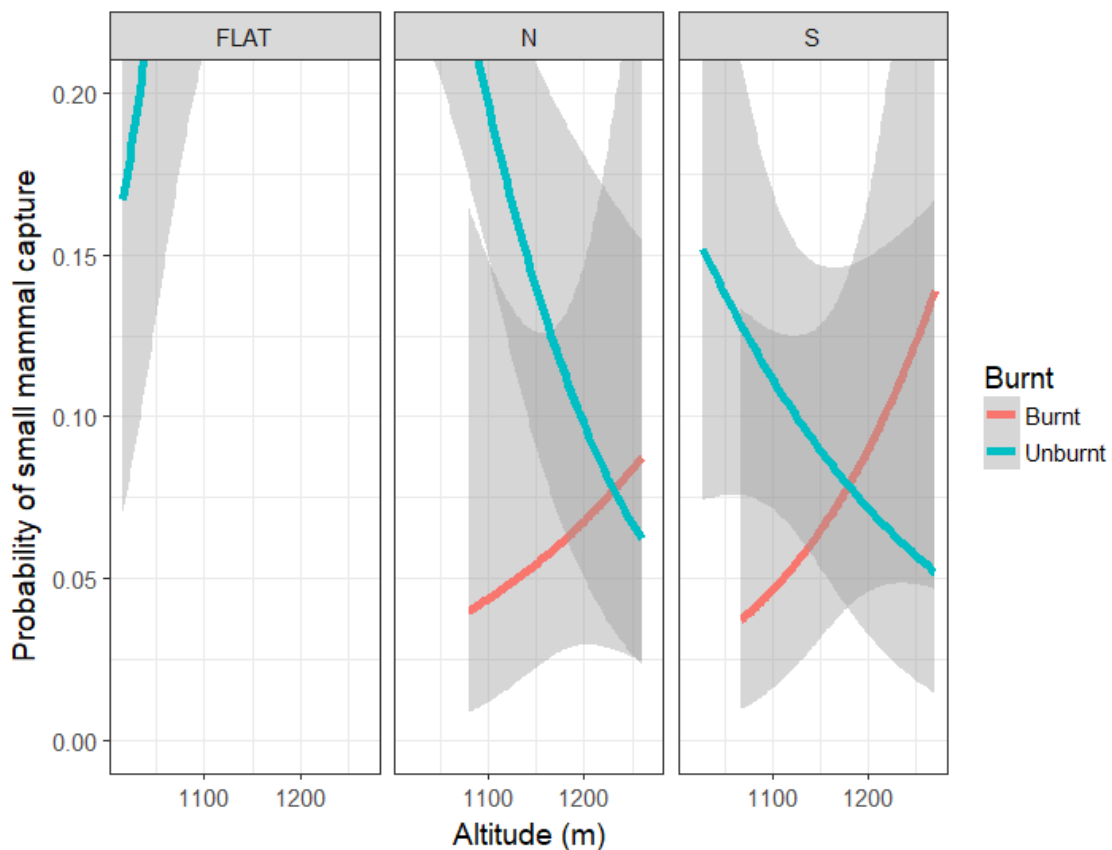


Figure 4.3b. An illustration of the model explaining small mammal capture probability at Blue Hill Nature Reserve, illustrating the influence of altitude & the interaction between aspect (N = North, S = South) and burnt status. Note capture probabilities decrease with altitude for unburnt sites, but this trend reverses for burnt sites. The reason for this is not clear at this stage. Lines indicate logistic regression slopes, with grey shading represents 95% confidence intervals.

In addition to the above ecological conclusions, the 2017 expedition also added two new species to the small mammal list: grey climbing mouse *Dendromus melanotis* (Figure 4.3c) and fat mouse *Steatomys krebsii* (Figure 4.3d).



Figure 4.3c. Grey climbing mouse *Dendromus melanotis* captured for the first time at Blue Hill Nature Reserve at one of the burnt sites during 2017.



Figure 4.3d. Fat mouse *Steatomys krebsii* captured for the first time at Blue Hill Nature Reserve at one of the burnt sites during 2017.

4.4. Discussion

Small mammal abundance from Sherman traps

To the best of our knowledge, this is the first study that has investigated the impact of a fire event on the probability of capture of small mammals in the fynbos. Capture patterns are weighted by Namaqua rock mouse, the most frequently captured species. Of the variables considered, if the site had been burnt or not was the most important variable in determining probability of capture, with higher capture rates in unburnt fynbos compared to sites that had been burnt six months earlier. To a degree this result was unexpected: large numbers of seeds are released from mature fynbos after fire events, theoretically resulting in a temporary super-abundance of seed food types. In more nomadic species, this abundance of food has been used to explain high abundance of the seed-eating protea seedeater *Crithagra leucoptera* after a fire event (Lee and Barnard 2014). However, the time since fire may have been too short to have resulted in population level responses, since the survey period occurred just after the winter period when at least some of the study species are likely to be reproductively inactive (Jackson and Bernard 1999). Thus, it is feasible that the lower capture rates were a consequence of direct fire-related mortalities prior to any population recovery. Certainly, very high capture rates were attained during 2017 in the remaining unburnt fynbos.

Previously, for the 2015/2016 capture period, altitude had been found to be a significant predictor of capture, with decreasing probability of capture with increasing altitude. We show here now that this trend is confounded by burn history and applies only to the unburnt fynbos. There was a non-significant trend for a reversal of this pattern in recently-burnt fynbos. The reasons for this are not immediately clear, as environmental parameters would rather suggest the opposite trend: vegetation productivity follows the moisture gradient, with taller plants in drainage lines, which would provide food, nesting and shelter resources for rodents in unburnt fynbos. Conversely, the loss of this vegetation would expose small mammals to predators, which in certain species prefer the use of drainage lines (Mann 2014) and thus lower altitude in our study context .

While there was no clear difference between north- and south-facing slopes, which have radically different vegetation at this site, sites with no quantifiable aspect (flat) had the highest capture rates. Unburnt south-facing slopes at Blue Hill Nature Reserve at the trap locations were dominated by *Protea* species, with north-facing slopes dominated by *Aspalathus*, *Restio*, and *Aloe* species (Lee and Barnard 2013). *Protea* are serotinous and retain their seeds on the plant until a fire, and such slopes would be expected to have lower food availability for this granivore community; while the plant community on north-facing slopes are dominated by plants that release their seeds on an annual basis.

Conclusions and future research

Future expeditions should attempt to repeat surveys at previously sampled trap sites to explore temporal changes in capture rate. More attention needs to be paid to the variables at each trap position, including percentage rock cover and dominant plant types, following previous research (Bond et al. 1980). In addition, in order to make this research novel, extending the sampling sites to include fynbos of different ages (time since fire) should be explored. Understanding how vegetation and topography influence rodent abundance and biomass will substantially aid the understanding in the predicted movements of their predators in the landscape.

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5. Steenbok: density estimates and predictors of occurrence from the arid southern Karoo, South Africa.

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5.1. Introduction

The steenbok *Raphicerus campestris* is a small (9 – 13 kg) antelope that has the second widest distribution of all the African antelope because of its ability to survive in areas where larger, faster and more aggressive ungulates have disappeared due to human pressure (Furstenburg 2005). With a large population and large range, the species is listed as Least Concern by the IUCN Red List (IUCN SSC Antelope Specialist Group 2016). In South Africa, the steenbok is widely distributed, but absent from coastal forests and thickets. The preferred habitats are undulating plains with scattered trees and shrubs and medium vegetation height (Furstenburg 2005).

According to Furstenburg (2005) steenbok can occur at a density of one individual per 6 – 14 ha in optimal habitats, 17 – 33 ha for marginal habitats, or >35 ha for poor habitat conditions. Density estimates of 2.68 – 5.35 ind/km² have been obtained for a game farm in the eastern Karoo (Stenkewitz et al. 2010). Mean home range of five free ranging steenbok in the Kruger National Park was small at 0.62 km² (Du Toit 1993). To the best of our knowledge, there are no published density estimates from outside protected areas.

The steenbok has been described as a selective feeder, utilising a wide variety of forbs and the leaves of trees and shrubs, but very few grass species (Huntley 1972). It has also been speculated that food requirements may contribute to low population density. The species is not reliant on standing water and can thus persist in arid environments (Cloete and Kok 1986).

The Karoo *sensu lato* describes the arid south-western zone of South Africa and consists of two botanically distinct biomes: the Succulent Karoo, with predominantly winter rainfall, and Nama Karoo with predominantly summer rainfall (Dean and Milton 1999). While in previous decades threats were mostly restricted to issues related to livestock management (Dean 1995), the Karoo now faces multiple infrastructure and developmental threats. There are environmental changes and biodiversity losses associated with the new role of arid ecosystems as power factories (gas, uranium, wind and sun energy), as sources of mineral resources, and as retreats from the city, as well as from associated development-related damage, including road building, vegetation clearing, soil compaction, water extraction and pollution (Milton and Dean 2015).

We conducted a standardised distance-sampling survey across the southern Karoo to determine density and correlates of occurrence for steenbok.

5.2. Material and methods

Study area

The Nama and Succulent Karoo together cover 454,027 km² (Olson et al. 2000), approximately 26% of South Africa and at least 25% of Namibia. Annual rainfall is low (50–600 mm) and is often highly localised, varying annually in amount and timing (Desmet and Cowling 1999). The Karoo also experiences a dramatic range of temperatures, from the frost- and snow-prone high-altitude areas to the extremely hot northern regions bordering the Kalahari Desert (Dean and Milton 1999).

The Centre for Statistics in Ecology, the Environment and Conservation (SEEC), at the University of Cape Town, were consulted to select 60 pentads (five minutes of latitude by five minutes of longitude) within a study area identified by the Karoo BioGaps project. The Karoo BioGaps project aims to provide foundational biodiversity information to areas where shale-gas exploration concessions may be granted, predominantly in the southern Karoo. Pentads were selected by latin hypercube sampling (McKay et al. 1979) to ensure spatially near-random sampling with adequate coverage of the main environmental gradients (temperature, rainfall) and included arid sites in Albany thicket and Grassland Biomes (Figure 5.2a). An additional 4 pentads were sampled, making a total of 64 sites.

Point counts

We conducted 1193 point counts opportunistically within the 64 pentads (Figure 5.2a). Point counts of 10 minutes' duration were conducted throughout the target pentads, aiming for 20 point counts within a 24-hour period. Wind speed was measured at the start of each point count using a hand-held anemometer, and surveys were abandoned when it exceeded 5 m/s. Thus occasionally, fewer counts were achieved when counting conditions were poor. Within each 24-hour period point counts were conducted throughout the day (range: 05:56 – 18:44). Point locations were spaced 500 m to 1 km apart, depending on terrain and access permissions, as most pentads were on private land. An attempt was made to cover the width and breadth of each pentad, and to visit a range of elevations and point features. GPS position was recorded, as was elevation (metres above sea level). Surveys were conducted from 21 February to 25 October 2017, with most surveys (76%) conducted in the late-winter to mid-spring period (late July–October), referred to hereafter as the spring survey. Biosphere Expeditions assisted with collection of data for this survey in October 2017.

The survey was designed to collect information on birds and key 'wild' medium, diurnal mammals. Springbok, gemsbok and blesbok were not considered here as these are 'game' animals with stocking in this region determined by hunting activity. Here we only report on steenbok, as bird data has been analysed separately. For each group of birds and mammals detected, the following information was recorded: group size, activity related to distance sampling (perched or moving), and distance from observer (measured in metres using a laser rangefinder aimed at the centre of the group of animals).

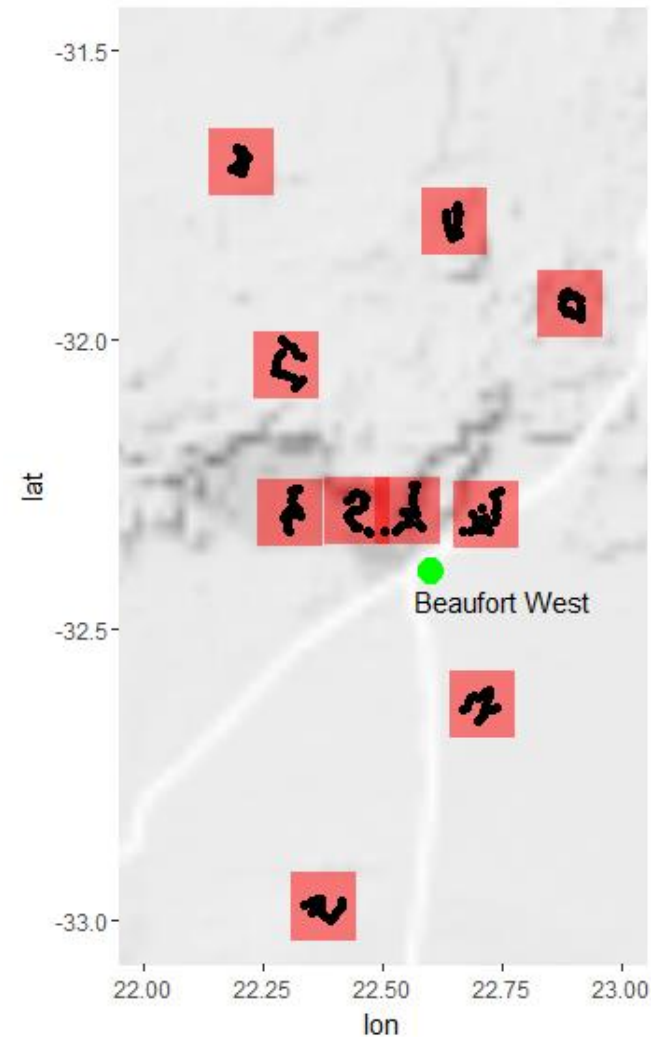
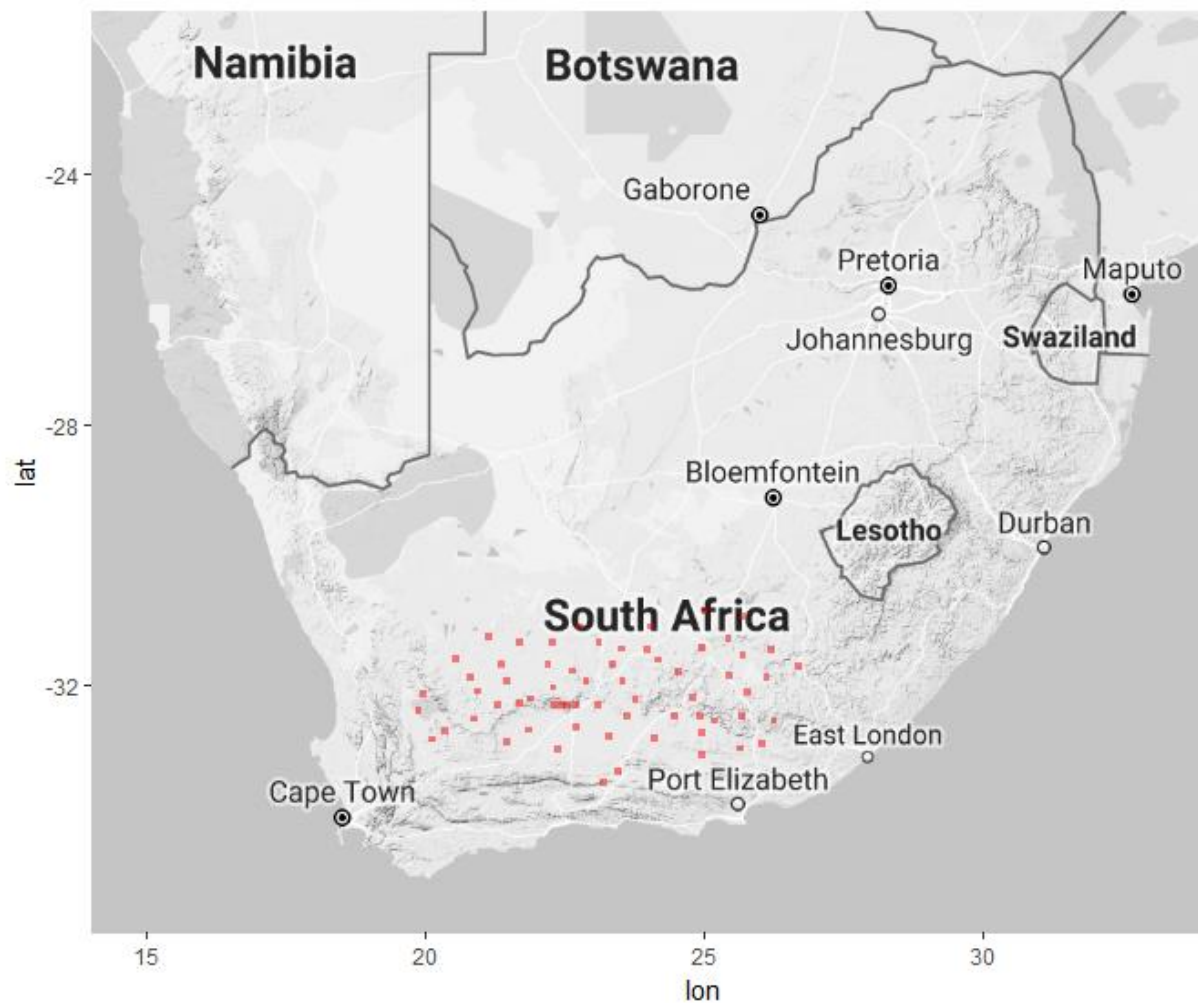


Figure 5.2a. Survey map with the location of surveyed pentads indicated as red squares (left) and example of point count locations (black dots) within pentads for the area around Beaufort West (right).

Calculation of densities from point counts

We used Distance for R (Miller 2017) to estimate densities at the pentad level, using the point count data. We fitted the data for each species using a variety of detection functions and adjustments, some with detection covariates: wind speed, daytime period (before or after 09:00), and season (late summer to early winter or spring). Detection functions tested included: hazard rate with no adjustments, half-normal with no adjustments, uniform, hazard rate and half normal with detection covariates, hazard rate with cosine adjustments and polynomial adjustments, half normal with cosine or Hermite polynomial adjustments. The best model was selected based on AIC and Cramer von-Mises values, which indicate goodness of fit. For missing group size data, we used species-specific mean group size. Truncation of data to exclude detections at outlying distances generally aids modelling of an appropriate detection function, so it was truncated at 90% of maximum distance. Density estimates were formulated as number of individuals per square kilometre.

Spatial determinants of steenbok presence

At each point, the average percentage grass and acacia tree (*Vachellia* sp.) cover was recorded (0 – 100). The average of each of these percentages across all points for each pentad were calculated. In addition, the presence of sheep was noted as a binary variable (present – absent) at each point, and then for each pentad the sum of points with sheep was calculated. These variables together with longitude, latitude and elevation were used as predictor variables in modelling of presence or absence of steenbok across the landscape.

To examine landscape, land-use and point features influencing steenbok presence at the point count level (steenbok recorded or not during a 10 minute count), generalised linear models using the binomial family and logit function were implemented. Where possible, the lmerTest package (Kuznetsova et al. 2017) was used to explore variables using pentad as a random effect. To identify the most important variables, the following model was created, and then the best model identified using the step function in R v3.5.0 (R Core Team 2017) as follows:

```
stats::step(glm(steenbok_pa~Lat+Lon+Grass_cover+  
Sheep+Altitude+acacia_cover, family = binomial, data=pentad_summary))
```


5.3 Results

Overall, we recorded 275 encounters with groups of mammals over the study period, across 12 species (Table 5.3a), of which the steenbok was most common with 88 group encounters at 75 points, across 32 of the 64 pentads. Density was 0.27 ± 0.05 (mean \pm SE) individuals per km², ranging from not recorded in some pentads to a maximum of 2.54 ± 1.17 .

Of the variables explaining steenbok presence in a pentad, only percentage grass cover and percentage acacia cover were retained in the best model, both negative predictors of steenbok presence, but with only acacia cover significantly so (grass: -0.007 ± 0.003 , $t = -1.79$, $p = 0.078$; acacia: -0.034 ± 0.011 , $t = -3.176$, $p = 0.002$). The distribution of these are illustrated in Figure 5.3.

Table 5.3a. Encounters with groups of mammals over the study period.

Species	Groups
Steenbok <i>Raphicerus campestris</i>	88
Rock hyrax <i>Procavia capensis</i>	75
Vervet monkey <i>Chlorocebus pygerythrus</i>	37
Greater kudu <i>Tragelaphus strepsiceros</i>	17
Cape ground Squirrel <i>Xerus inauris</i>	13
Bat-eared Fox <i>Otocyon megalotis</i>	11
Meerkat <i>Suricata suricatta</i>	8
Baboon <i>Papio ursinus</i>	7
Grey duiker <i>Sylvicapra grimmia</i>	7
Yellow mongoose <i>Cynictis penicillata</i>	6
Klipspringer <i>Oreotragus oreotragus</i>	3
Aardwolf <i>Proteles cristata</i>	1

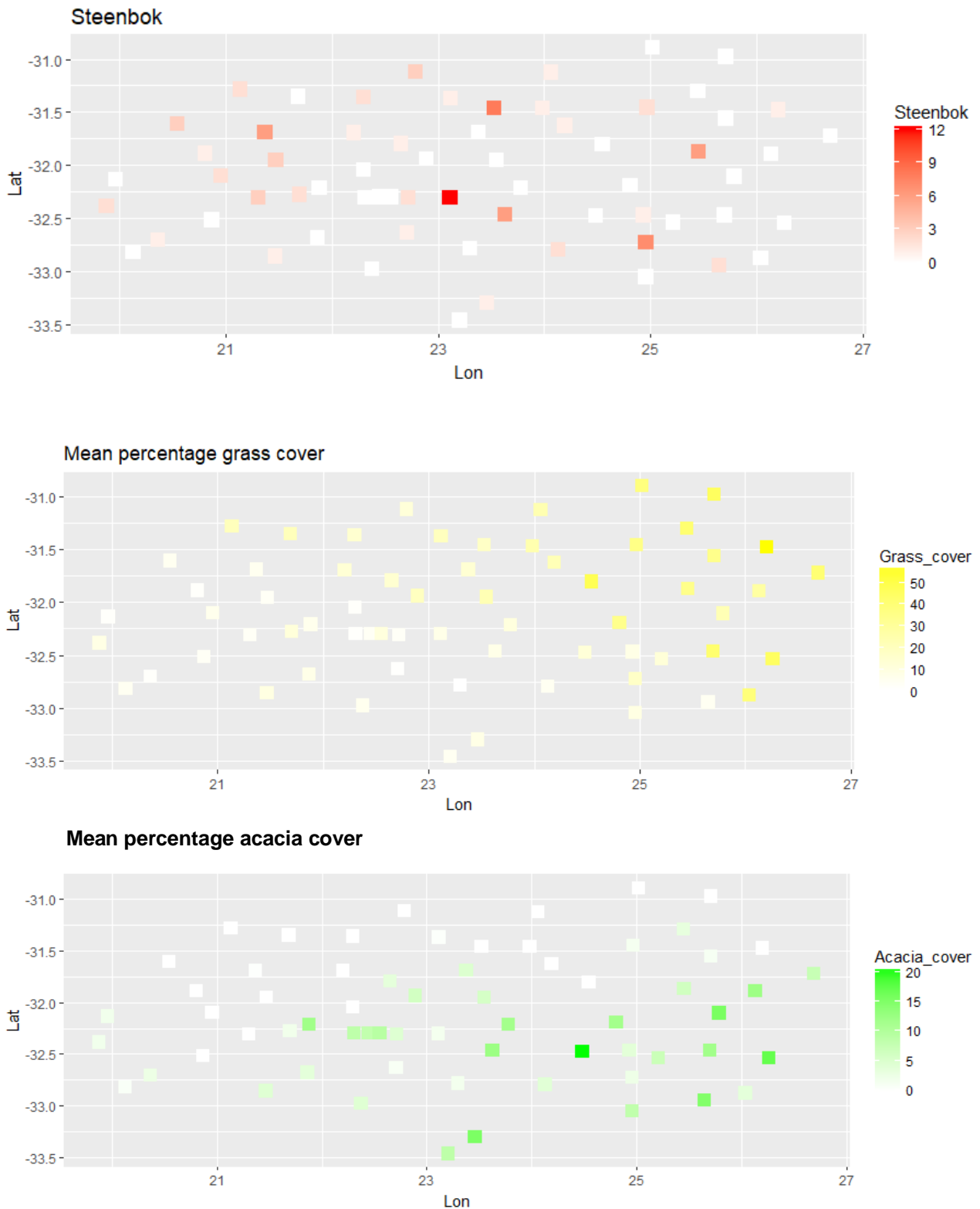


Figure 5.3. Total steenbok group encounters across the pentads (squares). Steenbok presence was negatively correlated with increasing grass cover and increasing acacia cover. The summary of these variables by pentad are illustrated, with increasing coverage represented by darker colour intensity.

5.4. Discussion

For our survey across the southern Karoo region of South Africa, we obtained low density estimates for steenbok compared to other studies (Stenkewitz et al. 2010). While overall density estimates were low, there was large variation between pentads. If these estimates represent background density estimates across the Karoo, this would mean a population of approximately $400,000 * 0.27 \pm 0.05 = 108,000$ (88,000 – 128,000).

This information on densities and population estimates should prove useful in reviewing the conservation status of certain arid zone species. To the best of our knowledge, a detailed conservation assessment has not yet been undertaken for many of the wide-ranging Karoo species. Density estimates and inferred population sizes, together with knowledge of species' range sizes, are critical components for determining species' threat status and designing appropriate conservation strategies (IUCN Standards and Petitions Subcommittee 2017).

Knowledge of range and population size is also critical for planning the potential expansion of protected areas and conservation areas in the Karoo biome, and for identifying critical habitat refuges. The burgeoning threats of shale gas mining (fracking), uranium mining, and loss of habitat due to large-scale infrastructure projects, such as the development of renewable energy facilities, in this region necessitate a rapid improvement in our knowledge of Karoo species, their habitat requirements and distribution.

Despite the significance of our results, the study has several limitations: surveys were conducted during the day only and as such may have missed nocturnal activity. Also, our density estimates are derived from a sample area of <1% of the Karoo and the point sampling strategy was not truly random.

5.5. Acknowledgements

Special thanks to Ryan Black and family, Hennie Visagie, Tim Hobson, Willie Jordaan, Andre Theron and family, and Okkie Wiehaan and family for hosting the authors in their homes during the survey. Gaynor Rupert is thanked for her financial contribution to BirdLife South Africa, which made this work possible. We thank Dale Wright, Campbell Fleming, Daniel Marnewick and Mark Anderson for their support.

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6. Assessments of yield and fecundity of *Cyclopia intermedia* (honeybush) in relation to selected environmental variables, at Blue Hill Nature Reserve

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Summary/Abstract

Cyclopia intermedia (honeybush) is a plant species endemic to the Fynbos Biome of South Africa. It is used commercially for the production of tea, for which most of the current supply is harvested from the wild. Surprisingly, there has been little research conducted on the species in the wild with regards to preferred growing locations, or variables associated with plants that would produce the most biomass. The species is a resprouter, regrowing from underground rootstock after fires, and as such resprouts once it has been cut. However, there is concern that premature cutting may be compromising rootstock recovery in relation to non-harvested plants, and that there may be selection during harvesting against the very characteristics of plants (height and stem number) that would increase yield. We calculate a simple to use relationship that would allow one to easily quantify yield based on height and number of stems. We show that plants in valleys have the highest yield values and potential fecundity (measured as numbers of buds per stem), and that plants grow taller (and hence have higher yield) when surrounding vegetation cover is high. We speculate that this is due to moisture gradients, but further work is required to clarify these patterns.

6.1. Introduction

Cyclopia species (Honeybush) (Figures 6.2b & c) are indigenous and endemic plants of the fynbos biome of South Africa, harvested and cultivated for tea production (McGregor 2017). Most of the tea currently comes from wild harvested populations (Joubert et al. 2011). The tea has become popular due to purported medicinal properties (Joubert et al. 2008, McKay and Blumberg 2007). The honeybush plant is a long-lived perennial, which takes the form of an erect, robust shrub up to 2 m tall that sprouts after fire. It is the dominant *Cyclopia* at Blue Hill Nature Reserve and the most widespread species in the genus (range approx. 13,900 sq km), with a distribution that stretches across the Cape Fold mountains. It is mainly a higher altitude species. Much of the species distribution range falls in mountain water catchment areas and the populations are therefore well represented in formally protected areas e.g. Baviaanskloof Nature Reserve. Current IUCN status of the plant is Least Concern, although there are concerns over population declines as a consequence of commercial harvesting and as such it is listed as Declining in the Red List of South African Plants (2017). Although cultivated, information on wild honeybush is becoming increasingly important, as this industry is as yet in its infancy and there still is relatively little known on how to maximise yield. We look at potential yield in a wild population to look for information that may be of use to the industry in terms of maximising yield of wild and cultivated populations.

Wild harvesting can have implications for plant recruitment and target species populations. Harvesting at inappropriate times during a plant's reproductive cycle can reduce seed output and hence target plant populations in the future. The reproductive cycle in honeybush (budding to seed shedding) lasts as long as eight months. The plants generally flower in spring (September/October), but the buds begin to form in April/May. The pods form after flowering in October/November and seed is set thereafter (November/December). There are concerns around recovery of the plant in terms of its underground rootstock after cropping and the time interval needed for sufficient recovery of reproductive capacity to set seed and replenish soil seed stores. In areas where rainfall is good, plants flourish after cropping and reach a good size in 18 months. They may appear to be ready to harvest, but will not have had time to produce seed banks that will be competitive with non-harvested species. In addition, since payment to harvesters is by weight harvested, there may be pressure to select larger plants over smaller plants, leading to selective pressure against preferred plant growth forms.

Aims of the study were to (a) quantify potential yield in relation to habitat / environmental variables, (b) quantify potential fecundity (bud set) as a function of plant growth features and (2) measure the overall health of plants.

6.2. Material and methods

Monitoring plots

The study site is Blue Hill Nature Reserve, a CapeNature Stewardship with a biodiversity plan managed by Eastern Cape Parks and Tourism Agency. The biodiversity stewardship management plan provides provision for the harvesting of honeybush, provided this is reported. No harvesting has taken place on the land since 2010, which was prior to the designation of the property as a stewardship nature reserve in 2013. A fire in the areas associated with honeybush presence occurred in February 2012. Thus at the time of plant measurements, re-growth on all plants was five years.

During October 2017, 284 honeybush plants were measured at five plots (Figure 6.2a) with the help of Biosphere Expeditions citizen scientists (Figure 6.2b). A further four plots of 116 plants were marked in December 2017.

The location of plants were marked with metal droppers (for visibility, but robust enough to resist baboon attention), with individual metal tags wired to the poles together with flagging tape. Poles were painted according to the honeybushes' treatment type for an offtake experiment e.g. red = control (no harvesting); green = 100% cut; green + blue = 100% cut with ash; green-red-green = 100% cut over two years; red + green = 50% cut only. Ash for the treatment of cut plants was white ash that had been derived by burning invasive black wattle and pine wood in a wood stove. The wood burnt did not contain seeds, and it is unlikely seeds would have survived the high heat burning process. A 500 ml container was used to apply the ash, which weighed on average 120 g. The ash treatment was applied to mimic the nutrient load that might accompany a wildfire: the usual situation where plants resprout.

Location of marked Honeybush plants at Blue Hill Nature Reserve

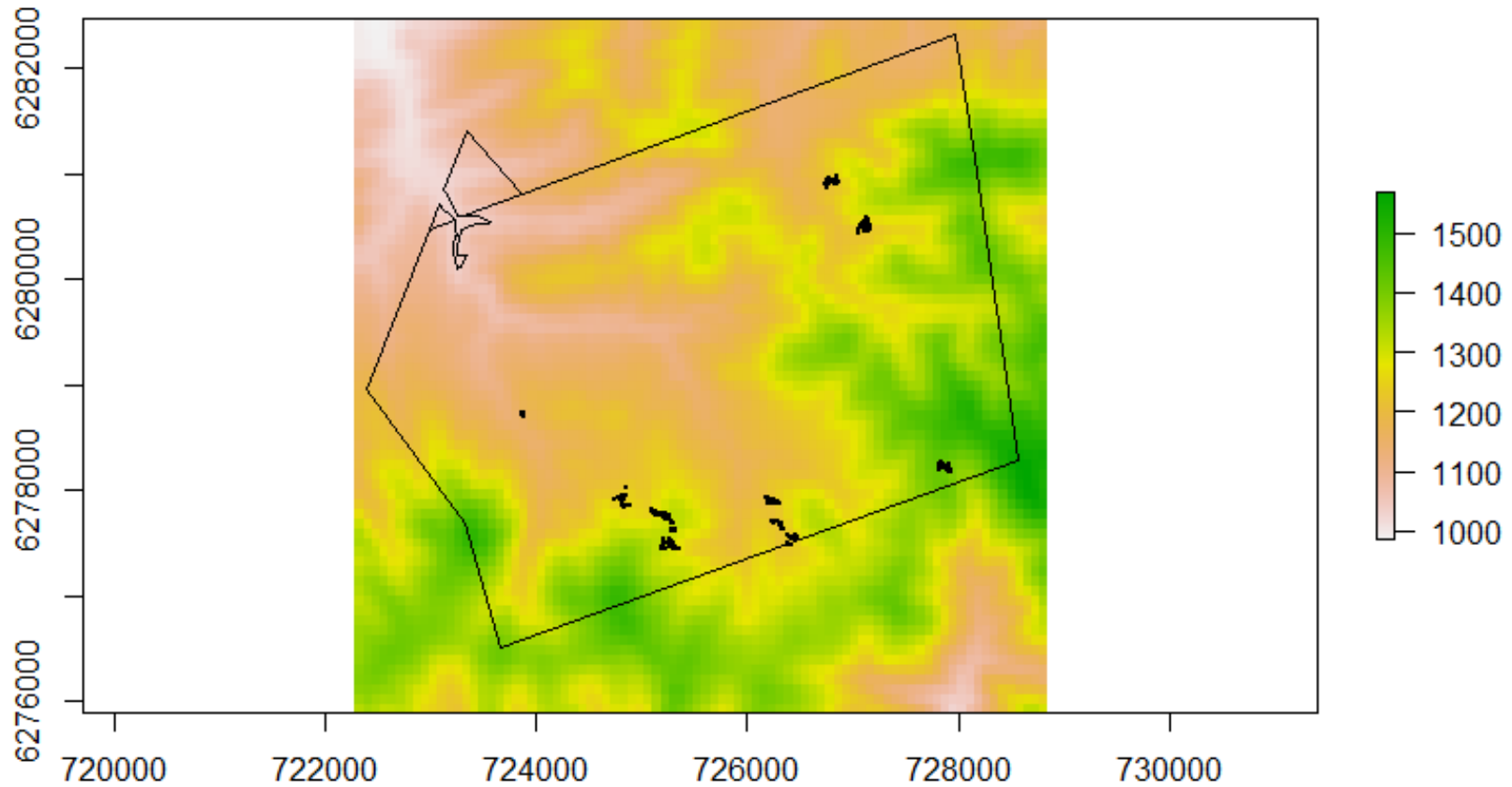


Figure 6.2a. The location of 400 honeybush plants (black points) measured for this report at Blue Hill Nature Reserve (black boundary), Western Cape, overlaid on a digital elevation map (metres) with green high and white low (ca. 1000 m).



Figure 6.2b. Citizen scientists at work with the honeybush plants.

Plant measurements

The following information was recorded for each plant:

- Number of live stems at 30cm above ground level
- Number of dead grown stems
- Maximum height (cm)
- Crown width (cm)
- Stem colour
- Plant condition as leaf count of 15 cm for top 3 tallest live stems plus presence/absence of die-back
- Evidence of previous harvest
- Number of buds per 15 cm of stem (x3)

The number of buds was only recorded in October.

Environmental variables:

- Aspect (degrees)
- Slope angle (0-90)
- Percentage rock, soil and vegetation cover within 2.5 m radius
- Presence of major plant functional groups (broad leaved shrubs, fine-leaved shrubs, succulents, protea, dwarf shrub, grass, restio, other honeybush) within 50cm of the plant
- Parasite presence e.g. dodder
- Clinal position (valley bottom, lower slope, mid slope, upper slope).

Data were recorded in a bespoke Cybertracker app on a Huawei smartphone, which also recorded altitude and GPS location.

Plant harvesting

In March 2018, 157 weight measurements were taken for plants harvested using traditional harvesting techniques (sickle or secateurs) following the recommendations of (McGregor 2017). Weight was recorded in grams using an electronic balance in the field.

Statistics and analysis

Yield: To calculate a metric of potential yield, we multiplied height by the number of live stems and log transformed the result to normalise the resulting distribution (see supplementary Figure 6.7a for the histogram of the distribution). We chose height over plant width as the latter metric showed considerable variation (supplementary Figure 6.7b); and height and width were correlated (Pearson's product-moment correlation: $t = 19.047$, $df = 399$, $p\text{-value} < 2.2e\text{-}16$). We used a standard linear regression to quantify the relationship between potential yield and actual yield (weight in grams) of the 157 harvested plants. We tested if potential yield was a function of stem colour (yellow or orange to red) or clinal position using a one-way analysis of variance (ANOVA) implemented using the `aov` function in R statistical environment.

For the continuous variables altitude, slope angle, percentage soil, vegetation and rock cover within 2.5 m of a plant, the relationship with potential yield, height and number of live stems was examined using standard linear regression models. Mean \pm standard deviation are provided unless indicated otherwise.

Fecundity: We created an index of potential fecundity by taking the mean of the number of bud counts across the three stems of each plant for which this was measured. Many plants showed no evidence of buds, and so this index is heavily zero weighted. We thus consider fecundity as a binomial index: presence or absence of buds. We modelled the presence of buds as a function of stem colour, clinal position, plant height and stem counts using binomial logistic regression.

Plant health: Plants were scored as healthy (<10% of growing tips dead; see below), presence of minor dieback (11 – 50% dead tips); heavy dieback (>50% dead tips); major dieback (dieback extending extensively beyond tips) (Figure 6.2c).



Figure 6.2c. Honeybush plants with mild dieback (left) and healthy (right).

6.3. Results

Yield

There was a good linear relationship between the potential yield metric (plant height x number of live stems) and weight ($t = 17.3$, $p < 0.001$, $\text{adj-}R^2 = 0.66$, $\text{df} = 155$), represented by the formula: $\text{Weight (g)} = 0.43 \times \text{Potential Yield} - 20.4$.

The position of a plant in a landscape was related to yield (ANOVA $F = 4.93$, $p < 0.001$, $\text{df} = 4$), with plants in the valleys having higher yield compared to those on the slopes (Figure 6.3a). There was no difference in potential yield as a function of stem colour (log stems*ht: yellow = 7.93 ± 0.9 ; orange = 7.81 ± 0.9).

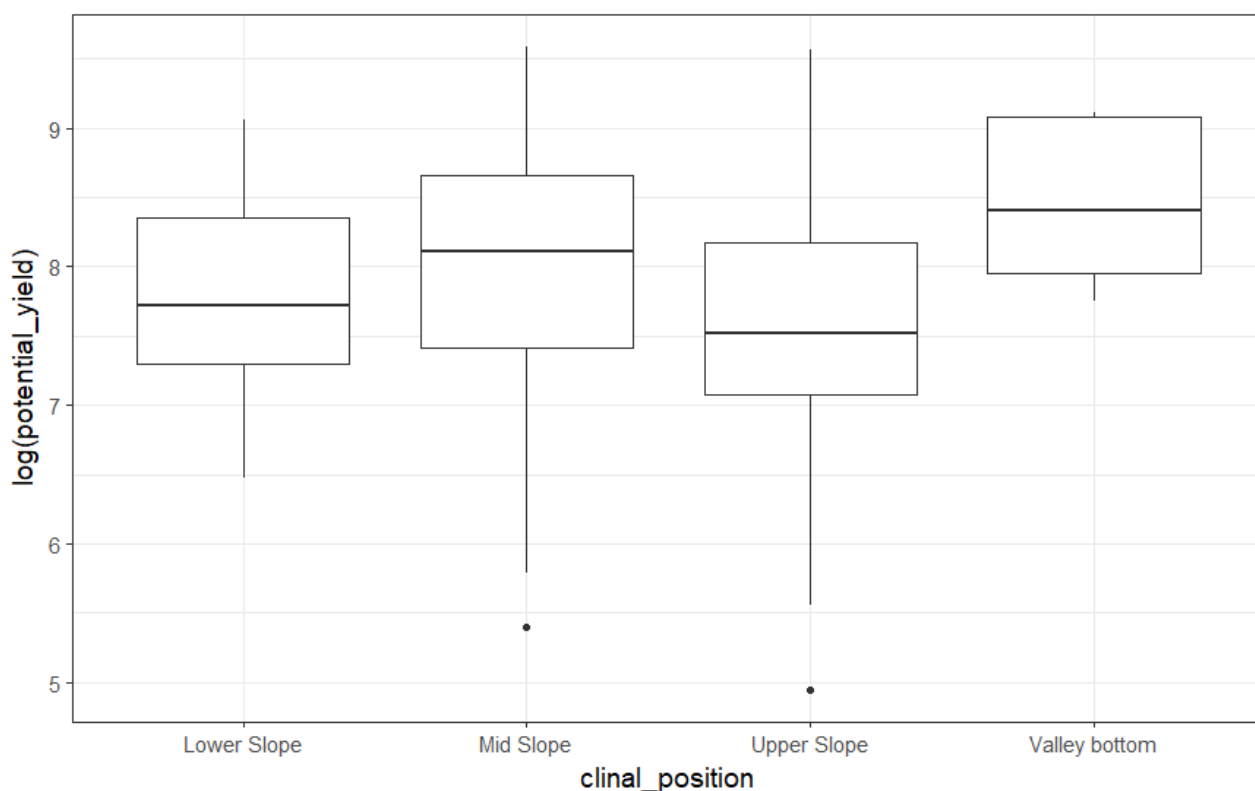
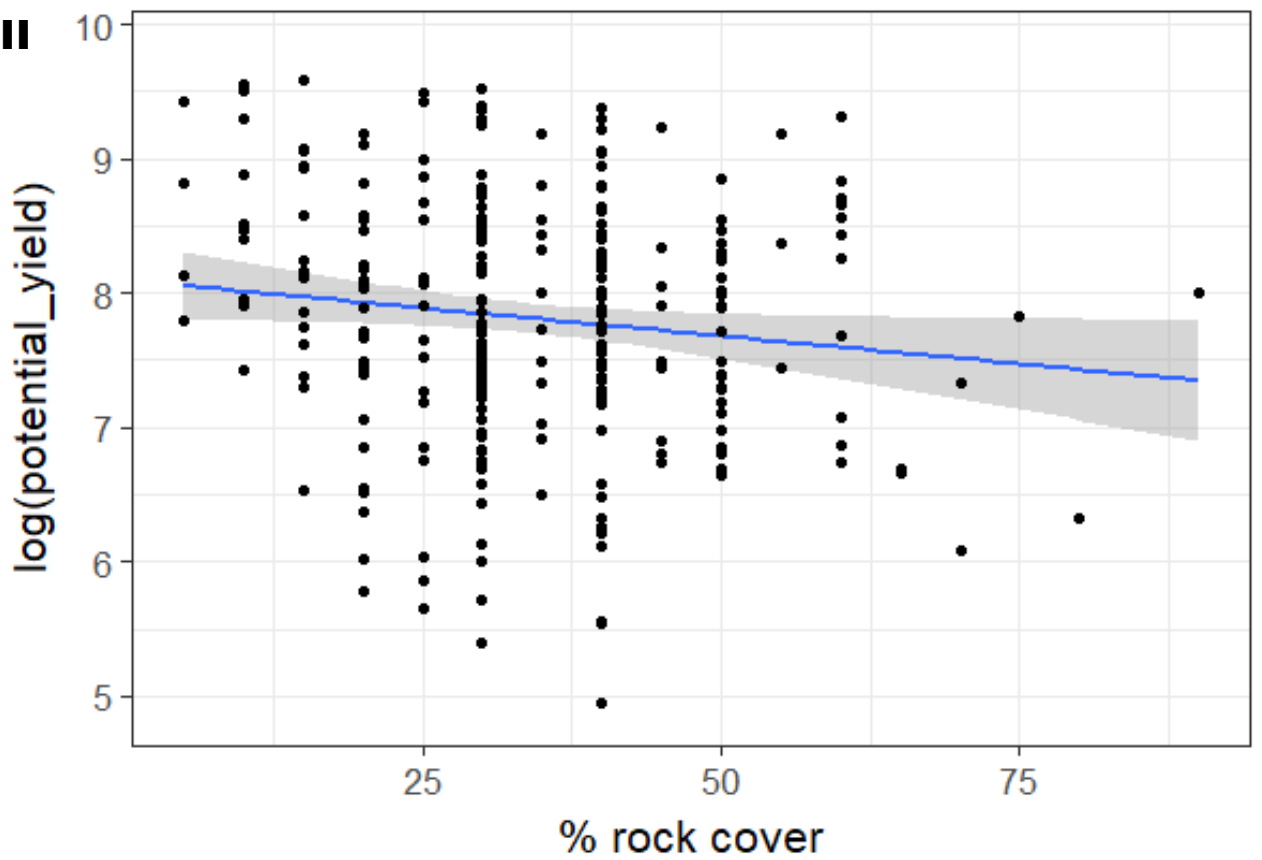
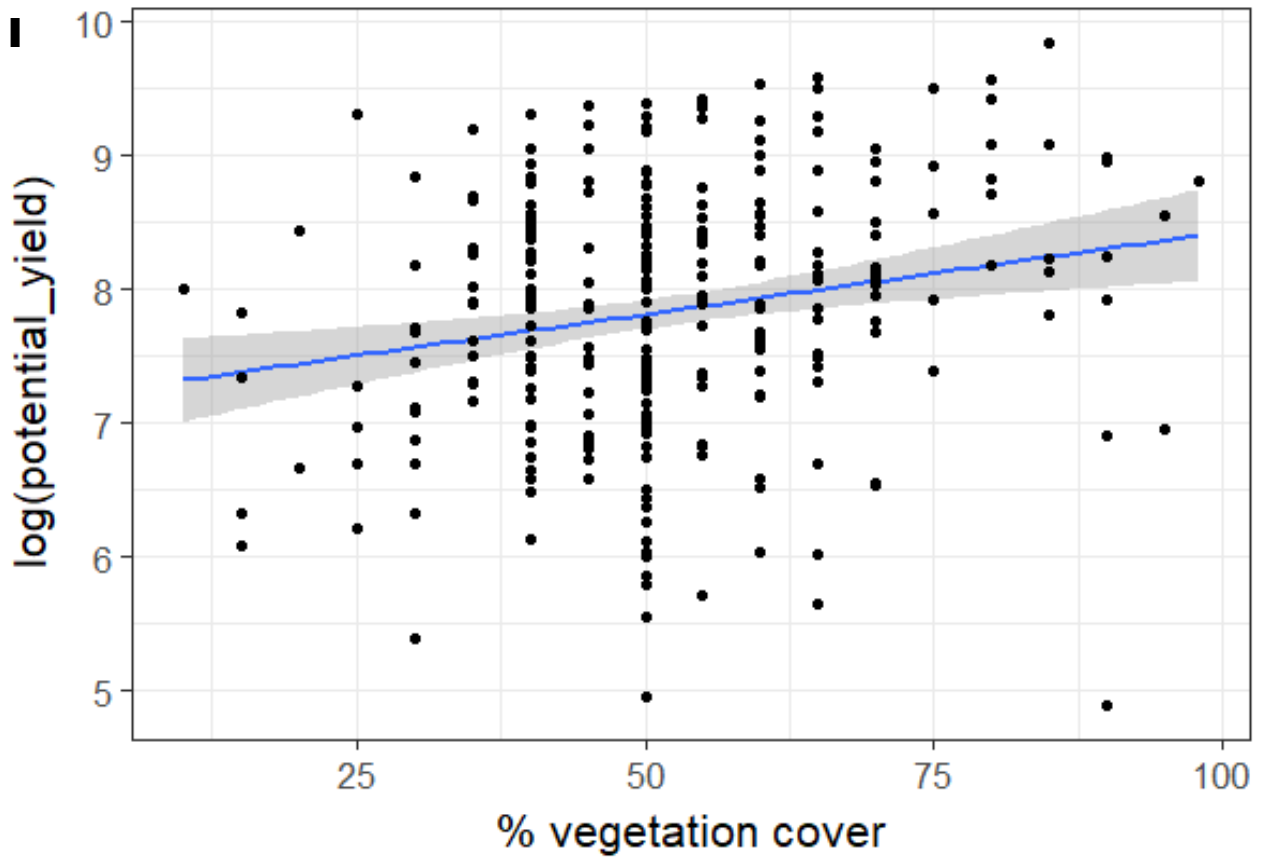


Figure 6.3a. The relationship between potential honeybush yield and position in the landscape (clinal position) illustrated with a box and whiskers plot. For this dataset yield was highest for plants in valleys compared to slopes.

There was a significant positive correlation between potential yield and percentage vegetation cover within 2.5 m of the target plant ($t = 4.4$, $p < 0.001$, $\text{df} = 398$, Figure 6.3bI); and a significant negative correlation between percentage rock cover and potential yield ($t = -3.01$, $p = 0.003$, $\text{df} = 383$, Figure 6.3bII), while there was no correlation between potential yield and percentage exposed soil cover ($t = -1.60$, $p = 0.11$, $\text{df} = 394$). Breaking down potential yield into its components of height and number of stems, while both were correlated with percentage vegetation cover, the relationship was stronger for plant height (plant height: $t = 6.37$, $p < 0.001$, $\text{df} = 399$, Figure 6.3bIII; number of live stems: $t = 3.31$, $p = 0.001$, $\text{df} = 398$, Figure 6.3bIV). There was also a weakly significant negative correlation between potential yield and altitude ($t = -2.11$, $p = 0.04$, $\text{df} = 398$). There was no relation between yield and slope angle ($t = -0.56$, $p = 0.58$, $\text{df} = 398$).



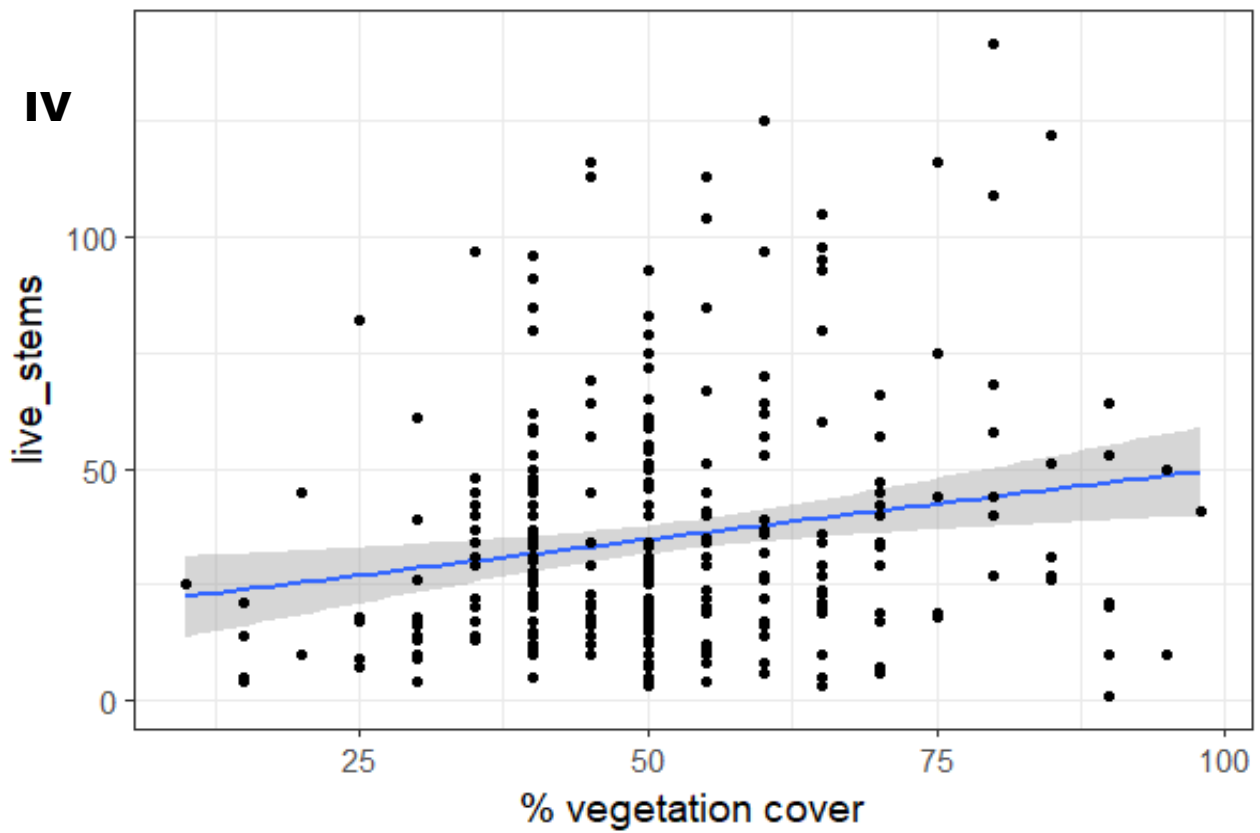
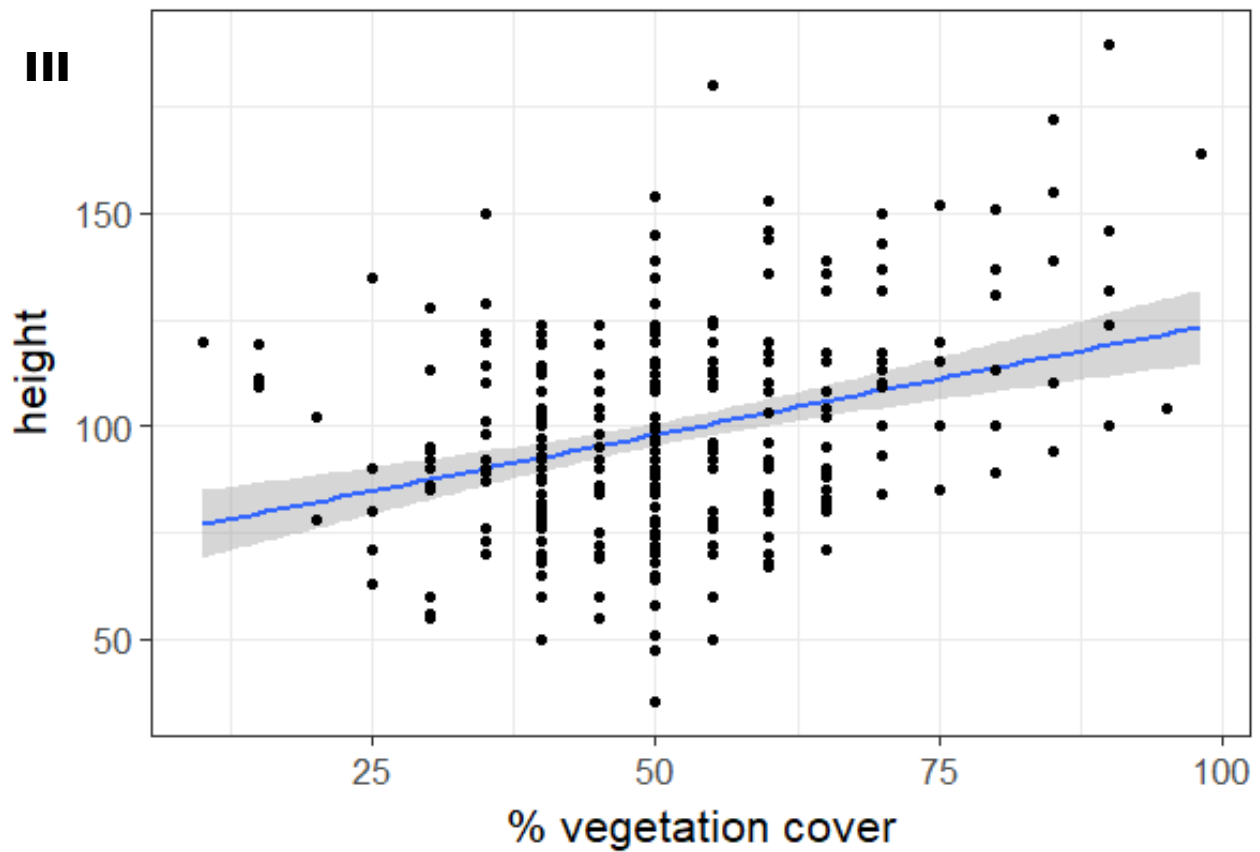


Figure 6.3b I-IV. Potential yield as a function of vegetation cover (I) and rock cover (II). Components of potential yield: plant height (III) and number of live stems (IV). Blue lines are regression slopes, with grey shading indicating the standard error.

Fecundity

The presence or absence of buds was strongly linked to plant height (logistic regression coefficients: 0.027 ± 0.006 , $z = 4.33$, $p < 0.001$, $df = 284$; Figure 6.3c), and less so with the number of live stems (logistic regression coefficients: 0.009 ± 0.005 , $z = 1.75$, $p = 0.08$, $df = 284$). Whether bud presence is related to clinal position is less clear, mostly due to uneven sampling across the categories, and taller plants tended to be associated with the valley bottoms (supplementary Figure 6.7c). There was no relationship between probability of bud presence and stem colour (logistic regression intercept coefficients (orange stems): 1.02 ± 0.16 , $z = 6.27$; yellow stems: -0.28 ± 0.27 , $z = -1.02$).

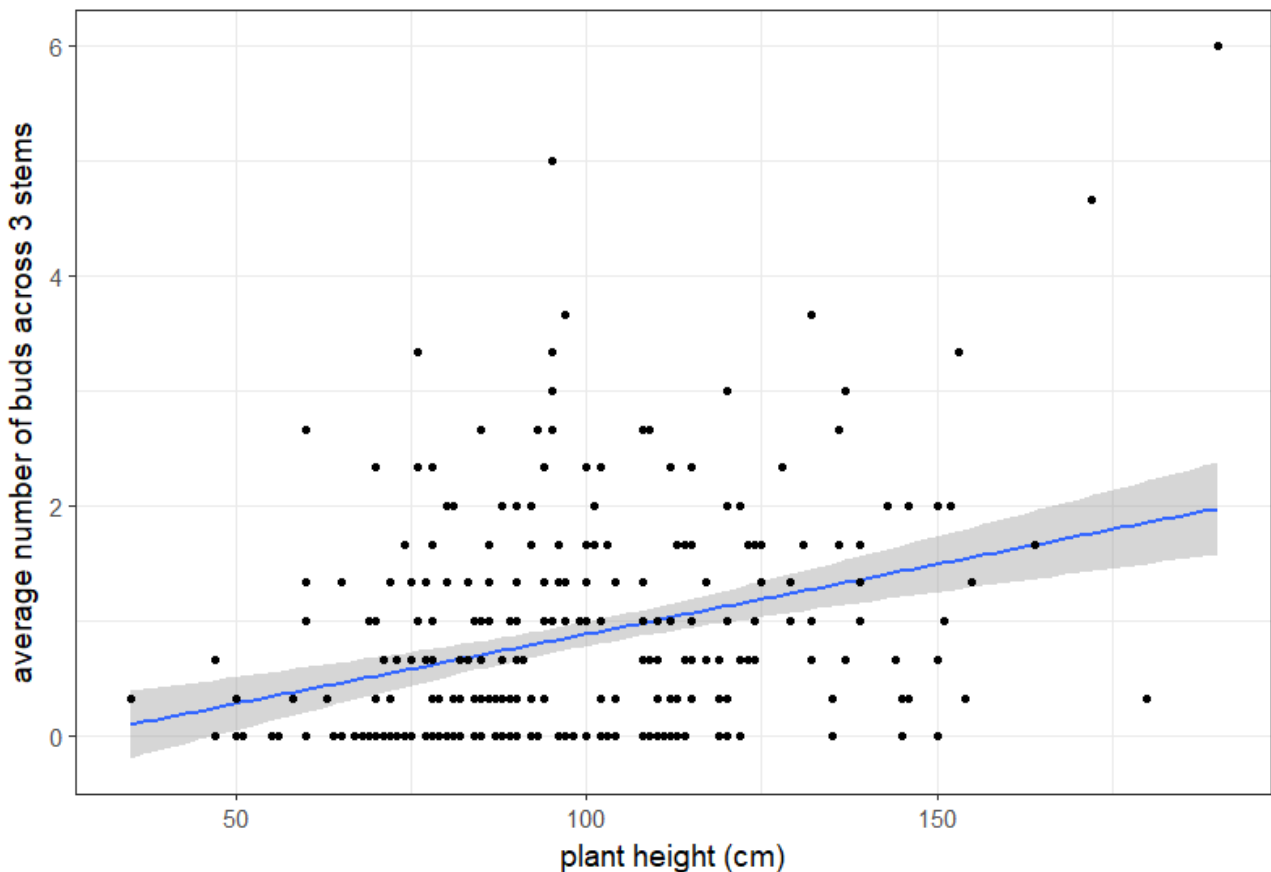


Figure 6.3c. There is a broad pattern for more buds on taller plants. The fit of this relationship between these variables is illustrated with a regression slope (blue line) with standard error illustrated by grey shading.

Overall plant health

Only 33% of plants were scored as healthy, leaving 67% of plants scoring some category of dieback (Figure 6.3d). Of these, mild dieback was mostly evident, with only some plants registering severe dieback. A few dead plants ($n < 5$) were also found, but not documented. Plants were generally free of parasites (one gall recorded), no dodder was found, and no scale or cochineal type insects were recorded. Some leaves displayed evidence of insect herbivory (noted, but not documented).

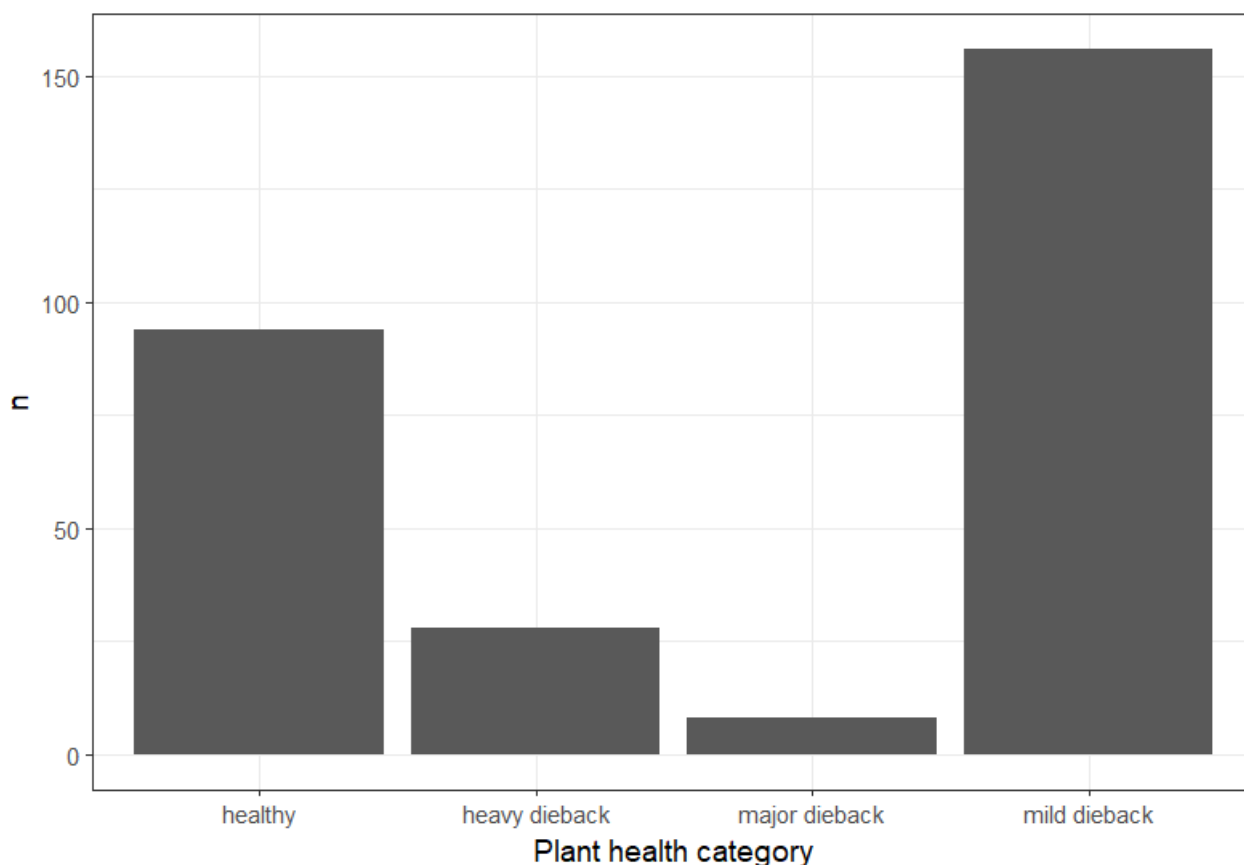


Figure 6.3d. Honeybush plant health, with the total number of plants in each category illustrated.

6.4. Discussion

Overview and caveats

This report provides preliminary information about a population of wild honeybush from the Blue Hill Nature Reserve in the Kouga mountains. The plants all form part of one population and results may differ between populations, e.g. the southern Langkloof, which has higher rainfall. The relationship between actual yield per plant and our metric of potential yield also needs to be validated at other locations. As it can be time-consuming to count stems, it may be quicker to calculate the circumference of the basal area for an improved formula for calculating yield in the future.

Yield

In the field there is often a distinctive difference in the colouration of the stems of *C. intermedia* plants. This can range from yellow to almost purple. The reasons for these differences in colour are as of yet unknown, but do not seem to be linked to our metric of potential yield. Likewise, colour was not linked to fecundity.

It should be noted that young plants can also have greenish stems: but there were none of these at this study location. McGregor (2017) states that plants with green stems should not be harvested: it is likely these plants have higher tannin content and this affects the quality of the tea.

Of interest was that potential yield was a function of both clinal position (highest in the valley bottoms compared to the slopes), as well as the percentage rock and vegetation cover. We suspect at this stage that these may be a function of soil depth: areas with high exposed rock cover likely have lower quantities of soil, which is unlikely also to be the case in the valley bottoms, where soil deposition from the slopes occurs. It is likely that the positive relationship between vegetation cover and yield may be similarly a function of soil depth: there is higher growth potential for plants with greater soil depth. This relationship remains to be validated. However, the strongest correlation with potential yield was with percentage vegetation cover, suggesting that plants at least may grow taller in the presence of other plants.

Fecundity

While we accept that a plant's reproductive potential is likely to be a lot more complicated than the presence or absence of buds, a count of buds will be critical to any later measures of fecundity, e.g. seed counts. Thus, it is of concern that the presence of buds was best explained by plant height: tall plants are certainly targeted by those harvesting wild honeybush as a priority. Wild harvesting is thus now an intrinsic selection pressure against tall plants, with implications for future population recruitment.

Overall plant health

The overall poor condition of the plants recorded in this study, as evidenced by shoot dieback, cannot be put down to a single reason at this time. It may be the consequence of two years of below average rainfall (there is plenty of evidence of dieback of young proteas on north-facing slopes of the reserve), or simply that the plants have now reached their reproductive peak and are in decline. More data are needed before firm conclusions can be made.

6.5. Acknowledgements

Thank you to the citizen scientists, especially Sandy Kerr, for making the aluminium numbered tags. Thanks also to Jeff and Arlene Jackman and the students of Oudeberg Nature Reserve for assisting with the marking and measuring of some plants.

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6.7. Supplementary information

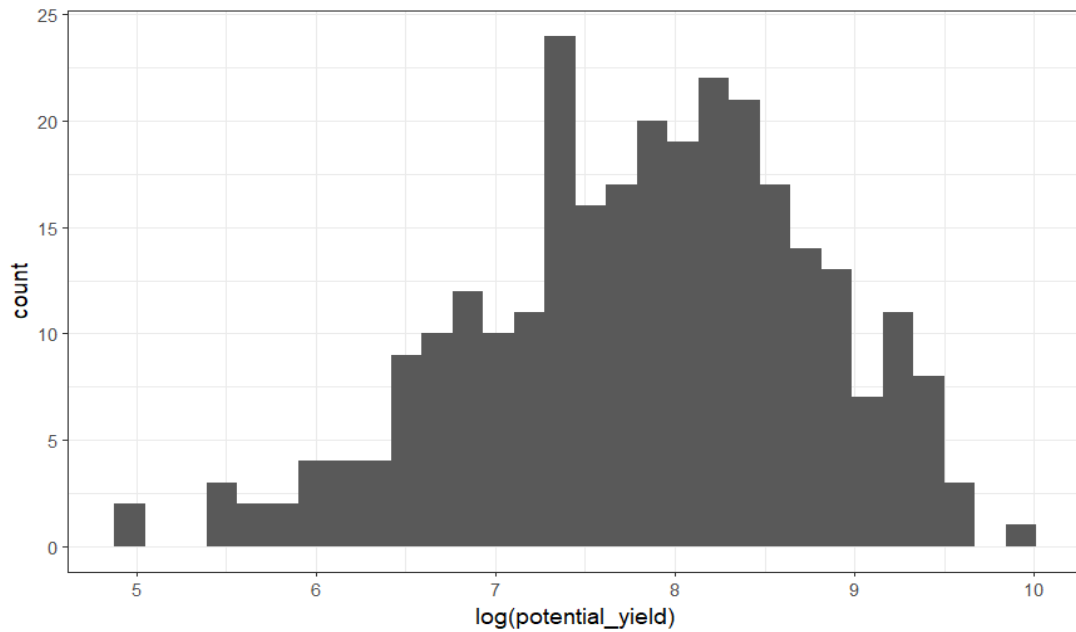


Figure 6.7a. A histogram of the log of an index of potential yield from honeybush (stems * height), the values used in modelling of covariate influence on potential yield

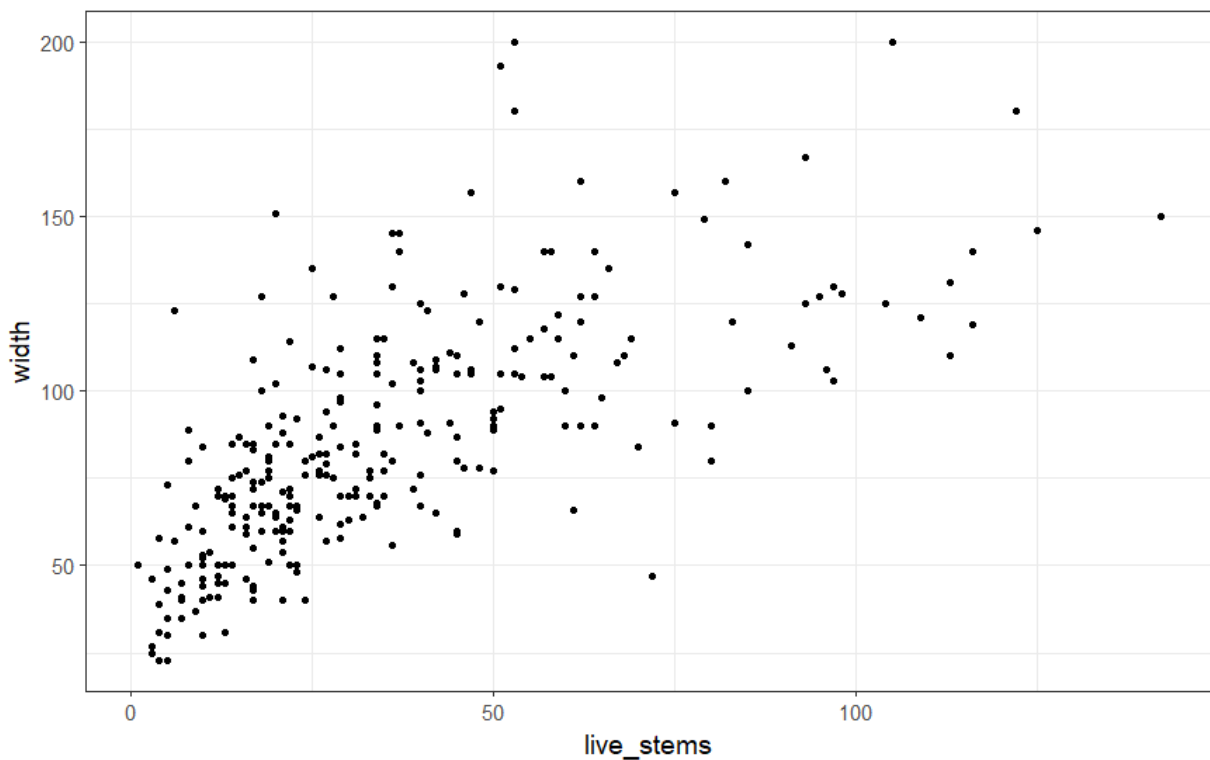


Figure 6.7b. The relationship between honeybush plant width (cm) and number of live stems at 30cm, indicating a correlation between these metrics with considerable variability.

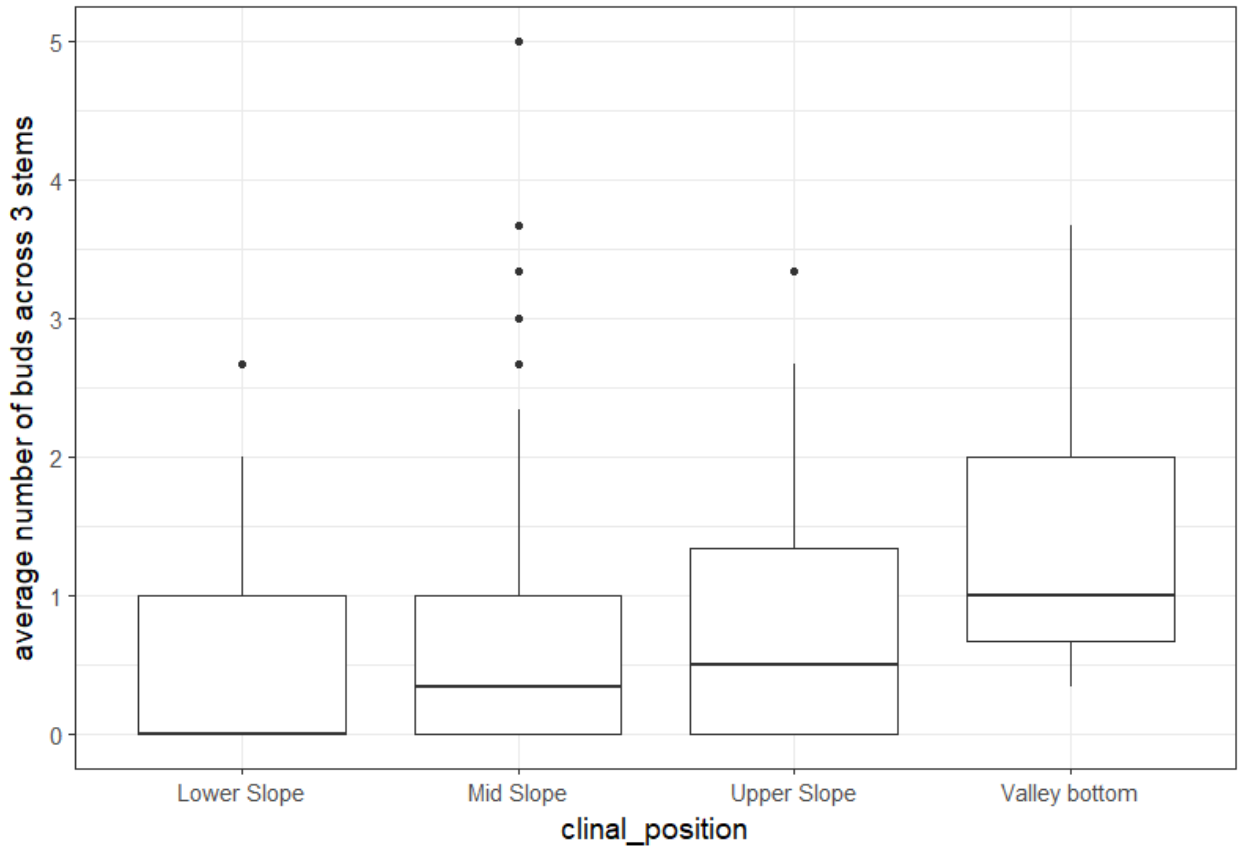


Figure 6.7c. Average number of buds for plants across different clinal positions. While bud counts were superficially higher in the valley bottoms, this is also where the taller plants were found.

Appendix I: Expedition diary and reports



A multimedia expedition diary is available at <https://biosphereexpeditions.wordpress.com/category/expedition-blogs/south-africa-2017/>.



All expedition reports, including this and previous expedition reports, are available at www.biosphere-expeditions.org/reports.

More pictures, videos, media coverage of the expedition are available via www.biosphere-expeditions.org/southafrica.