Journal of Arid Environments 75 (2011) 836–841

Contents lists available at ScienceDirect

Journal of Arid Environments

journal homepage: www.elsevier.com/locate/jaridenv

# Comparing biodiversity between protected savanna and adjacent non-protected farmland in the southern Kalahari

Bernd Wasiolka*, Niels Blaum

*Department of Plant Ecology and Nature Conservation, University of Potsdam, Maulbeerallee 2, 14469 Potsdam, Germany*

ARTICLE INFO

*Article history:*
Received 6 November 2009
Received in revised form
5 April 2011
Accepted 6 April 2011
Available online 4 May 2011

*Keywords:*
Kgalagadi Transfrontier Park
Rangeland
Degradation
Species richness
Conservation
Reptile

ABSTRACT

In this study we investigated the effect of different land use options (wildlife versus livestock) on species richness of plants and reptiles in the protected Kgalagadi Transfrontier Park (KTP) versus adjacent non-protected farmland within the same savanna habitat type (Aoub dune veld) in the southern Kalahari, South Africa. Our results show that both plant and reptile species richness as well as plant cover and reptile abundance was significantly higher in the protected KTP than in the non-protected farmland. The higher proportion of shrub but lower proportions of perennial grass cover, herb cover, and herb species richness in the farmland can be explained by higher stocking rates and the differences in feeding behaviour between native wild ungulates (e.g. *Antidorcas marsupialis*, *Oryx gazella*) and livestock (mainly sheep). The reptile's prey availability and microhabitats (perennial grass tussocks and rodent burrows) for thermoregulation and protection against predators were significantly lower in the farmland. To conclude, our results clearly show that long term effects of different land use options (wildlife in protected KTP versus extensive livestock production in the non-protected farmland) even within the same habitat type have led to significant changes in vegetation composition, availability of microhabitat structures and in the reptile community.



## 1. Introduction

Protected areas are widely recognized as a key determinant in protecting and conserving biodiversity (UN, 1992) and defined (IUCN, 1994) as:

> "An area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means".

The need for identifying changes in species diversity and analysing the drivers of change within protected areas, initiated biodiversity studies with manifold foci: (i) analysing the impacts of anthropogenic threats (e.g. wood harvesting, hunting, grazing) on protected areas (e.g. Bruner et al., 2001; Olupot et al., 2009), (ii) assessing the changes in species diversity since proclamation (e.g. Newmark, 1996; Stephens et al., 2001), (iii) establishing species inventories (e.g. Martinoli et al., 2006) and (iv) analysing the general state (e.g. habitat quality, fragmentation) of the protected area (e.g. Liu et al., 2001).

However, relatively little attention was given to analyze differences in species diversity between protected and neighboring non-protected areas and in particular, how much more species can be conserved in protected areas compared to neighbouring non-protected areas. Nevertheless, in most studies, where species diversity was compared between protected and non-protected areas, the originally similar habitat types differed substantially in vegetation cover, composition and structure at the time of observation (Caro, 2001). As expected, in such cases, species diversity, composition and abundance differed largely between the different habitat types. For example, in Mexico, rodent diversity was compared between protected forest and the non-protected crop farming land (Horvath et al., 2001), and small mammal diversity was compared between protected short grass savanna sites, non-protected secondary forest, and former plantation sites in Ghana (Decher and Bahian, 1999). Studies, where species diversity was compared between protected and non-protected areas within the same habitat type (vegetation composition and structure) but differ in land use are scarce. For example, in South Africa, reptile and arthropod diversity was investigated in protected areas and adjacent rangeland in xeric succulent thicket (Fabricius et al., 2003).

The Kgalagadi Transfrontier Park and the surrounding commercial farmland in the semiarid savanna of the southern Kalahari forms a useful system to study the effects of different land use options

* Corresponding author. Tel.: +49 0331 9771959; fax: +49 0331 9771948.
*E-mail address:* wasiolka@uni-potsdam.de (B. Wasiolka).

0140-1963/$ – see front matter © 2011 Elsevier Ltd. All rights reserved.
doi:10.1016/j.jaridenv.2011.04.011

(wildlife versus predominantly livestock) on species diversity in protected versus non-protected area within the same habitat type. Differences in plant composition between the protected Kgalagadi Transfrontier Park and the neighbouring livestock farms caused by the two different land use options are expected to occur. However, whether and to what degree these differences in vegetation composition and diversity between both land use options will also affect animal diversity remains unclear. We selected reptiles as focal organisms to assess differences in animal diversity for two major reasons. First, reptiles form an important part in the Kalahari food web: Kalahari lizards are insectivorous and depend on a large variety of arthropods for foraging (Pianka, 1986; Branch, 1998). The majority of the snakes are preying either on rodents or other reptile species (Branch, 1998). Reptiles are also an important food source for a variety of raptors (Fargallo et al., 2009) and small carnivores (Skinner and Smithers, 1990). Second, reptiles form species rich animal communities in arid and semiarid regions (Pianka, 1986).

In this study, we compared plant diversity, reptile diversity and animal generated microhabitat structures (rodent burrows). Particularly for reptiles, rodent burrows play an important role as sheltering sites (Davidson et al., 2008), especially in open savanna habitats where natural refuges like, e.g. boulders are scarce (Davidson and Lightfoot, 2007).

## 2. Material and methods

### 2.1. Study period and area

The study was conducted from March to May 2006 and 2007 in the semiarid savanna of the southern Kalahari, Northern Cape Province, South Africa. As protected study area the South African side of the Kgalagadi Transfrontier Park was selected and the adjacent commercial farmland south of Twee Rivieren (main entrance of KTP) as the non-protected area (Fig 1).

The savanna habitat in the KTP is one of the oldest and most pristine protected areas in Africa. In the early 1930s, the South African Kalahari Gemsbok National Park (1931) and the adjacent Botswana Gemsbok National Park (1938) were established, which were unified to form the KTP since 1999. The total size of the KTP is 38,000 $km^2$ of which the South African side covers an area of almost 10,000 $km^2$.

The non-protected area lies in a mosaic of commercial livestock farms. Farm sizes differ from 2,000 ha to 35,000 ha and land use is dominated by mixed grazing of sheep, cattle and goats. Native antelopes (mainly springbok (*A. marsupialis* (Zimmermann)), gemsbok (*Oryx gazelle* (Blainville)), common duiker (*Sylvicapra grimmia* (Linneaus)), and steenbok (*Raphicerus campestris* (Thunberg)) occurred throughout the study area. The size of the non-protected study area was approximately 10,000 $km^2$, (from Twee Rivieren ~110 km southwards and ~90 km in West–East direction). For the field study, we selected 11 privately owned commercial livestock farms (Bloukranz, Branduin No. 61, Hoekrans, Inversnaid, Lena, Loch Lemond, Loch Maree, Rappels, Rooiduin, Swartpan and Vry Soutpan) with over 50 years of livestock farming history (Blaum et al., 2009a).

The savanna vegetation is classified as the Aoub dune veld, with a continuous layer of grasses (e.g. *Aristida meriodinalis* (Henrard), *Centropodia glauca* (Cope), *Eragrostis lehmanniana* (Nees), *Stipagrostis amabilis* (De Winter), *Stipagrostis ciliata* (De Winter)) and a layer of scattered woody vegetation including trees (e.g. *Acacia erioloba* (Edgew.), *Acacia haematoxylon* (Willd.) and *Boscia albitrunca* (Gilg-Ben.)) and shrubs (e.g. *Acacia mellifera* (Benth.), *Lycium hirsutum* (Dunal.) and *Rhigozum trichotomum* (Burch.)) (Mucina et al., 2005). Annual rainfall in the study area varies between 150 and 300 mm (Van Rooyen, 2001). The average rainfall for both study years for the unprotected area was

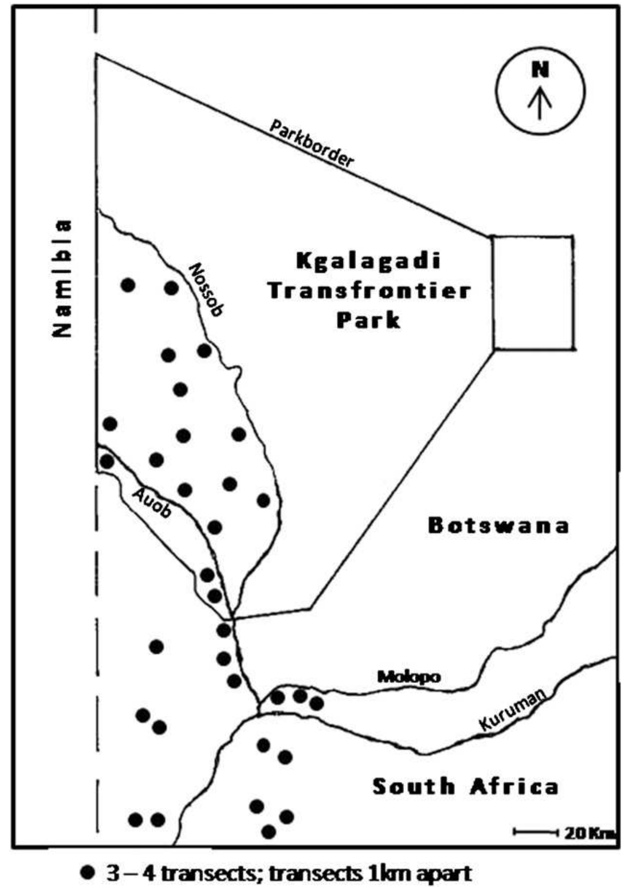


Fig. 1. Map of study sites in the protected KTP and in the non-protected farmland in the southern Kalahari, South Africa. Dots represent 3 to 4 transects with 1 km transect interspacing.

150.8 mm (South African Weather Service) and the non-protected area received 150.6 mm (Data: BIOTA Weather station Farm Alpha, southern Kalahari).

### 2.2. Vegetation survey

For both study years, vegetation cover and composition was determined along 55 transects for each focal area (protected KTP vs. non-protected farmland). Transects were selected randomly left or right from existing management tracks. Transects were placed between 1 and 4 km (randomly chosen) away either from junctions of management tracks and/or tourist roads (KTP) or from the farmhouse or farm entrance (farmland). At all chosen sites 3–4 transects were surveyed with a transect interspacing of 1 km. Surveys were conducted along transects (500 m long) across dunes and dune streets. For each transect, plant species richness was determined and percent ground cover (canopy cover) of trees (single-stemmed woody plants taller >3 m), shrubs (multi-stemmed woody plants, 1–3 m tall), dwarf shrubs (multi-stemmed woody or semi-woody plants, 0.3–1 m tall), perennial grasses, annual grasses and herbaceous vegetation estimated on 10 subplots (4 m × 4 m) with an intersubplot spacing of 50 m.

### 2.3. Reptile survey

Reptile diversity and abundance was estimated for both study areas and years along the same 500 m transects (55 transects per study area) that were used for the vegetation survey. We used time and area constrained Visual Encounter Surveys (VES) (Cooper and Whiting, 2000; Meik et al., 2002) to record the number of visible reptiles per species for each transect (500 m × 5 m). To avoid habitat disturbance we explicitly did not move any covering plant material like logs or leaf litter. To minimize observer bias all transects were observed by the same person. We also recorded the number of reptiles per species, which were sitting in or not more than 20 cm away from rodent burrows. Rodent burrows were counted along the first 100 m in a corridor of 10 m width of each transect.

### 2.4. Data analysis

We calculated mean plant species richness, total canopy vegetation cover (%), reptile species richness and abundance (no. of individuals). Additionally, we calculated mean vegetation cover (%) of the two listed shrub encroaching species in the area (*A. mellifera* and *R. trichotomum*) (Department of Agriculture, South Africa, 2011) and the numbers of rodent burrows to test for differences between the two areas. No significant differences were found between the two study periods for all data sets. Therefore, we pooled all transects of both research years for both study areas (protected and the unprotected area) for further analysis. To test for differences between the protected and the unprotected area the means of the respective data sets were compared. For all the statistical analysis we applied *t*-tests using SPSS Inc., Version 11.5. A *P*-value of <0.05 was regarded as statistically significant for all tests.

## 3. Results

### 3.1. Vegetation survey

An overview of plant species richness and vegetation cover of trees, shrubs, dwarf shrubs, perennial grasses, annual grasses and herbs are given in Tables 1 and 2. A detailed species list for protected and non-protected area is given in Appendix 1.

Plant species richness ($t_{54} = 20.87$; $p = 0.000$) and total vegetation cover ($t_{54} = 8.07$; $p = 0.000$) were higher in the protected KTP than in the non-protected farmland (Fig. 2a, b). The loss of plant species can be mainly attributed to a loss in herb species in the non-protected farmland (Table 1). The cover of the main shrub encroaching species, *A. mellifera* and *R. trichotomum* (KTP: 1.59%; farmland: 3.71%), were significant higher in the non-protected farmland compared to the protected KTP ($t_{54} = -4.05$; $p < 0.001$).

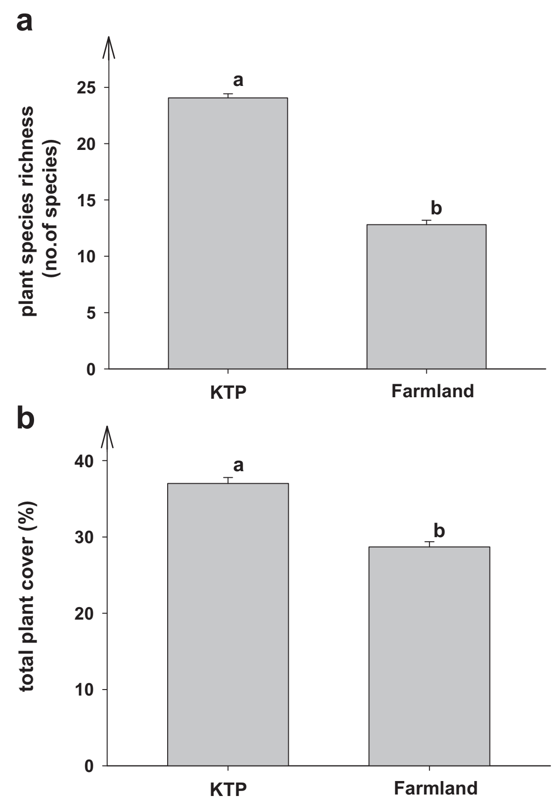


**Fig. 2.** Plant species richness (a) and total plant cover (b) per transect in the protected KTP and non-protected farmland. Bars show mean values ± SE. Bars with different letters (a, b) indicate significant differences between the two respective areas (plant species richness: $t_{54} = 20.87$; $p < 0.000$; total plant cover: $t_{54} = 8.07$; $p < 0.000$).

### 3.2. Reptile and rodent burrow survey

Table 3 shows the reptile species inventory for the KTP and the commercial farmland. Both, reptile abundance ($t_{54} = 10.15$; $p = 0.000$) and species richness ($t_{54} = 12.18$; $p = 0.000$) were significant higher in the KTP than on farmland (Fig. 3a, b). Also the number of rodent burrows was significantly higher in the KTP ($t_{54} = 7.69$; $p = 0.000$) (Table 4). Particularly, burrows of the Brant's whistling rat (*Parotomys brantsii* (A. Smith)) were very abundant in the KTP (nearly 90% of all rodent burrows) but not observed in the farmland (Table 4). Only in the KTP did we observed reptiles

**Table 1**
Species richness in no. of species (SE) of trees, shrubs, dwarf shrubs, perennial grass, annual grass, herbs and total plant species across all transects in the protected Kgalagadi Transfrontier Park (KTP) and the non-protected farmland averaged over the two study years (March–May 2006 & 2007).

| Location | No. of species | | | | | | |
|---|---|---|---|---|---|---|---|
| | Tree | Shrub | Dwarf shrub | Perennial grass | Annual grass | Herb | Total |
| KTP | 3 (0.0) | 6 (1.0) | 9.5 (0.5) | 10 (0.0) | 1 (0.0) | 41 (7.0) | 70.5 (8.5) |
| Farmland | 4 (0.0) | 3.5 (0.5) | 6 (1.0) | 8 (0.0) | 1 (0.0) | 10.5 (0.5) | 43 (1.0) |

**Table 2**
Mean vegetation cover per transect (SE) of trees, shrubs, dwarf shrubs, perennial grasses, annual grasses, herbs and total cover for the protected Kgalagadi Transfrontier Park (KTP) and the non-protected farmland averaged over the two study years (March–May 2006 & 2007).

| Location | Mean vegetation cover (%) | | | | | | |
|---|---|---|---|---|---|---|---|
| | Tree | Shrub | Dwarf shrub | Perennial grass | Annual grass | Herb | Total |
| KTP | 2.6 (0.27) | 1.98 (0.28) | 2.18 (0.2) | 18.72 (0.9) | 1.59 (0.18) | 10.67 (0.44) | 37.74 (0.77) |
| Farmland | 0.74 (0.17) | 3.84 (0.45) | 0.21 (0.05) | 16.11 (0.99) | 7.11 (0.55) | 2.73 (0.22) | 30.72 (0.69) |

**Table 3**
Reptile abundance (SE) for 11 species for all transects in the protected Kgalagadi Transfrontier Park (KTP) and non-protected farmland (PL, *Pedioplanis l. lineoocellata*; PN, *Pedioplanis namaquensis*; HL, *Heliobolus lugubris*; NT, *Nucras tessellata*; AA, *Agama aculeata*; MO, *Mabuya occidentalis*; MS, *Mabuya sparsa*; MV, *Mabuya variegate*; BA, *Bitis arietans*; DM, *Dipsina multimaculata*; NN, *Naja nivea*) averaged over the two study years (March–May 2006 & 2007).

| Location | Reptile species (no. of individuals) | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | PL | PN | HL | NT | AA | MO | MS | MV | BA | DM | NN |
| KTP | 96 (3.0) | 13.5 (0.5) | 23.5 (1.5) | 0 (0) | 98.5 (16.5) | 6.5 (4.5) | 62 (7.0) | 18 (1.0) | 0.5 (0.5) | 0.5 (0.5) | 1 (1.0) |
| Farmland | 67 (7.0) | 0 (0) | 2.5 (0.5) | 0.5 (0.5) | 0.5 (0.5) | 1.5 (1.0) | 12 (1.0) | 16.5 (4.5) | 0 (0) | 0 (0) | 0 (0) |

(*Agama aculeate* (Merrem.), *Mabuya sparsa* (Peters) and *Mabuya occidentalis* (Peters)) to use rodent burrows for sheltering (Table 4).

## 4. Discussion

In this study we compared plant diversity, reptile diversity and abundance of important microhabitat structures (rodent burrows) between savanna habitats in the protected Kgalagadi Transfrontier Park and the adjacent non-protected farmland. As expected, vegetation cover and species richness of perennial grasses, dwarf shrubs and herbaceous species was significantly higher in the protected KTP as wells as shrub species richness. In addition to the presence of by far more species in the protected KTP, were less areas affected by shrub encroachment (*A. mellifera* and *R. trichotomum*) a common problem in savannas caused by heavy grazing impact (e.g. Skarpe, 1990; Jeltsch et al., 1997). Also reptile species richness, abundance and microhabitat structures (rodent burrows) were significantly higher in the protected KTP.

Farmland sites showed typical signs of degradation caused by heavy livestock grazing: low plant and structural diversity (Blaum et al., 2007; Wasiolka et al., 2009; Wasiolka et al., 2010), low proportions of perennial grasses and herbaceous vegetation but high proportions of shrub cover (particularly *A. mellifera* and *R. trichotomum*) and bare ground (Wiegand et al., 2005). Indeed, the overall stocking rates for wild ungulates and domestic livestock (calculated as large stock units (LSU)/km$^2$, according to Dean and Macdonald (1994) in the farmland was more than four times higher than in the KTP (KTP: 0.95 LSU/km$^2$: SANParks 2004–2009; farmland: 4.16 LSU/km$^2$: Blaum et al., 2009a). The differences in feeding behaviour and forage plant preferences between livestock (sheep (dominant livestock), goat and cattle) and wild ungulates (mainly Gemsbok (*Oryx gazella*), Springbok (*A. marsupialis*), Red Hartebeest (*Alcelaphus buselaphus* (Pallas)) and Blue Wildebeest (*Connochates taurinus* (Burchell)) increase the negative effects of the higher stocking rates in the non-protected farmland. While sheep, the dominant livestock in the farmland, almost exclusively feed on perennial grass and herbaceous vegetation (Bothma, 2002), wild ungulates also browse woody vegetation. Additionally, the variety of wild ungulates in the protected KTP and their ability to utilize a wider range of plant species results in an overall well-balanced mixed foraging of grasses, herbs and woody plants (McGranahan, 2008) and are less likely to alter vegetation over the long term (Bothma, 2002). By regulating the balance between woody species, grass and herbs indigenous game plays an important role in savanna maintenance (Augustine and McNaughton, 2004) and consequently in preventing shrub encroachment (Hudak, 1999; McGranahan, 2008).

These large differences in vegetation characteristics between protected KTP and non-protected farmland affect reptile abundance and species richness negatively for three major reasons: (i) low prey availability, (ii) high predation risk and (iii) low availability of microhabitat sites for thermoregulation.

Recent studies in the southern Kalahari showed that heavy grazing induced changes in vegetation also significantly reduce arthropod abundance (e.g. Blaum et al., 2009b; Wasiolka et al., 2009). Arthropods are the major prey of the exclusively insectivorous lizards in the Kalahari (Pianka, 1986; Branch, 1998). Similarly,

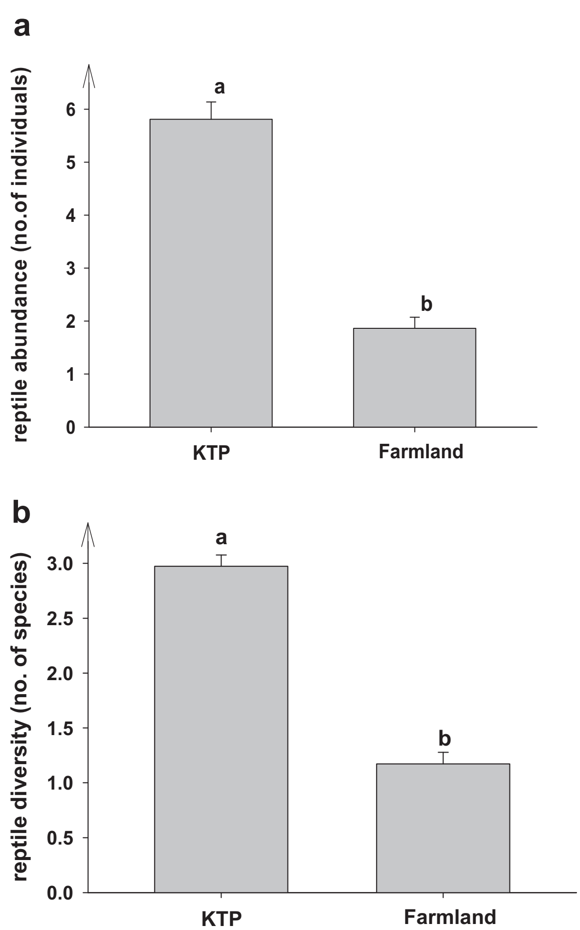


**Fig. 3.** Reptile abundance (a) and species richness (b) per transect in the protected KTP and non-protected farmland. Bars show mean values ± SE. Bars with different letters (a, b) indicate significant differences between the two respective areas (reptile abundance: $t_{54} = 10.15$; $p < 0.000$; Reptile species richness: $t_{54} = 12.18$; $p < 0.000$).

**Table 4**
Overview of mean numbers (SE) of rodent and *Paratomys brantsii* burrows and the use of those burrows by lizards (%) for the protected Kgalagadi Transfrontier Park (KTP) and the non-protected farmland for the 2007 study years (March–May 2006 & 2007).

| Location | Mean no. of rodent burrows | Mean no. of *Paratomys brantsii* burrows | Numbers (%) of lizards using *Parotomys brantsii* burrows | | |
|---|---|---|---|---|---|
| | | | *Agama acuelata* | *Mabuya occidentalis* | *Mabuya sparsa* |
| KTP | 56.87 (7.15) | 50.65 (6.59) | 46 (54.80) | 2 (18.20) | 10 (14.50) |
| Farmland | 4.07 (0.63) | 0 (0) | 0 (0) | 0 (0) | 0 (0) |

lizards and smaller snakes are an important dietary component of larger snakes (Marais, 2004) and hence, benefit from the high reptile abundance in the protected area.

Loss of vegetation cover, particularly perennial grass tussocks, which are used by reptiles as refuges (Norbury, 2001) increase predation risk (e.g. Bentley et al., 2000; Civantos, 2000). Indeed, the higher predation risk for reptiles in Kalahari rangelands with low vegetation cover is supported by the high amount of tail losses (used as a measurement of predation risk) in the Spotted Sand Lizard (*Pedioplanis l. lineoocellata* (Dumeril & Bibron)) in degraded habitats, characterized by a loss of overall plant cover, particularly perennial grass cover (Wasiolka, 2008).

The differences in vegetation composition and the lower vegetation cover in the non-protected farmland also reduced the availability of microhabitat sites for thermoregulation. In particular the proportion of perennial grass tussocks, where ground temperature maxima during daytime are lowest, were significantly lower in the farmland (ground temperature maxima below perennial grass: 43.2 °C, shrub: 48.4 °C, annual grass: 56.4 °C) (Wasiolka, unpublished data). The increase of shrubby vegetation at the expense of perennial grass tussocks in the farmland significantly intensifies environmental extremes reptiles have to cope with (e.g. Attum and Eason, 2006). Open areas in both the KTP and adjacent farmland provided sufficient opportunities for warming up particularly during the cold winter months where night temperatures often fall below 0 °C.

In contrast, in unprotected xeric succulent rangelands reptile abundance was higher compared to a neighboring protected nature reserve (Fabricius et al., 2003). The authors explained this difference with lower predation risk by birds, improved hunting conditions and higher availability of open areas for thermoregulation in the rangeland.

The high numbers of rodent burrows in our study provided an additional microhabitat structure for sheltering and thermoregulation of reptiles. Particularly for the Ground Agama (*Agama aculeata*), rodent burrows of the Brants' Whistling Rat (*P. brantsii*), which we only found within the park boundaries, played an important role for the presence of this reptile species. Similarly in North American grasslands, rodent burrows of Gunnison's Prairie Dogs (*Cynomys gunnisoni* (Baird)) and Banner-tailed Kangaroo Rats (*Dipodomys spectabilis* (Gray)), were used by reptiles for sheltering and thermoregulation (i.e. avoiding extreme low and high day temperatures) (Davis and Theimer, 2003; Davidson et al., 2008). Burrows of the Brants Whistling Rat were also frequently used by *Mabuya occidentalis* and *Mabuya sparsa*. This was surprising for *Mabuya sparsa*, which is an arboreal skink, typically found on trees, larger shrubs or in the vicinity of those (Branch, 1998; Cooper and Whiting, 2000).

Therefore, *P. brantsii* is modifying the habitat by offering additionally save sites for reptiles and, by doing so, they are changing reptile behaviour and likely increasing their overall abundance and diversity. Hence, *P. brantsii* can be regarded as an ecosystem engineer. Ecosystem engineers are playing an important role in modifying, maintaining and creating the environment for other species (Jones et al., 1994; Wright and Jones, 2006) in a variety of ecosystems and in the process alter habitat and community structures. Rodents act as ecosystem engineers by providing microhabitat structures for reptiles (e.g. Shipley and Reading, 2006; Davidson et al., 2008) but also for a large number of other animal species (e.g. Bangert and Slobodchikoff, 2006; Davidson and Lightfoot, 2007).

Tree density largely affected the abundance of arboreal reptiles. *Mabuya sparsa* was significantly higher in the protected KTP, where tree density was higher than in the farmland. Tree density on farmland was very low except for the dry river bed habitats. While the harvesting of Camelthorn trees (*A. erioloba* and *A. haematoxylon*) for firewood production for sale or subsistence (Tews et al., 2006) is a common practice in Kalahari rangelands, wood harvesting is prohibited in the protected KTP.

To conclude, our results clearly show that long-term effects of different land use options (wildlife in protected KTP versus extensive livestock production in the non-protected farmland) even within the same habitat type have led to significant changes in vegetation composition, availability of microhabitat structures and in the reptile community. While lizard communities can also benefit from land use, e.g. clear cutting of forests (Greenberg et al., 1981), the higher species richness and abundance of reptiles and the higher species richness and cover of vegetation in the protected KTP in contrast to the adjacent commercial farmland strongly support the relevance of protected areas for biodiversity conservation. Despite the invaluable relevance of protected areas for biodiversity conservation, we believe that the explicit consideration of socio-economic studies in non-protected savanna areas may identify alternative management options aiming to meet both provision of human livelihoods and the conservation of biodiversity.

## Acknowledgements

We would like to thank all the farmers of our study area as without their permission and support the study would have been impossible. Baie dankie. We thank the Northern Cape Nature Conservation Service, South Africa who made this project possible by granting permission to work in the Southern Kalahari (Permit No. 0014/06 & 0134/2007). Furthermore, we thank South African National Parks (Project No. 2006-01-17BWAS) for permission to work in the Kgalagadi Transfrontier Park. This project was supported by the German Ministry of Education and Research (BMBF) within the BIOITA Project Southern Africa (01LC0024I).

## Supplementary data

Supplementary data associated with this article can be found, in the online version, at doi:10.1016/j.jaridenv.2011.04.011.

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