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Source: Journal of Vertebrate Biology, 69(2)

Published By: Institute of Vertebrate Biology, Czech Academy of Sciences

URL: https://doi.org/10.25225/jvb.20001

Journal of Vertebrate Biology

Open Access

J. Vertebr. Biol. 2020, 69(2): 20001 DOI: 10.25225/jvb.20001

SPECIAL ISSUE: AFRICAN SMALL MAMMALS

Effects of prescribed burning on rodent community ecology in Serengeti National Park

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▸ Received 4 January 2020; Accepted 30 August 2020; Published online 13 November 2020

Abstract. A study on the effects of prescribed burning on rodent community ecology was conducted in Serengeti National Park, Tanzania. The study aimed at generating ecological knowledge about the changes in rodent communities when areas of the park are intentionally burned to regulate grasslands or reduce undergrowth that can lead to uncontrolled forest fires. A completely randomized design (CRD) factorial layout with two treatments (burned and unburned) and two replications was applied. A total of 148 animals comprising six species of rodent and one insectivore were captured over 2,940 trap nights. Among the trapped individuals, 41.9% were adults, 16.1% juveniles and 41.9% sub-adults. Males and females were at parity between treatments. Species abundance was estimated using the minimum number alive (MNA) method for different rodent species and was found to vary with treatment where *Mastomys natalensis* declined in burned plots whilst *Arvicanthis niloticus* increased. However, species diversity did not differ across treatments ($F_{1, 10}$ = 0.15, p = 0.70). Differences in the reproductive condition of female *M. natalensis* (z = 4.408, df = 15, p < 0.001) and *A. niloticus* (z = 2.381, df = 15, p = 0.017) were observed between treatments showing that higher numbers of reproductively active females were observed in burned plots in March, whilst in unburned plots more were observed from November to February. Conservation strategies involving periodic habitat burning should, therefore, consider small mammal reproductive periods to ensure that species potentially at risk are not adversely affected and able to rapidly recover from the effects of burning in temporarily lowering food resources and longer term impacts of increased predation caused by reduced cover.

Key words: *Mastomys natalensis*, *Arvicanthis niloticus*, population, breeding pattern, age structure, recruitment

Introduction

Fire has been one of the most common disturbances to animals in most ecosystems for millennia (Maishanu et al. 2017, Jones et al. 2019). It has occurred naturally for millions of years since the early development of terrestrial ecosystems (Bond & Keane 2017). Currently, fire results from anthropogenic factors (Strauch & Eby 2012, Maishanu et al. 2017) as well as natural agents such as volcanoes, earthquakes, lightning and sparks from rock falls (Maishanu et al. 2017). Wildfires cause changes in vegetation type and diversity resulting in fundamental changes to ecosystems

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and the fauna which live there (Msindai 2014, Green et al. 2015). In order to prevent widespread uncontrolled fires, prescribed controlled burning of small areas has been adopted (Johnson & Hale 2002, Block et al. 2016). This method of control is used throughout Africa in national parks (Owen 1971, Strauch & Eby 2012) to improve and maintain grassland foraging conditions for herbivores, facilitate tourism (Maishanu et al. 2017), reduce fuel load, stimulate the growth of new plant species, reduce competition between fire adapted species (Adams et al. 2013) and control animal parasites such as ticks and tsetse flies (Hassan et al. 2007). However, not all prescribed burning is advantageous in protected areas as it can prevent natural ecosystem progression and can sometimes lead to the uncontrolled fires it aims to prevent (Strauch & Eby 2012). Poorly planned burning may create conditions that threaten human life and property and may kill wild animal species, particularly keystone small mammal species (SENAPA 2010). Such impacts create a cascade of ecological changes, from species to ecosystem level.

Rodents are important keystone species in many ecosystems, accounting for 40% of mammalian species in the world (Chekol et al. 2012, Bantihun & Bekele 2015). Rodents act as predators, seed dispersers, pests, pollinators and primary grazers (Magige & Senzota 2006, Mueller 2019) as well as serving as prey for predatory birds, mammals and reptiles (Senzota 1990, Hassan et al. 2007). Burning vegetative cover affects rodent populations through the direct impacts of heat and gases (Engstrom 2010), and indirectly through changes in vegetation (Bowman et al. 2017), resulting in loss of cover and food (Bantihun & Bekele 2015).

Most research on the effects of prescribed burning in the Serengeti has been on vegetation (Hassan 2011) and large mammals (Hassan et al. 2007). Despite being one of the more susceptible animal groups to fire in savannah ecosystems, rodent populations require further study to understand potential variations in diversity and abundance in response to controlled burning programmes and particularly to protect endangered species (Bowman et al. 2017). Therefore, the objectives of our research were to understand potential changes in small mammal community composition, abundance, diversity, population fluctuation and breeding between burned and unburned areas in Serengeti National Park.

Material and Methods

Description of the study area

The study was conducted at Alokole plains in Serengeti National Park (Fig. 1). The area is characterized by grassland (SENAPA 2010) and woodlands (Byrom et al. 2014). The park covers an area of 14,763 km^2 (SENAPA 2010) and lies between 1°28′-3°17′ S, 33°50′-35°20′ E (Timbuka & Kabigumila 2006), with an altitude ranging from 920 to 1,850 m (SENAPA 2010), while mean annual temperature varies from 13-28 °C (SENAPA 2010). The park is surrounded by several protected areas including Ngorongoro Conservation Area, Maswa Game Reserve, Kijereshi Game Controlled Area, Speke Gulf Game Controlled Area, Ikorongo-Grumeti Game Reserves and Loliondo Game Controlled Area in Tanzania, and Maasai-Mara National Reserve in Kenya (Kideghesho 2010, Msindai 2014). The location of this park makes it the heart and cornerstone of the Serengeti-Mara Ecosystem (Msindai 2014) by supporting diverse populations of birds, herbivores, carnivores and small mammals (Byrom et al. 2014, Msindai 2014). Permission to carry out the research was granted by Tanzania National Parks, permit number TNP/HQ/E.20/07C. Ethics permission and clearance was granted by Sokoine University's ethics board (ref SUA/ADM/R.1/8/229).

Experimental design

A trial using mark-recapture live trapping grids was established following a completely randomized design (CRD) with factorial layout, two treatments (burned and unburned) and two replications. Each field plot contained a trapping grid of 60 × 60 m (approximately one acre). All four grids were set in October 2018 and the burnt grids were burned in mid-November 2018. To prevent the fire spreading to untargeted areas a firebreak was made around the grids before burning. All grids were more than 300 m apart to avoid potential interaction of rodents between grids as it is known that most rodent species occupy a home range of 200 to 2,000 m^2 in wild grassland areas of eastern Africa (Mulungu et al. 2015).

Trapping procedure and data collection

Permanent trapping using the mark-recapture technique was conducted with a total of 196 Sherman live traps (23 × 9.5 × 8 cm, H.B. Sherman Traps Inc.). Each grid consisted of seven parallel lines of Sherman live traps located 10 m apart. The traps were arranged in seven trapping stations per

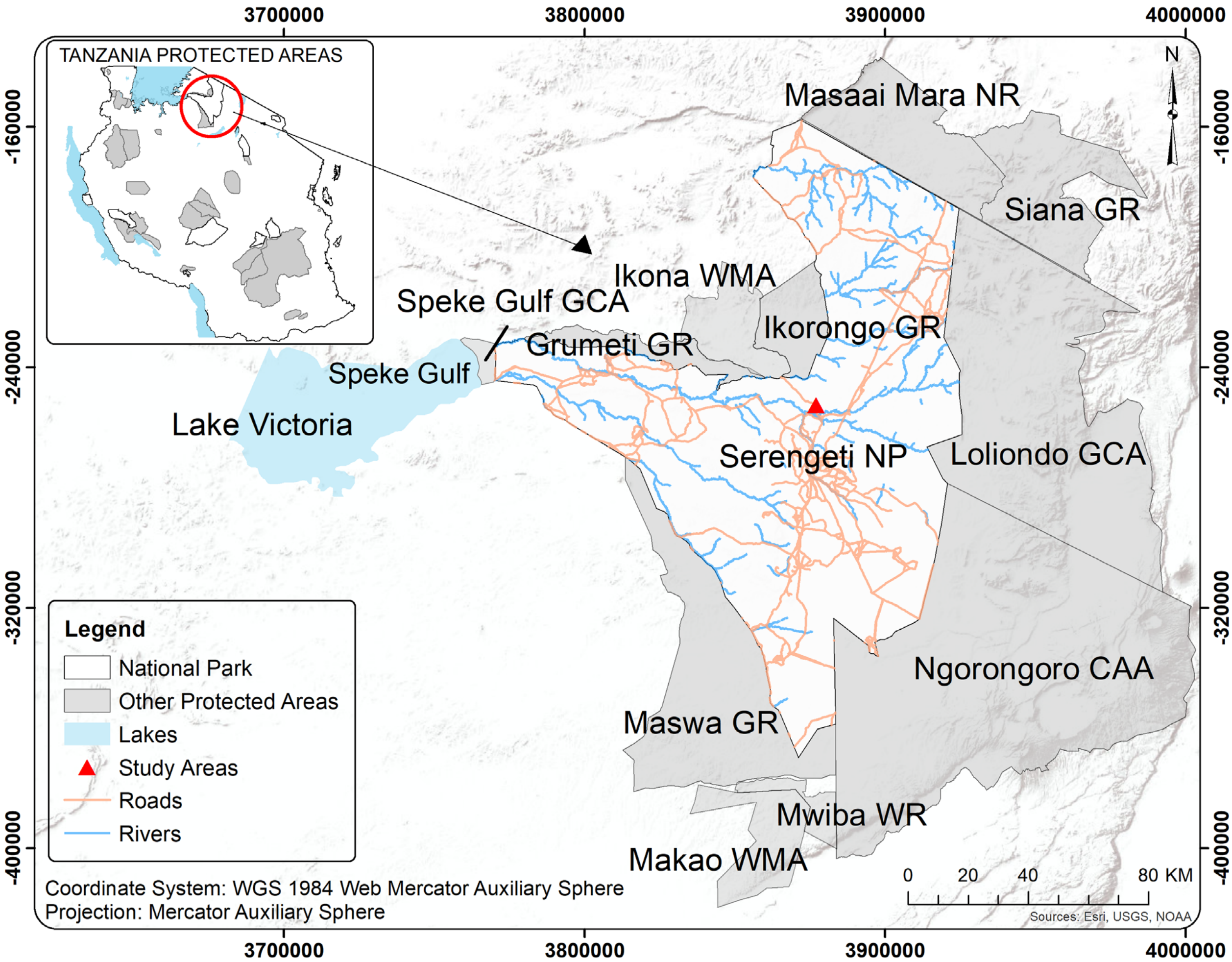

Fig. 1. Study area location within the Serengeti National Park, Tanzania.

line each 10 m apart (making a total of 49 trapping stations per grid). Traps were set in the evening of the first day and were baited with a mixture of peanut butter and maize flour and were checked and re-baited early in the morning and late evening in order to capture both nocturnal and diurnal species for three consecutive nights per month (Senzota 1982, Mulungu et al. 2008, Magige 2016). Initial trapping took place before burning (October 2018) and was repeated monthly from November (one week after burning) to March 2019, with the exception of January where no trapping was possible due to operational issues.

Small mammals specimens captured were identified to species through morphological characteristics and the known distribution range (Happold et al. 2013, Kingdon 2015). All newly captured animals were marked by toe clipping using a unique number code to enable identification during subsequent trap checks and released at the site of capture. Cover and burrows at or near the point of capture were noted for use as release points after data collection. The data were recorded by treatment type (burned and unburned) and grid location, collecting standard body measurements, weight, sex and reproductive condition (either a perforated or closed vulva, pregnant and/or lactating in females and scrotal or non-scrotal testes in males).

Data analysis

Species composition was obtained by calculating the proportion (%) of various rodent species in relation to the total species found in each trapping grid. Age structure was determined for *Mastomys natalensis*, the only species that could be categorized into juveniles, sub-adults or adults, following the relationship between age and body weight (Leirs & Verheyen 1995). Individuals weighing > 24 g were classed as adults, between 21 and 24 g as sub-adults, while those weighing ≤ 20 g were categorised as juveniles. As observed variance was greater than the mean, a Generalised Linear Model (GLM) was used to determine the influence of habitat type (burned *vs.* unburned), and time (months) on the abundance of the different age groups of *Mastomys natalensis*. The Least Significant Difference test ($LSD_{0.05}$) was used to compare the unburned and burned treatments and time of trapping (months).

Pearson's Chi-squared test was used to compare the expected and observed sex ratios between burned and unburned habitats. Sex ratio was determined as the ratio of the number of females to the total captured for each species (Jennions & Fromhage 2017), with the formula:

$$r = \frac{f}{m + f}$$

where r = gender (sex) ratio, m = number of males, f = number of females.

In this study recruitment was defined as the number of new individuals (first-captures) captured in each habitat and month. In analysis, we assumed that potential differences in detectability between old captures and new captures were consistent between treatments and time, and therefore, the comparison of proportion of captures between treatments would be unbiased. Data were analysed using a Generalized Linear Model with a Poisson distribution to determine the influence of habitat type and month on the abundance of newly captured rodents.

The minimum number alive (MNA) index was used to estimate true abundance in each treatment. MNA in mark-recapture is defined as the number of individuals caught in that capture session in each habitat and those that were caught both previously and subsequently (Krebs 1966). MNA is used to estimate the population of rodents in a small number of trapping occasions and individuals due to the use of information from prior and subsequent capturing sessions (Pocock et al. 2004). To determine effects of burning on rodent abundance, GLM tests were used to determine the influence of treatment, and month on the abundance of different species.

Diversity was calculated using the Simpson index and the Shannon index (Shannon & Weaver 1949). Both of these indices are a function of the proportion of individuals found in each species. The Simpson Diversity Index (λ = 1 – D) (Jiang et al. 2017) is a measure of diversity which takes into account the number of species present, as well as the relative abundance of each species giving weight to dominant species (Magige 2013). The value of the index ranges from 0 (low species diversity) to 1.0 (high species diversity). Simpson's Diversity Index (1 – D) is defined as:

$$\lambda = 1 - \left(\frac{N\,(N-1)}{\sum n\,(n-1)} \right)$$

where n = the total number of organisms of each individual species, N = the total number of organisms of all species.

The Shannon index is defined as:

$$H^1 = -\sum_{i=1}^{R} p_i \ln pi$$

where p_i is the proportion of the observations found in category i.

The two diversity tests were used to determine if there were differences between unburned and burned treatments using a one way ANOVA and t-test.

Breeding patterns were determined by establishing the percentage of reproductively active females according to treatment and month (Mlyashimbi et al. 2018). A GLM with reproductive condition as the response variable and time (month) and treatment (burned or unburned) as explanatory variables was performed assuming a logit-link function with Poisson distribution. This approach was used because the reproductive condition of the animals did not follow a linear pattern over the entire period. All analysis was performed using the program Paleontological Statistics (PAST) 9.1.3 Service Pack 4 XP_PRO platform (Hammer et al. 2001) and R Version 3.5.1 (Zuur & Ieno 2016).

Results

Community structure and composition

A total of six species and six genera belonging to three families of Rodentia and non-rodent species (*Crocidura* spp.) were identified over 2,940 trap nights (Table 1). Five species (*Mastomys natalensis*, *Arvicanthis niloticus*, *Mus* spp., *Aethomys* spp. and *Crocidura* spp.) were present in burned and unburned plots while *Graphiurus* sp. was observed only in burned plots and *Steatomys parvus* only in unburned plots (Table 1). Species composition and individual species capture rates varied considerably. In burned areas *M. natalensis* was the dominant species comprising 41.4% of all individuals, while in unburned areas *A. niloticus* was the dominant species caught accounting for 63.1% (Table 1).

Table 1. Species composition of small mammals in unburned (plot 1 & 2) and burned (plot 3 & 4) fields in the study area.

S/N	Species	Plot 1	Plot 2	Plot 3	Plot 4	Unburned	%contribution	Burned	%contribution	Total	%contribution
	Rodents										
1	*Mastomys natalensis* (Smith, 1834)	4	8	24	28	12	21.82	52	55.91	64	43.24
2	*Arvicanthis niloticus* (Geoffrey, 1803)	7	25	1	6	32	58.18	7	7.53	39	26.35
3	*Aethomys* spp. (Thomas, 1915)	1	2	3	20	3	5.45	23	24.73	26	17.57
4	*Mus* spp. (Clerck, 1757)	3	1	7	2	4	7.27	9	9.68	13	8.78
5	*Steatomys parvus* (Rhoads, 1896)	1	0	0	0	1	1.82	0	0.00	1	0.68
6	*Graphiurus* spp. (Smuts, 1832)	0	0	0	1	0	0.00	1	1.08	1	0.68
	Non rodent										
7	*Crocidura* spp. (Wagler, 1832)	1	2	0	1	3	5.45	1	1.08	4	2.70
	Total	17	38	35	58	55	100	93	100	148	100

Age structure

In all trapping periods, more adults (41.9%) and sub-adults (41.9%) were captured than juveniles (16.1%) in the two treatments. Only in November were significant differences ($z = 2.49$, $df = 15$, $p = 0.013$) observed in the number of sub-adults captured (Fig. 2).

Sex ratio

The percentage of females in unburned fields was particularly high in November and December for both *A. niloticus* and *M. natalensis* and particularly low in March. The sex ratio of *M. natalensis* showed no significant difference between the two treatments ($\chi^2 = 0.44$, $df = 1$, $p = 0.508$); however, differences were observed across the months ($\chi^2 = 7.52$, $df = 1$, $p = 0.006$). The sex ratio of *A. niloticus* did not differ from the expected 1:1 between treatments ($\chi^2 = 0.43$, $df = 1$, $p = 0.513$).

Recruitment

A total of 77 new captures were observed (unburned N = 39 and burned N = 38) (Table 2). Analysis indicated a significant difference in the total number of new captures between treatments (burned and unburned) and time (month) in March ($z = 2.13$, $df = 69$, $p = 0.033$) and October ($z = 1.99$, $df = 69$, $p = 0.047$), whilst no differences were observed over November, December and February ($p > 0.05$). Similarly, significant differences at the species level between treatments were observed for *M. natalensis* ($z = 2.60$, $df = 21$, $p = 0.009$), but all other species did not differ significantly ($p > 0.05$).

Abundance

At the start of the trial in October before burning, there were no differences detected in abundance between the four plots ($p > 0.05$). *M. natalensis*, *A. niloticus*, *Mus* spp. and *Aethomys* spp. were the abundant species in all habitats. In the burned area, *M. natalensis* was the most abundant rodent species ($z = 3.56$, $df = 23$, $p < 0.001$), while *A. niloticus* was the most abundant rodent species in the unburned habitat ($z = 4.633$, $df = 23$, $p < 0.001$). *Steatomys parvus* (n = 1) was recorded from the unburned habitat and *Graphiurus* sp. (n = 1) was recorded from burned habitat and both were the least captured species during the trial (Table 1).

Diversity

Species richness was similar in both the burned ($\lambda = 0.8$) and unburned treatments ($\lambda = 0.79$) (Table 3). Simpson's diversity index for all trapping periods did not differ between burned and

Table 2. Rodent new individual capture per species between burned and unburned fields.

Months/Treatment		October				November				December				February				March				Total	
		Unburned		Burned		Unburned		Burned		Unburned		Burned		Unburned		Burned		Unburned		Burned		Burned	Unburned
Species/ Replication	Recruitment	P1	P2	P3	P4	P1	P2	P3	P4	P1	P2	P3	P4	P1	P2	P3	P4	P1	P2	P3	P4		
Mastomys natalensis	New capture	0	4	2	6	1	0	4	0	0	0	0	0	0	0	0	2	1	0	0	0	14	6
	Recapture	0	4	3	4	2	0	6	5	0	0	5	4	0	0	1	4	0	0	3	3	38	6
Arvicanthis niloticus	New capture	0	1	1	2	1	1	0	0	2	3	0	0	2	5	0	1	0	9	0	2	6	24
	Recapture	0	0	0	0	0	0	0	0	0	1	0	0	2	1	0	0	0	4	0	1	1	8
Aethomys spp.	New capture	0	1	0	2	0	0	1	2	1	0	0	1	0	0	0	1	0	1	0	2	9	3
	Recapture	0	0	0	0	0	0	0	3	0	0	1	5	0	0	0	2	0	0	1	2	14	0
Mus spp.	New capture	0	1	2	0	0	0	0	0	0	0	2	0	2	0	0	0	0	0	3	2	9	3
	Recapture	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
Steatomys parvus	New capture	0	0	0	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	0	1
	Recapture	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Graphiurus spp.	New capture	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1	1	0
	Recapture	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Total	New capture	0	7	5	10	2	1	5	2	3	3	2	1	5	5	0	4	1	10	3	7	39	37
	Recapture	0	4	3	4	2	0	6	8	0	1	6	9	3	1	1	6	0	4	4	6	53	15

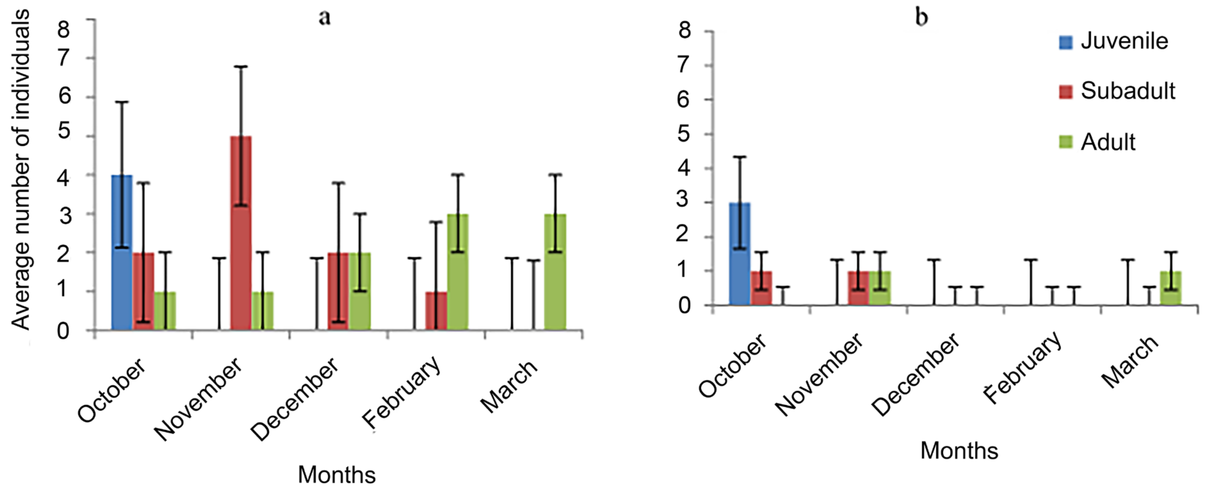

Fig. 2. Relative differences in the average (±SD) number of juvenile, sub-adult and adult *Mastomys natalensis* in (a) burned and (b) unburned habitats.

unburned plots ($F_{1,\,10} = 0.15$, $p = 0.700$), nor was there a difference in the Shannon's indices between the treatments ($t = 1.10$, df = 79, $p = 0.270$).

Population fluctuation

Variations in the population were observed on both treatments for *M. natalensis* and *A. niloticus*. For *M. natalensis*, the population peaked in October and November then declined in subsequent months possibly as a result of colonization in burned plots. *Aethomys* spp. numbers increased for two months after burning (in November) then decreased in the following months, while *Mus* spp. disappeared after burning (Table 1). The population in the unburned area peaked in February and March probably due to local recruitment and colonization by *A. niloticus* (Table 1). Temporal variations in population were observed between habitats and months with a decrease of *M. natalensis* for every month in both burned and unburned areas while *A. niloticus* individuals increased every month in the unburned area (Fig. 3). The effects on the *A. niloticus* population were observed after burning

Table 3. Rodent species abundance and diversity in burned and unburned fields.

Habitats	Burned	Unburned
Richness (absolute number of species)	6	6
Number of individuals	93	55
Simpson diversity index	0.8	0.79
Shannon diversity index	1.39	1.195

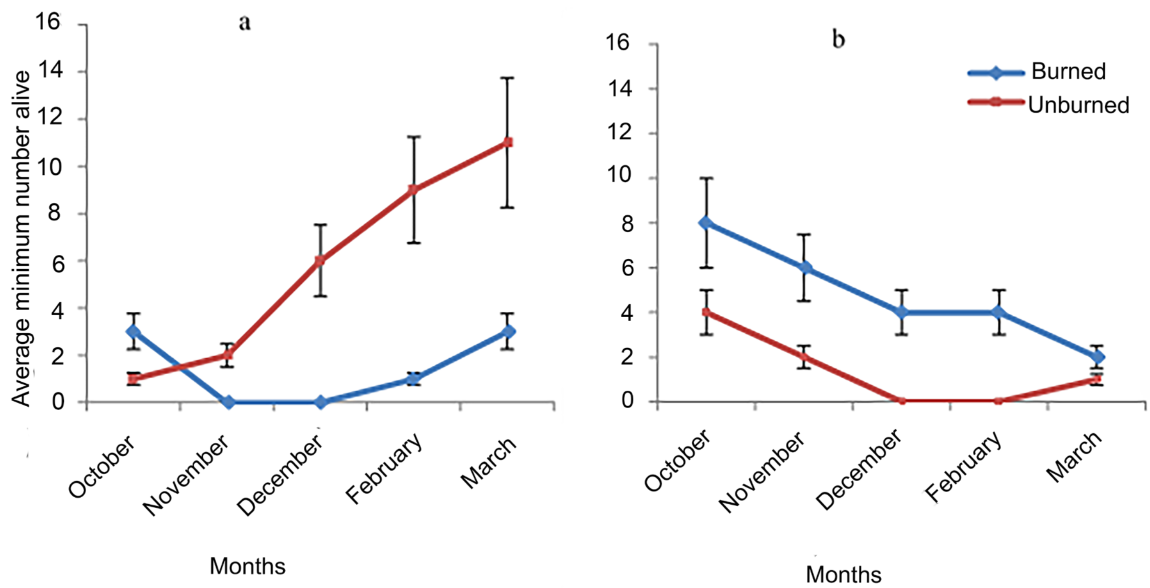

Fig. 3. Average (±SE) population fluctuation of: a) *Arvicanthis niloticus* and b) *Mastomys natalensis* between burned and unburned fields.

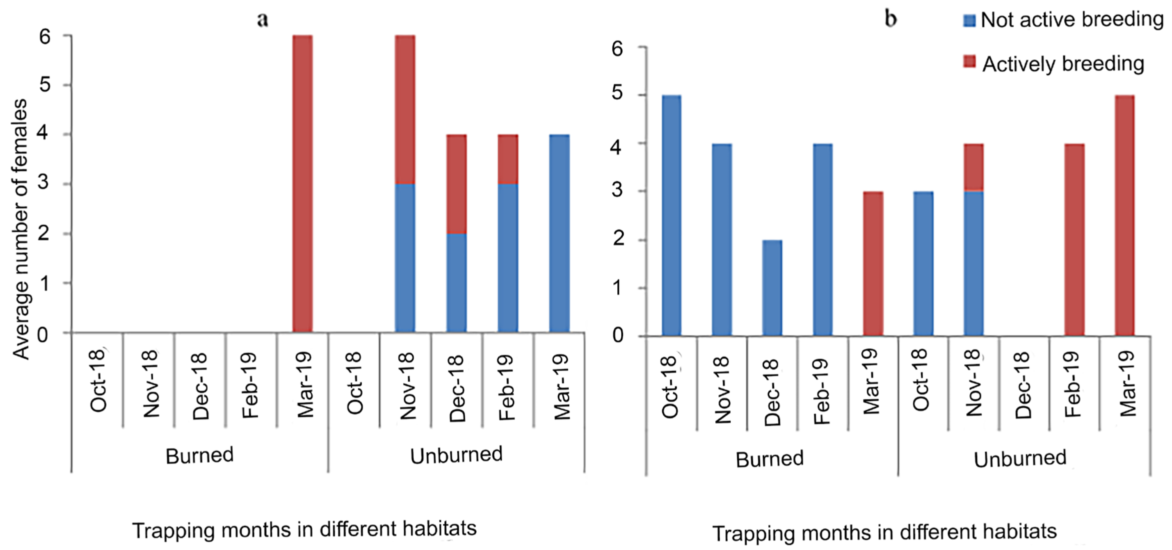

Fig. 4. Breeding patterns of female a) *Mastomys natalensis* and b) *Arvicanthis niloticus* in burned and unburned fields.

in February (z = 0.99, df = 4, p = 0.023) to March (z = 1.74, df = 4, p = 0.042). However, no effects of burning were observed for *M. natalensis* at any point in the study period.

Breeding patterns

The breeding activity of *M. natalensis* females in unburned fields was highest in February to March and lowest in November. In the burned area, females were only sexually active in March (Fig. 4a). The reproductive activity of female *M. natalensis* was different between burned and unburned areas (z = 4.41, df = 15, p < 0.001). Reproductive activity also differed between treatments for female *A. niloticus* (z = 2.38, df = 15, p = 0.017), but for this species, peaks in number of reproductively active females in burned fields were observed in March while in the unburned area reproductive activity was observed from November to February (Fig. 4b).

Discussion

The current study provides information on the short-term impact of burning savannah grasslands on small mammal community composition. The high number of *M. natalensis* in burned areas was probably due to their opportunistic behaviour in disturbed habitats. Studies on food use patterns (Mulungu et al. 2011) and population dynamics (Massawe et al. 2006, Mayamba et al. 2019) show that *M. natalensis* utilises different food types according to their abundance and/or availability in disturbed habitats. The dominance of *A. niloticus* in the unburned area was probably due to sufficient ground cover and its need to feed on grass seeds and grass leaves. Senzota (1982) observed ground cover to be important for survival by shielding *A. niloticus* from predators, which is particularly important for diurnal species.

The results of this work show some changes in the age classes of the rodent population. Burning and predation are probably the most important causes of the change of age structure in the area. Monadjem & Perrin (2003) and Byrom et al. (2014) observed that the change from vegetated to bare land may attract predators into the area. The rodents may generally live in constant fear of capture by predators when living under unfavourable conditions caused by fire. In addition, this study observed an increase in the number of adults in the burned area. This may be due to their larger home range, more active movement, and higher social ranking as identified by Assefa & Srinivasulu (2019). A decrease of sub-adults in the burned fields could be argued to be due to decreased vegetation and predation. The same results have been demonstrated by Leahy et al. (2016) and MacFadyen et al. (2012) that indicated higher predation on sub-adults.

The sex ratio between treatments was at parity, indicating equal movement of males and females in search of food and mating. This has been demonstrated in other studies showing that differences in behaviour and predation pressure are common characteristics in regulating balance between the sexes in different animal species (Greenberg et al. 2006, Zwolak & Foresman 2008, Mulungu et al. 2013, Borremans et al. 2014). New captures were higher than recaptures in both treatments. This could have been due to the high mobility of animals in search of favourable areas. An increase in new vegetative regrowth in burned treatments appeared to attract new rodents. Kennis et al. (2008) and Borremans et al. (2014) suggested that good cover and green vegetative material is required for rodent movement and nourishment. Species recolonization after burning was confirmed by early capture of *Aethomys* spp. and *M. natalensis* as new growth of plants is likely to have attracted these species. It has previously been reported that the rapid recovery of rodent numbers in a burned area appears to be correlated with the fast regrowth and redevelopment of the ground cover in slash or longleaf habitat (MacFadyen et al. 2012, Bowman et al. 2017).

Earlier studies have captured up to 36 rodent species in Serengeti National Park (Timbuka & Kabigumila 2006, Byrom et al. 2014), which is a much higher species richness compared to this study which only recorded six rodent species. This difference is likely due to study design and wider habitat selection. For example, Magige & Senzota (2006) and Magige (2013) concentrated on the human-wildlife interface and altitudinal gradient, where migratory rodent species and those that are non-migratory such as *Acomys* spp. and the roof rat (*Rattus rattus*) could be trapped while searching for food and shelter from adverse weather conditions. Some rodent species have been reported to be opportunistic and habitat specific, for example, *R. rattus* flourishes in areas inhabited by humans and *Acomys* spp. in rocky outcrops (Timbuka & Kabigumila 2006) and thus could not be trapped in our study. In other regions where similar studies have been conducted, 4 to 10 species of rodent have been reported (Fitzgerald et al. 2001, Bowman et al. 2017), which aligns with the current study.

The current study shows higher abundance of *M. natalensis* in the burned than the unburned plots over the study period. This finding contrasts with other studies, which show lower numbers in the burned plots than in neighbouring unburned plots (Bowman et al. 2017), as we observed with *A. niloticus*. This might be an indication that *M. natalensis* tends to colonize new or disturbed area (Leirs & Verheyen 1995, Makundi et al. 2007). The decreased abundance of *A. niloticus* in burned plots probably reflects the inability of a diurnal species dependent on specific plants for food and shelter to sustain its population (Senzota 1982, Dejene & Reddy 2016). The indirect effects of burning mediated by changes in the plant community were expected to have impacts on the rodent population. Bowman et al. (2017) detected the effects on rodent populations over four months post-burn, and no differences were detected soon after fire. The population change in small mammals after burning suggests that small mammals might play a role in the slow appearance and disappearance of their dependent animals (predators) in the life cycle of the Serengeti-Mara ecosystem. The increased abundance of *Aethomys* spp. after fire may be a response to an increase in early greenness with improvement in food quality and quantity thus contributing to species diversity.

The equality in species diversity between the burned and unburned treatments indicates that both habitats had adequate resources (i.e. vegetation) to support a variety of species. Vegetation is known to support different rodent species in several habitats as it provides macro- and microhabitats in addition to food (Cramer & Willig 2002, Byrom et al. 2014). This differs from other studies which indicated a low biological diversity of rodents in burned plots as compared to unburned plots (Bowman et al. 2017) in parks (i.e. undisturbed area) as compared to disturbed areas (Magige & Senzota 2006, Timbuka & Kabigumila 2006, Magige 2016) which could be due to frequent disturbances such as ecological disturbances and human activities including agriculture, grazing, grass cutting and land clearance which interfere with the ecological niches of rodents (Senzota 1982, Magige 2016).

The present study indicates that almost all females were reproductively active in March in burned and unburned habitats, indicating that there is no relationship between burning and reproductive activity of females as the onset of breeding fell within the normal rainy season. It has been reported that the onset of the breeding season is accelerated by food availability (Duque et al. 2005), and breeding or reproductive activity might cease when a catastrophe interferes with resource availability and that the effect could be detectable after a certain time. Thus reproductive activity observed in March was probably caused by the strong relationship between rainfall and quick vegetative growth after burning (Senzota 1982, Magige & Senzota 2006, Magige 2013) that is linked to the reproductive activity of female rodents (Makundi et al. 2007, Mulungu et al. 2016).

Conclusion

Species composition of rodents was maintained in burned study sites. However, relative abundance was influenced, *M. natalensis* being dominant in burned habitats and *A. niloticus* dominant in unburned habitats. In terms of community structure, the number of sub-adults decreased in burned habitats compared to unburned areas. Sex ratio was at parity in both burned and unburned habitats. New individuals trapped in burned areas were probably attracted by the new vegetation. The effect of burning did not influence the breeding patterns of rodents during the peak of female reproductive activity at the start of long rains in March. Conservation strategies involving periodic habitat burning should consider small mammal reproductive periods and baseline species diversity to ensure that potentially at-risk species are not adversely affected and able to rapidly recover from the effects of burning on temporarily lowering food resources and longer term impacts of increased predation caused by reduced cover.

Acknowledgements

We thank the ACE II-IRPM and BTD project, for funding this study. Special thanks go to Dr. R. Fyumagwa, Director of the Serengeti Wildlife Research Centre and Mr. W. Mwakilema, Chief Park Warden Serengeti National Park (SENAPA) for granting permission and facilitating research, and particularly in organizing transport field trips to various areas in the SENAPA. We appreciate the assistance of R.I. Kigunguli, K.M. Senso, R. Kichembu, S.K. Mkamango, M. Maki and R. Nchakwi (TAWIRI driver) for their tireless work and tolerance during data collection. Author contributions: A.M. Manyonyi, S.B. Mariki, and L.S. Mulungu conceived the ideas, designed the methodology and performed data collection. A.M. Manyonyi, S.B. Mariki, L.L. Mnyone and S.R. Belmain analysed the data. All authors contributed to writing the manuscript, critically reviewed drafts and gave final approval for publication.

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