

SUPPORTING INFORMATION

Drought and fire determine juvenile and adult woody diversity and dominance in a semi-arid African savanna

METHODS

3. Data Analyses

The rankabundance function from the BiodiversityR package in R, version 4.0.2 (RStudio-Team, 2016) was used to calculate rank-abundance curves for each fire treatment – year combination from the field data. Rank-abundance curves indicate relative dominance of species within a community and can help visual change in abundances over time. Woody species diversity in each treatment was calculated as relative species richness, with the diversity result function, also from BiodiversityR.

The major advantage of *gllvm* over other analytic methods for multivariate abundance data was computational speed while maintaining a high level of accuracy by using the automatic differentiation software C++. Additionally, starting values for the GLMs of each response are carefully chosen by the model approach, based on the Dunn-Smyth residuals which represent the optimum parameter distribution of the data. More detail on *gllvm* can be found in Niku et al. (2019). These models fit a generalized linear model (GLM) to the abundance of each woody species and calculate species-specific intercepts and regression coefficients in relation to predictors. Models then interpret relationships between the whole community and predictors to identify both variation in species composition and indicator species responses. Spatial autocorrelation was not included due to the close vicinity of all

plots in a relatively homogenous environment as part of the original experimental design. The response variables of species abundances were quantitative and discrete. We assumed that the response variables (y) were independent from each other but were conditional of the predictor values (x). We further assumed that the distribution and the mean-variance relationships of y was known and that there was a linear relationship between the mean of y and every x .

Using an offset term in the GLLVM approach to predict changes in species abundances relative to the entire community cannot consider rare species which may be missed due to uneven sampling across the three fire treatments. We therefore used another approach to run the GLLVMs but without an offset term: We subset the abundance dataset by bootstrapping the data. Bootstrapping means that the data was resampled randomly by a desired number of subsamples from the entire dataset which then replaced the original data (Efron and Tibshirani, 1994, Efron, 2003). Because we had uneven sampling size across all three fire treatments, the abundance data followed a negative binomial distribution, and we needed to find a confidence interval of 95% for the difference of the three sample means (Manly, 1992, Kulesa et al., 2015, Efron, 1982), we tested between 100 – 10,000 replications in parametric bootstrapping if counts of adult and juvenile species' abundance data per fire treatment across years of measurement could be corrected to a normal distribution. Test results showed that at least 1,000 replications were necessary to come closest to a normal distribution (Figure S4). We decided to use 10,000 replications for more robust replication of the entire population (Efron, 1982). Thus we randomly resampled the abundance data 20x per fire treatment and year of measurement with 10,000 replications, and thereby we considered the minimum number of 20 sampled points (on the no burn treatment) and eliminated the uneven sample size of the original data while assuming a near normal distribution. Bootstrapping was carried out in R, version 4.0.2 (RStudio-Team, 2016). The resampled data was then used in the GLLVM equation, but without an offset term:

$$\text{GLM}(i,j,n) = \text{gllvm}(y(i,j,n) \sim x_1 * x_2 + \text{LV} + (1|\text{plot}|\text{subplot}|\text{point}))$$

where i = size class (adults, juveniles), j = species, y = woody species abundance data, x_1 = fire treatment (FB, MB, NB), x_2 = year of sampling (2016, 2018, 2019), n = number of points sampled in each fire treatment and LV = latent variable. The response variables were counts of individuals among woody plant species (= abundance), with year of sampling and fire treatment as categorical predictor variables. One latent variable (LV) was still incorporated into model calculations to test for intra-species correlations. The random effect $(1|\text{plot}|\text{subplot}|\text{point})$ accounted for the nested design and repeated sampling in 2016, 2018, and 2019. Similar to the GLLVMs with the offset term, we ran seven models each for adults and juveniles which were ranked using the Akaike Information Criterion (AIC). Other than excluding the offset term, the data was analysed and evaluated the same way as described under 3. Data Analysis in the methods of the main article.

RESULTS

1. Adult abundance and diversity

The abundance of *Dichrostachys cinerea*, declined from 2016 to 2018 over the entire area of 1.26-ha from 322 (+/- 4.8) individuals (= 55.5%) to 61 (+/- 1.9) individuals (= 18.8%), and remained here in 2019 (= 17.9%) (Table 1, Table S1, Figure 2). The second most abundant species in 2016 was *Acacia nigrescens* with 83 (+/- 0.6) individuals (= 14.3%) that declined to 60 (+/- 0.3) individuals in 2018 (= 18.5%), and 2019 (60 +/- 0.3) individuals (= 17.6%) (Table 1, Table S1, Figure 2). The third most abundant species in 2016 was *Combretum apiculatum* with 74 (+/- 0.8) individuals (= 12.8%), which decreased to 61 (+/- 0.6) individuals in 2018 (= 18.8%), and then to 60 (+/- 0.6) individuals (= 17.6%) in 2019 (Table 1, Table S1, Figure 2). These three species remained the most abundant throughout the study period accounting for at

least 52.5% of total adult plants (Table 1, Table S1, Figure 2). Absolute abundance of the three most dominant species (*Dichrostachys cinerea*, *Combretum apiculatum*, *Acacia nigrescens*) decreased from 2016 to 2019 (Table 1, Table S1), although abundances of less dominant species or rare species increased or remained the same (Table 1, Table S1).

In adults and across all three fire treatments, diversity decreased from 2016 to 2018, but then increased again in 2019, while diversity was overall highest on unburned plots.

2. Juvenile abundance and diversity

Dichrostachys cinerea was the most abundant species across all treatments and over time, and many juvenile individuals were counted as resprouts at stem bases in 2019 and were possibly classified as adults in previous years of measurement (Figure S1). Abundance of *Dichrostachys cinerea* decreased in individuals from 1439 (+/- 6.4) individuals (= 42.6%) in 2016 to 1051 (+/- 3.9) individuals (= 55.1%) in 2018, and then decreased to 998 (+/- 4.8) individuals (= 45.7%) in 2019 (Table 2, Table S1, Figure 2). The second most abundant species in 2016 was *Acacia exuvialis* with 449 (+/- 3.8) individuals (= 13.3%), which decreased to 240 (+/- 1.5) individuals (= 12.6%) in 2018, and then increased slightly to 245 (+/- 1.9) individuals (= 9.2%) in 2019 (Table 2, Table S1, Figure 2). The third most abundant species in 2016 was *Peltophorum africanum* with 332 (+/- 1.9) individuals (= 9.8%), which decreased to 34 (+/- 0.3) individuals (= 1.8%) in 2018, and to 28 (+/- 0.2) in 2019 (Table 2, Table S1, Figure 2). Species that appeared in 2018 were: *Ozoroa engleri*, *Commiphora glandulosa*, *Gymnosporia senegalensis*, *Combretum molle*, *Combretum mossambicense*, *Euclea natalensis*, *Dalbergia melanoxylon*, and *Ormocarpum trichocarpum* (Table 2, Table S1, Figure 2). Species that disappeared in 2019 were: *Rhus gueinzii*, *Commiphora glandulosa*, *Combretum molle*, *Euclea divinorum*, and *Euclea natalensis* (Table 2, Table S1, Figure 2). Species that appeared in 2019

were: *Ehretia amoena*, *Grewia bicolor*, *Ximenia caffra*, and *Flueggea virosa* (Table 2, Table S1, Figure 2). Overall, total juvenile abundance decreased with drought, from 3378 individuals in 2016 to 2185 individuals in 2019 (65% decrease).

Species richness increased on frequently burned plots from 2.4 (+/- 1.3) in 2016 to 2.6 (+/- 1.3) in 2018 and further to 2.7 (+/- 1.3) in 2019 (7% increase from 2016 to 2019) (Figure 3). On moderately burned plots, species richness decreased from 3.1 (+/- 2) in 2016 to 2.8 (+/- 1.4) in 2018, and then decreased slightly to 2.7 (+/- 1.5) in 2019 (13% decrease from 2016 to 2019) (Figure 3). On unburned plots, species richness increased from 1.9 (+/- 1.4) in 2016 to 2.1 (+/- 1.4) in 2018 and further increased to 2.8 (+/- 1.6) in 2019 (32% increase) (Figure 3). Additionally, 7 of 12 of the new juvenile species found in 2018 or 2019, which increased diversity, were not of the same family (Table 2, Table S1).

3. Patterns of demography between years of measurement and on fire treatments when using bootstrapped data

Similar to models run with the original data and an offset term, for adults and juveniles, AIC model testing results showed that the model with fire treatment and sampling year with additive effect was the best model to fit the data, as it had the lowest AIC value (adults: AIC = 11092.13, juveniles: AIC = 20942.28), and no other model had a delta AIC of < 2 (Figure S3c and d). The models explained 52% variance for adults and 32% for juveniles. Similar to models run with original data and an offset term, there was a consistent effect of year of sampling on woody plant species abundances across fire treatments.

Relative changes of each species' abundance proportional to the entire woody species community were similar to results from models run with original data and an offset term, where abundances of adult *Combretum mossambicense* and *Gymnosporia senegalensis* were

the two species with the highest increase in abundances from 2016 to 2018, and were still increasing in 2019, together with *Flueggea virosa*, *Ehretia amoena*, *Grewia bicolor*, and *Ormocarpum trichocarpum* (Figure S3a). Abundances of these species were also increasing in juveniles from 2016 to 2019 (Figure S3b).

Adult species abundance relative to moderately burned plots increased only for *Lannea schwinfurthii*, *Ozoroa engleri* and *Sclerocarya birrea*, while *Ehretia amoena* had the highest increase in abundance on frequently burned plots (Figure S3a). Decreases in adult abundances were highest in *Cassia abbreviata* on unburned plots, and in *Combretum mossambicense* and *Peltophorum africanum* on frequently burned plots (Figure S3a). For juveniles, only *Commiphora glandulosa* increased on both unburned and frequently burned plots (Figure S3b). Decreases in abundances were highest for *Combretum molle* on unburned plots, and *C. mossambicense*, *Combretum hereroense* and *C. molle* on frequently burned plots (Figure S3b).

Across both adults and juveniles, *Ximenia caffra* was not randomly sampled via bootstrapping and thus missed when analysing relative changes of each species abundance proportional to the entire woody community.

TABLES

Table S1: Overview of changes in species abundance of adults (AT) and juveniles (JUV) over time (2016 – 2019) on each fire treatment (FB = frequently burned, MB = moderately burned, NB = no burn/unburnt).

Family	Species	Change in abundance over time per fire treatment										
		Fire treatment	FB			MB			NB			
		Year	2016	2018	2019	2016	2018	2019	2016	2018	2019	
Anacardiaceae	<i>Lannea schweinfurthii</i>	AT	0	0	0	0	2	2	0	0	0	
		JUV	2	1	1	56	11	10	1	0	1	
		AT	0	0	0	0	1	1	0	0	0	
		JUV	0	1	0	0	5	4	0	0	0	
	<i>Ozoroa engleri</i>	AT	0	0	0	0	0	0	0	0	0	
		JUV	0	1	0	0	5	4	0	0	0	
		AT	0	0	0	0	0	0	0	0	0	
		JUV	2	0	0	10	1	0	0	0	0	
	<i>Rhus gweinzii</i>	AT	0	0	0	0	0	0	0	0	0	
		JUV	2	0	0	10	1	0	0	0	0	
		AT	0	0	0	0	0	0	0	0	0	
		JUV	8	13	18	87	2	8	4	4	3	
<i>Sclerocarya birrea</i>	AT	0	0	0	1	0	0	0	0	0		
	JUV	8	13	18	87	2	8	4	4	3		
	AT	0	0	0	0	0	0	0	0	0		
	JUV	0	0	138	0	0	72	0	0	35		
Boraginaceae	<i>Ehretia amoena</i>	AT	0	0	0	0	0	0	0	0	0	
		JUV	0	0	138	0	0	72	0	0	35	
		AT	0	0	1	4	2	3	3	1	1	
Burseraceae	<i>Commiphora africana</i>	JUV	0	24	21	4	0	23	29	55	46	
		AT	0	1	0	0	0	0	1	1	0	
		JUV	0	0	0	0	1	0	0	3	0	
Celastraceae	<i>Gymnosporia senegalensis</i>	AT	0	6	2	0	6	3	0	9	7	
		JUV	0	43	22	0	24	14	0	17	15	
		AT	0	0	0	0	0	0	0	0	0	
Combretaceae	<i>Combretum apiculatum</i>	JUV	91	16	24	110	29	22	2	4	23	
		AT	13	9	11	13	12	11	48	40	38	
		JUV	111	5	8	48	4	0	0	0	0	
		AT	8	8	7	2	6	4	1	0	0	
	<i>Combretum hereroense</i>	JUV	61	13	2	54	1	2	1	1	1	
		AT	7	10	13	2	10	5	4	4	5	
		JUV	0	0	0	0	0	0	0	0	0	
		AT	0	0	0	0	0	0	0	0	0	
	<i>Combretum imberbe</i>	JUV	0	16	0	0	0	0	0	0	0	
		AT	0	0	0	0	0	0	0	0	0	
		JUV	0	0	0	0	0	0	0	0	0	
		AT	0	0	0	0	8	9	0	13	14	
	<i>Combretum molle</i>	JUV	0	11	39	0	43	80	0	0	0	
		AT	0	0	0	0	0	0	0	0	0	
		JUV	0	0	0	0	0	0	0	0	0	
		AT	3	1	1	3	3	3	0	0	0	
	<i>Combretum mossambicense</i>	JUV	12	0	0	11	0	11	0	0	0	
		AT	3	1	1	3	3	3	0	0	0	
		JUV	2	0	0	0	1	0	0	0	0	
		AT	0	0	0	0	0	0	0	0	0	
	Ebenaceae	<i>Euclea divinorum</i>	JUV	2	0	0	0	1	0	0	0	0
			AT	0	0	0	0	0	0	0	0	0
			JUV	0	0	0	0	0	0	0	0	0
			AT	0	0	0	0	0	0	0	0	0
<i>Euclea natalensis</i>	JUV	0	1	0	0	0	0	0	0	0		
	AT	0	0	0	0	0	0	0	0	0		
	JUV	0	0	0	0	0	0	0	0	0		
	AT	0	0	0	0	0	0	0	0	0		

Family	Species	Change in abundance over time per fire treatment									
		Fire treatment	FB			MB			NB		
		Year	2016	2018	2019	2016	2018	2019	2016	2018	2019
Fabaceae	<i>Acacia exuvialis</i>	AT	6	1	5	3	0	1	3	1	2
		JUV	155	118	102	249	114	71	45	8	28
		AT	56	44	43	14	8	7	13	8	8
		JUV	150	47	50	125	32	22	11	12	30
	<i>Acacia nigrescens</i>	AT	3	3	1	2	3	3	3	5	4
		JUV	26	6	3	17	3	0	3	1	0
		AT	2	4	4	0	0	0	2	4	5
		JUV	40	2	3	0	4	0	19	0	4
	<i>Dalbergia melanoxylon</i>	AT	0	0	0	0	0	0	0	0	0
		JUV	0	0	14	0	2	7	0	0	0
		AT	45	8	9	219	6	32	58	43	20
		JUV	700	326	232	644	611	627	95	114	139
<i>Ormocarpum trichocarpum</i>	AT	0	2	3	0	0	0	0	0	1	
	JUV	0	27	30	0	48	63	0	7	4	
	AT	10	3	4	13	2	2	0	0	0	
	JUV	85	23	25	246	11	3	1	0	0	
<i>Philenoptera violacea</i>	AT	3	5	7	0	13	11	0	0	1	
	JUV	4	11	6	0	22	10	0	0	15	
	AT	0	0	2	0	0	0	0	0	1	
	JUV	0	0	12	0	0	2	0	0	0	
Malvaceae	<i>Grewia bicolor</i>	AT	0	0	0	0	0	0	0	1	
Olacaceaea	<i>Ximenia caffra</i>	AT	0	0	0	0	0	1	0	1	
		JUV	0	0	7	0	0	16	0	0	
		AT	0	0	6	0	0	2	0	0	
Phyllanthaceae	<i>Flueggea virosa</i>	JUV	0	0	10	0	3	0	0		
		AT	3	1	1	0	1	2	8	6	
		JUV	40	2	0	17	4	0	0	4	
Rhamnaceae	<i>Ziziphus mucronata</i>	JUV	40	2	0	17	4	0	0	4	

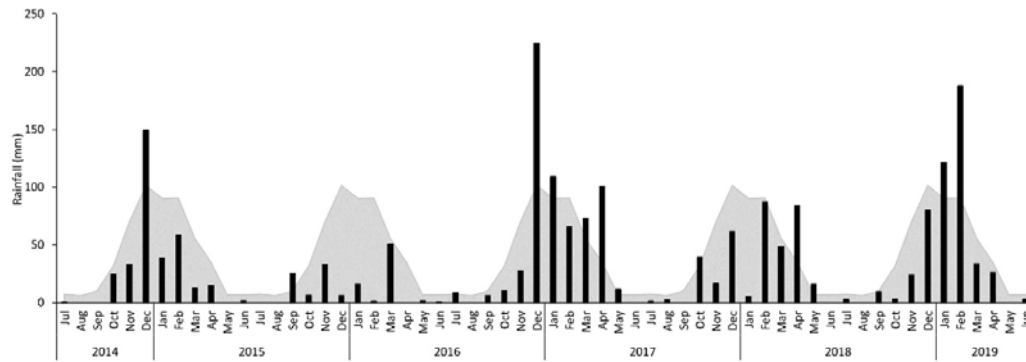
Table S2: Overview of effect sizes and standard deviations (sd) of change of adult woody abundance relative to frequently burned plots on moderately burned plots (MB) and unburned plots (NB), and relative to the year 2016 in 2018 and 2019. The table is ordered by family.

Family	Species	FireMB	sd	FireNB	sd	Year2018	sd	Year2019	sd
Anacardiaceae	<i>Lannea schweinfurthii</i>	13.65	0.26	-2.87	0	11.76	0.66	11.83	0.66
	<i>Ozoroa engleri</i>	12.61	0.37	-2.9	1.06	10.52	0.65	10.59	0.65
	<i>Rhus gueinzii</i>	-1.33	0	-1.4	0	-1.79	0	-1.79	0
	<i>Sclerocarya birrea</i>	13.62	0	-2.62	0	-11.64	0	-11.54	0
Boraginaceae	<i>Ehretia amoena</i>	-3.65	1.09	15.86	1.17	-3.94	0.8	14.82	0.72
Burseraceae	<i>Commiphora africana</i>	2.52	0.56	2.83	0	-1.05	0.31	-0.4	0.31
	<i>Commiphora glandulosa</i>	-16.9	0	1.98	0	0.65	0	-12.93	0
Celastraceae	<i>Gymnosporia senegalensis</i>	0.42	0	2.14	0.5	21.91	158.48	21.18	1.43
Combretaceae	<i>Combretum apiculatum</i>	0.4	0	2.61	1.24	-0.39	0	-0.3	0.36
	<i>Combretum hereroense</i>	-0.34	0.41	-1.87	1.06	0	0.47	-0.16	0.49
	<i>Combretum imberbe</i>	-0.28	1.06	0.45	1.43	0.44	0.51	0.46	0.58
	<i>Combretum molle</i>	-1.33	0.51	-1.4	0	-1.79	0	-1.79	0
	<i>Combretum mossambicense</i>	0.71	0	-16.54	1.96	19.67	0.82	19.84	0.84
	<i>Terminalia prunioides</i>	0.97	0.98	-13.08	0	-0.77	0	-0.7	0.27
Ebenaceae	<i>Euclea divinorum</i>	-1.33	0	-1.4	0.2	-1.79	0	-1.79	0.2
	<i>Euclea natalensis</i>	-1.33	0.02	-1.4	0.01	-1.79	0.02	-1.79	0.02
Fabaceae	<i>Acacia exuvialis</i>	-0.8	0.26	0.59	0.34	-2.02	0.3	-0.55	0.28
	<i>Acacia nigrescens</i>	-1.29	0.3	-0.35	0.31	-0.53	0.3	-0.5	0.31
	<i>Acacia tortilis</i>	0.45	0.32	1.82	0.36	0.08	0.36	-0.2	0.37
	<i>Cassia abbreviata</i>	-16.18	0.76	1.37	0.57	0.6	0.62	0.76	0.59
	<i>Dalbergia melanoxylon</i>	-1.33	0.04	-1.4	0.02	-1.79	0.04	-1.79	0.04
	<i>Dichrostachys cinerea</i>	1.49	0	2.49	1.23	-2.28	1.24	-1.89	0
	<i>Ormocarpum trichocarpum</i>	-14.77	0	-0.53	0.55	12.37	0.72	13.24	0.71
	<i>Peltophorum africanum</i>	0.29	0.24	-16.97	0.27	-1.78	0.23	-1.51	0.23
	<i>Philenoptera violacea</i>	0.7	0.59	-1.43	0.61	1.47	0.6	1.63	0.63
Malvaceae	<i>Grewia bicolor</i>	-12.19	0.29	0.61	0	-3.11	0.73	15.41	0.73
Olcaceae	<i>Ximenia caffra</i>	-2.91	0.67	16.83	0.68	-10.13	0.84	0.05	0.58
Phyllanthaceae	<i>Flueggea virosa</i>	-0.86	0.68	-13.75	0	-2.74	0.81	16	0.81
Rhamnaceae	<i>Ziziphus mucronata</i>	-0.19	0.41	2.75	0	-0.44	0.54	-0.04	0.51

Table S3: Overview of effect sizes and standard deviations (sd) of change of juvenile woody abundance relative to frequently burned plots on moderately burned plots (MB) and unburned plots (NB), and relative to the year 2016 in 2018 and 2019. The table is ordered by the family.

Family	Species	FireMB	sd	FireNB	sd	Year2018	sd	Year2019	sd
Anacardiaceae	<i>Lannea schweinfurthii</i>	3.1	0.82	0.55	0	-1.64	0.91	-1.35	0.98
	<i>Ozoroa engleri</i>	2.5	0.52	-5.88	0.75	10.46	316.1	9.93	85.23
	<i>Rhus gweinzii</i>	2.21	0.17	-7.06	0.24	-2.89	0.2	-12.74	0.2
	<i>Sclerocarya birrea</i>	0.49	2	-0.02	81.04	-1.55	359.96	-1.11	1.99
Boraginaceae	<i>Ehretia amoena</i>	-0.42	0.53	-0.07	193	-2.74	275.8	14.27	275.8
Burseraeae	<i>Commiphora africana</i>	-0.2	1.31	2.71	97.06	1.17	526.63	1.84	83.39
	<i>Commiphora glandulosa</i>	7.93	0.36	10.11	0.54	8.72	0.43	-3.95	0.45
Celastraceae	<i>Gymnosporia senegalensis</i>	-0.29	122.31	0.68	123.63	17.02	80.03	16.54	461.2
Combretaceae	<i>Combretum apiculatum</i>	0.58	1.05	0.29	1.68	-1.61	1.1	-1.12	1.59
	<i>Combretum hereroense</i>	-0.81	0.49	-17.41	0.65	-3.07	0.56	-3.44	0.52
	<i>Combretum imberbe</i>	-0.37	0.64	-1.31	0.77	-2.28	0.74	-3.19	0.71
	<i>Combretum molle</i>	-12.13	0.94	-11.07	1.34	11.36	0.47	-3.56	0.5
	<i>Combretum mossambicense</i>	1.29	43.12	-13.03	39.37	14.81	59.35	15.85	322.93
	<i>Terminalia prunioides</i>	2.47	0.56	-9.72	0.8	-16.27	0.75	-2.44	0.77
Ebenaceae	<i>Euclea divinorum</i>	-0.27	0.24	-7.27	0.33	-0.91	0.26	-9.18	0.27
	<i>Euclea natalensis</i>	-7.27	0.55	-6.01	1.01	7.99	0.59	-3.44	0.68
Fabaceae	<i>Acacia exuvialis</i>	0.42	2.71	-0.33	42.29	-0.64	29.34	-0.72	29.37
	<i>Acacia nigrescens</i>	-0.15	1.58	0.02	1.49	-1.33	1.68	-1.15	1.28
	<i>Acacia tortilis</i>	-0.47	0.53	-1.01	0.81	-1.71	0.53	-2.9	0.72
	<i>Cassia abbreviata</i>	-1.18	0.78	0.95	1.21	-1.52	0.81	-2.32	0.82
	<i>Dalbergia melanoxylon</i>	1.28	1.23	-8.39	37.15	6.83	86	10.03	86
	<i>Dichrostachys cinerea</i>	0.84	1.24	0.16	49.34	-0.63	1.24	-0.65	64.11
	<i>Ormocarpum trichocarpum</i>	0.91	46.35	-0.34	46.35	14.36	40.68	14.54	292.93
	<i>Peltophorum africanum</i>	-0.19	0.63	-4.49	1.48	-2.65	0.7	-2.8	0.77
	<i>Philenoptera violacea</i>	0.41	0.75	0.72	90.06	1.81	335.7	1.67	68.23
Malvaceae	<i>Grewia bicolor</i>	-1.55	1.1	-10.32	43.94	-3.86	1.28	11.37	144.6
Olacaceae	<i>Ximenia caffra</i>	1.07	0.45	-9.29	0.7	-3.94	164.11	12.21	164.11
Phyllanthaceae	<i>Flueggea virosa</i>	-0.96	0.91	-10	75.4	-3.36	5.53	10.88	103.17
Rhamnaceae	<i>Ziziphus mucronata</i>	0.16	0.31	2.42	0.44	-2.41	0.35	-4.3	0.36

FIGURES



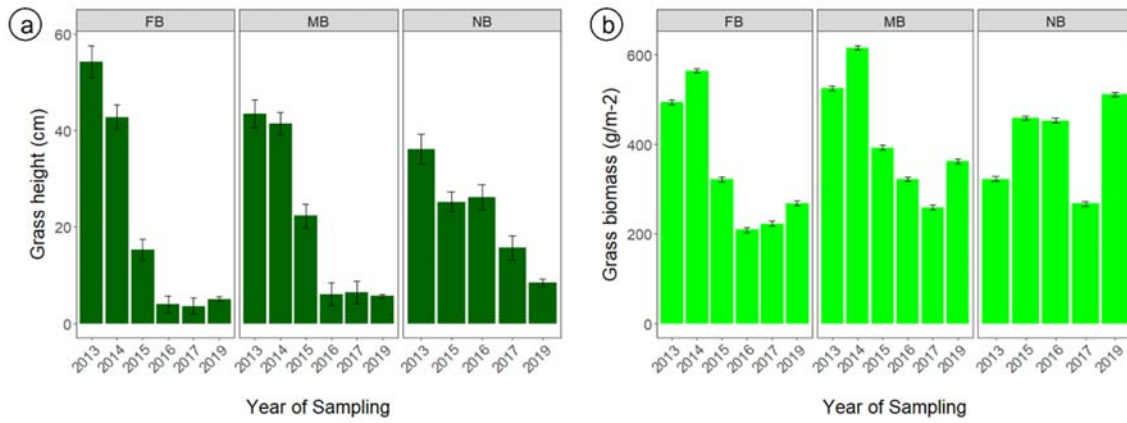


Figure S2: Changes in grass height (a) and grass biomass (b) at the end of the wet season (April) across fire treatments and years of measurement. Grass data was sampled from 2013 – 2017, and again in 2019. No data was sampled in 2018 because of project termination. No data was sampled in 2014 on unburnt plots due to time constraints.

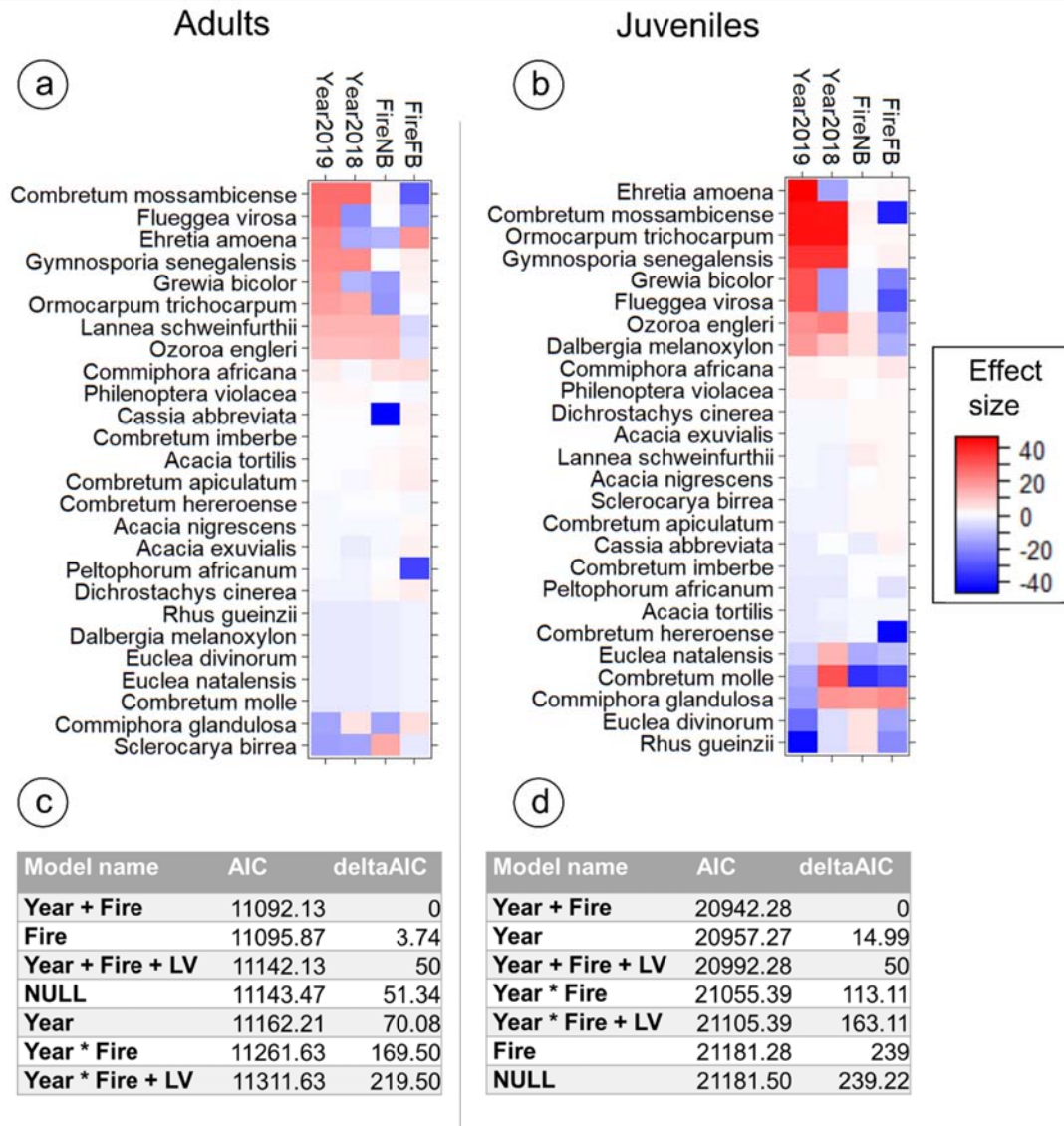


Figure S3: Effect sizes (ES) representing the change in abundance per woody species proportional to entire community and in relation to year of measurement and fire treatment (a, b), estimated with multivariate generalised linear latent variable models (GLLVM) models that used year and fire as predictor variables and were evaluated with Akaike information criterion (AIC) tests (c, d). Other than in the main article, the ES presented here are based on bootstrapped abundance data from 20 randomly resampled points per fire treatment and year of measurement and 10,000 replications. The calculation of ESs assumed 2016 and moderately burned/grazed (MB) treatment as a reference. Thus, increasing ESs show increasing abundances when compared to 2016, and FB respectively. Red fields show

increases, blue fields depict decreases in abundance. Plots are ordered from increasing (high ESs) to decreasing abundances (low ESs) in reference to the year 2019. The tables (c, d) show an overview of GLLVM test results from the AIC for optimal model selection for adults (c) and juveniles (d). The best model fit is indicated by the lowest AIC value, or delta AIC = 0, respectively. The response variables used in the GLMs were adult and juvenile abundances (number of individuals per species in each sampling point) respectively. Predictor variables were year of sampling (equal to cumulative rainfall values of two years prior to year of sampling); fire treatment applied to sampling point (frequently burned (FB), moderate burn (MB), and no burn (NB)) and one latent variable (LV) = latent variable accounting for factor of herbivory. Random effects of nested samples were assumed across sampled years and included in the models.

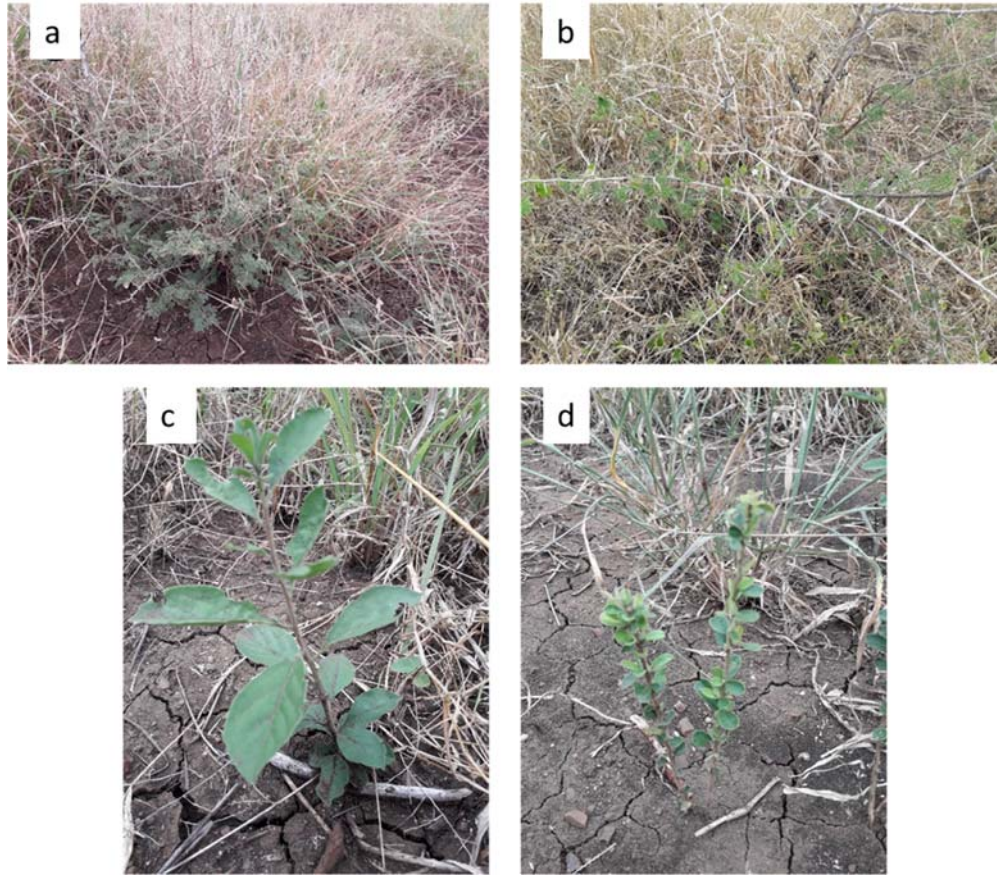


Figure S4: Resprouting of *Dichrostachys cinerea* at the base of stems (a, b). Noticeable are the dry and dead stems, and the juvenile resprouts below, which led to a reclassification from adults to juveniles. Images c and d show new recruits of *Ehretia amoena* (c) and *Combretum mossambicense* (d) growing out of cracks in the soil, created by drought (Fotos: Trotter 2019).

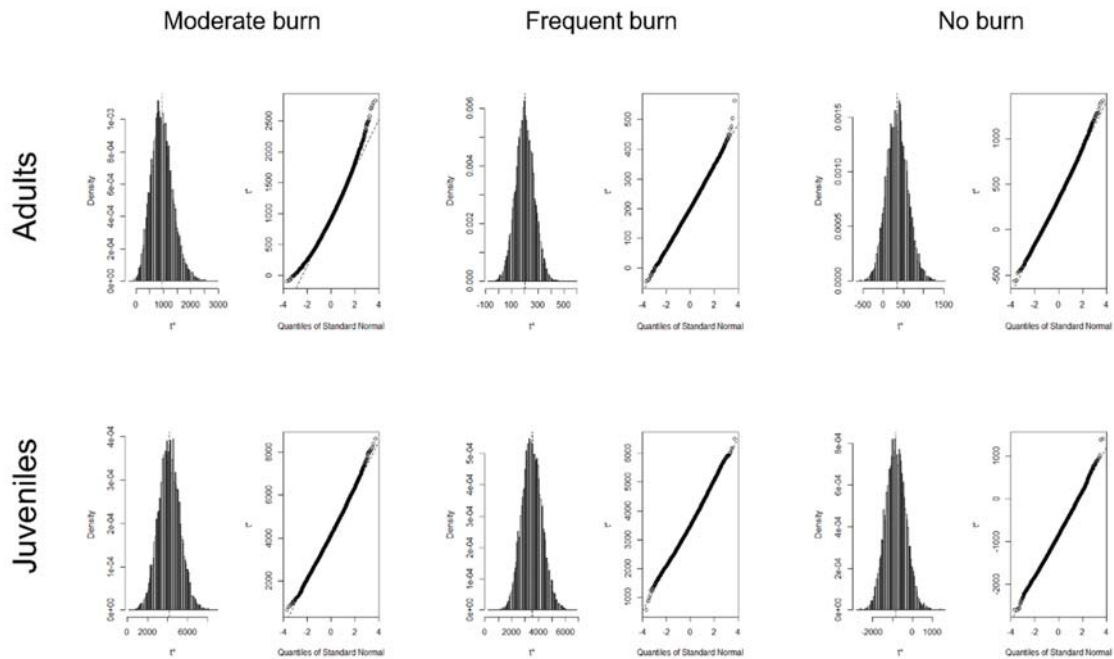


Figure S5: Density histograms and qq-plots illustrating the distribution of adult and juvenile species abundance data across each fire treatment for all years when the data is bootstrapped with 10,000 replicates. The bootstrapped data assumes near normal distribution on all three fire treatments across a confidence interval of 95%.

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