ORIGINAL ARTICLE

WILEY bioTROPICA · ASSOCIATION FOR TROPICAL BIOLOGY AND CONSERVATION

Enhancing ecosystem services through collaborative grass removal and fire exclusion in the Eastern Ghats

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Abstract

Human activities in mesic savanna ecosystems have resulted in plant communities that are heavily dominated by fire-tolerant grass species, are less diverse, and offer fewer ecosystem services such as palatable plant biomass. Experimental studies manipulating fire and grass presence have mostly been conducted in ecosystems invaded by exotic grass species. However, these experiments are also relevant to ecosystems that have become dominated by native grass species due to changes in human activities. Our study compared three different management treatments in the Eastern Ghats of India, where mesic savanna ecosystems are highly dominated by the unpalatable native *Cymbopogon* grasses, specifically: (1) control (both *Cymbopogon* grasses and fire are present), (2) fire exclusion, and (3) manual removal of *Cymbopogon* grasses and fire exclusion. We found that both grass removal and fire exclusion were necessary to significantly increase palatable herbaceous plant biomass and species diversity, and that species diversity responses were only significant at larger spatial grains of investigation. High site-to-site variation in the grass removal and fire exclusion treatment prevented us from detecting significant differences in species composition across treatments, even though particular palatable grass species benefited from the treatment. Our study is in line with research from other mesic savanna systems showing that fire management alone is not sufficient to transition ecosystems to a desired or historical state. We demonstrate how normally costly management practices, such as manual removal of dominant grasses, can be reasonably achieved through collaboration between scientists, community governed village institutions, and government and nongovernment agencies in socioecological systems.

KEYWORDS
commons, *Cymbopogon*, ecosystem services, experimental removal, mesic savanna, native invasive, village institutions

1 | INTRODUCTION

Savanna ecosystems harbor high plant biodiversity (Bond & Parr, 2010; Myers et al., 2000; Nerlekar et al., 2022) and provide critical ecosystem services, such as palatable plant biomass for grazing herbivores (Solbrig et al., 2013; Staver & Hempson, 2020). Fires in savanna ecosystems date back to the Late Glacial and Holocene periods (Kellman, 1975; Riedel et al., 2021) and are major factors influencing their diversity and composition (Bessie & Johnson, 1995; Bond & van Wilgen, 2012). The fire frequency of these ecosystems and the postfire responses of the plant community will depend on the environmental conditions and the traits of the species (Gibson et al., 1999). For example, many fire-tolerant grass species occur in savanna ecosystems, which can quickly resprout and recover postfire (Moore et al., 2019; Vidaller et al., 2019). Human activities, including anthropogenic climate change, exotic grass introductions, fire ignition, and plantation forest management can create conditions that increase the dominance of fire-tolerant grasses. This increase in dominance can lead to increased frequency and intensity of fires (Fusco et al., 2019) via positive feedbacks known as the grass–fire cycle (D'Antonio & Vitousek, 1992; Humphries, 1993). The resulting communities that are heavily dominated by fire-tolerant grasses may offer fewer ecosystem services, such as biodiversity and palatable biomass.

Most of our knowledge on the separate and interactive effects of dominant grasses and fire on plant communities comes from experiments in ecosystems that are dominated by invasive grasses that have changed fire regimes in the USA and Australia, such as *Cenchrus ciliaris* (Buffelgrass) (D'Antonio & Vitousek, 1992; Humphries, 1993), *Andropogon gayanus* (Gamba grass) (Cochrane, 2008), *Bromus tectorum* (Cheatgrass) (Bradley et al., 2018), *Eragrostis lehmanniana* (lehmann lovegrass) (Crimmins & McPherson, 2008), and *Imperata cylindrica* (Cogongrass) (Tomat-Kelly et al., 2021). Treatments aim to restore historic (preinvasion) fire regimes, decrease the abundance of exotic grass and increase the diversity and abundance of native plant species. Common treatments include creating fuel breaks that reduce the spread of fire (Agee et al., 2000; Ingalsbee, 2005) and removing dominant invasive grass species (Buisson et al., 2020; Crimmins & McPherson, 2008; McCain et al., 2010). Combining these treatments with active seeding or planting of herbaceous species is often found to be necessary to restore a diverse plant community (Davies et al., 2019; Kulpa et al., 2012).

Such experimental research is also relevant in continents in which native grasses have become more dominant in the understory of open-woody savanna systems with human activities. In India, grassland and savanna ecosystems are widespread, covering approximately 319,675 km² (Madhusudan & Vanak, 2022). These ecosystems play an important role in rural livelihoods (Madhusudan & Vanak, 2022; Vanak et al., 2015) and are thus important socio-ecological systems (Meinzen-Dick et al., 2021; Wolford et al., 2013). Savanna ecosystems have existed in India for over a million years, as indicated by fossil and molecular evidence from the diverse and

endemic C_4 grasses, predating any human modification of the landscape (Ratnam et al., 2016). For example, at least 15 grasses in the genus *Cymbopogon* (lemongrasses) are known from India and many are endemic to particular habitats and regions (Soenarko, 1977). This makes these ecosystems different from any of the ecosystems invaded by non-native C_4 grasses discussed above, which were not historically savannas.

Although a monsoonal climate, fire, and wild herbivores have historically shaped savanna plant communities in the Eastern Ghats, recent human activities have brought about conditions that enhance the dominance of selected grass species. Domesticated animals have been present in the Eastern Ghats for thousands of years (Allchin, 1963; Paddayya, 1968; Singh, 2013); however, the number of these animals has increased in recent years with rising human population (Planning Officer, C, 2018, 2022; Superintendent of Census Operations, 1964, 1965). The fire regime has changed in recent years (Neeraja et al., 2021) due to anthropogenic fires that promote new growth grasses that are more palatable to livestock combined with more arid conditions created by climate change (Senande-Rivera et al., 2022). These may favor the dominance of *Cymbopogon* grasses, which are fire and drought tolerant and are typically avoided by grazers except for a short period after burning (Sankaran, 2005).

Prior to 1987, *Cymbopogon* grasses in the Eastern Ghats were harvested as a thatching material for constructing roofs on homes (Kukrety, 2011; Strohbach & Walters, 2015). However, this grass harvesting declined starting in 1987 as missions to improve human living standards provided alternative materials for construction (Housing Department, Andhra Pradesh, 2013). Forest surveys from 1961 describe savanna ecosystems in the Eastern Ghats to have a scattered presence of *Cymbopogon* species (Superintendent of Census Operations, 1964). Further, pastoralists report that prior to 1987, *Cymbopogon* grasses were sparse and palatable fodder for livestock was more plentiful. Since 1987, they have observed that *Cymbopogon* has become more dominant in the understory, that fires have become more frequent and intense, and that there is a lower availability of palatable herbaceous species for a similar abundance of livestock animals (Dabadghao & Shankarnarayan, 1973; Kohli et al., 2011; Neeraja et al., 2021; Schmerbeck & Seeland, 2007; Snyman, 2015; Soenarko, 1977).

It is important to manage these *Cymbopogon*-dominated tropical mesic savanna ecosystems (FES, 2009; Kumar, 2014; Ratnam et al., 2011) in the Eastern Ghats toward a state that more resembles the recent past and offers ecosystem services such as biodiversity and palatable herbaceous plant biomass. Because *Cymbopogon* grasses can rapidly resprout and recover quickly from fire (Sankaran, 2005), it is possible that reducing the frequency of fire by creating firebreaks will be adequate to allow the recovery (Bachinger et al., 2016) of less fire-tolerant species, some of which will be palatable species (Archibald et al., 2019). Alternatively, an additional treatment in which unpalatable and fire-tolerant *Cymbopogon* are manually removed might be necessary to significantly increase the abundance and diversity of palatable herbaceous

plants, due to the high dominance of *Cymbopogon* grasses currently in the system and the likely strong role *Cymbopogon* grasses play as a competitor for space and light. To test the effects of fire exclusion alone or grass removal and fire exclusion together on the diversity and composition of tropical mesic savanna ecosystems in the Eastern Ghats, we conducted a large-scale experiment across the Anantapur and Chittoor districts. Specifically, we considered three experimental treatments (Figure 1b): (1) control (both *Cymbopogon* and fire are present), (2) fire exclusion, (3) manual removal of *Cymbopogon* and fire exclusion. We measured the biomass of palatable and unpalatable plants and quantified the diversity of plants at multiple spatial grains.

2 | METHODS

2.1 | Study region and site selection

Our study was located in the southern part of the Eastern Ghats, specifically in the Anantapur and Chittoor within Andhra Pradesh, India (13.76°-14.11° N and 78.41°-78.33° E) (Muthumperumal & Parthasarathy, 2009) (Figure 1a). This area has a tropical arid climate (FES, 2009) with precipitation occurring during June–December (Rajeevan et al., 2012). An average of 550 mm rainfall per year has been received across the last 28 years (1992–2020). Our study area is classified as tropical mesic savannas (Ratnam et al., 2011, 2016) and occurs between 379 and 556 m above sea level. This biome has open tree canopies and C_4 grasses in the understory (Ratnam et al., 2019). The understory is dominated by the Poaceae Family, and in particular the Genus *Cymbopogon* in Andropogoneae with C_4 pathway (McCain et al., 2010; Rajendrudu & Das, 1981; Ratnam et al., 2016), which are known for their high flammability in savannas across the world (Bianconi et al., 2020; Bond, 2008; Ripley et al., 2015).

Land in the study region is owned by either the Forest Department or the Revenue Department and is managed by agropastoralists in community governed village institutions (Meinzen-Dick et al., 2021). The agro-pastoralist communities treat these as common lands and rely on these areas to graze livestock, gather firewood, and gather nontimber forest resources (Chopra & Dasgupta, 2002; FES, 2003). The study sites are grazed by domestic animals such as cattle, buffaloes, sheep, and goats, including indigenous cattle such as Punganur, Ongole, Deoni, and Hallikar (NDDB, 2018). These mesic savannas provide habitat to many native birds such as francolins, harriers, partridges, quails, and larks (Manakadan, 2014; Madhusudan et al., 2021) as well as native mammals, including as *four-horned antelope* (*Tetracerus quadricornis*), black-naped hare (*Lepus nigricollis nigricollis*), little Indian field mouse (*Mus booduga*), Indian gerbil

(*Tatera indica*), blackbuck (*Antilope cervicapra*), chital (*Axis axis*), and sambar (*Rusa unicolor*) (FES, 2003). To conduct this research, we engaged directly with village institutions that promote the rights of their members through participation in management decisions and equitable benefit sharing (Kegamba et al., 2022).

2.2 | Experimental design and sampling

In 2019, we proposed an experiment to the community governed village institutions to test whether exclusion of fire or removal of both fire and *Cymbopogon* were necessary to convert these systems to a state that would provide valuable fodder for grazing animals. The community agreed to partner with us and gave permission to use the land for our experiment. The three experimental treatments are (Figure 1b): (1) control (both *Cymbopogon* and fire are present), (2) fire exclusion (*Cymbopogon* present and fire absent, CPFA), and (3) manual removal of *Cymbopogon* and fire exclusion (both *Cymbopogon* and fire absent, CAFA).

By the end of December 2019, 28 village institutions removed *Cymbopogon* from 136 areas covering a total of 0.71 km² using spades and crowbars. This removal method does create more disturbance than clipping the aboveground biomass of the plant, but this was the only option for removal because of the capacity of *Cymbopogon* to rapidly resprout when it loses its aboveground biomass. We do not find other species growing beneath the foliage of *Cymbopogon*, and thus, our removal method likely did not harm the belowground bud banks of other native forbs and grasses.

All of these areas are regularly burned, but had no fire in the past 2 years. Ideally, we would have identified sites that had not been burned for longer (5–10) years, however, such sites do not occur in our region except in areas that would have had confounding conditions (e.g., sites closer to the ocean that get more precipitation). We randomly choose 30 of these areas to be sites for the CAFA treatment. We paired these with 30 sites nearby that also had no fire in the past 2 years but for which *Cymbopogon* was present for the CPFA treatment, and with 15 nearby control sites that were burned during the most recent fire season (between February and June 2019). Average distance between paired sites was 60 meters and average distance between sites was 2.4 km. The absence of fire in the treatment sites in 2019 is due to chance, rather than differences in abiotic or biotic conditions. During fire season, fire can be initiated in any possible location within this region, and their spread depends on natural firebreaks, wind, and weather. To better understand the background fire frequency at the site level, we examined data on fire presence across 3 years (2017, 2018, and 2019) for 180 sites (each site is a circle with a radius of 10 m) that are close to our

FIGURE 1 (a) Location of the study area within the Eastern Ghats, India, and (b) location of study sites shown separately for each treatment. Treatments are: control (both *Cymbopogon* and fire are present, CPFP), fire exclusion (*Cymbopogon* present and fire absent, CPFA), and removal of *Cymbopogon* and fire exclusion (both *Cymbopogon* and fire absent, CAFA). (c) Diagram of a site as defined in this study (green area, 100 m by 20 m), 100 m transect (black dotted line) and plots (blue squares, 1 m by 1 m). (d) Photos of sampling herbaceous plants from (e) the CAFA treatment and (f) control treatments and (g) height of *Cymbopogon* (218 cm) compared to a human in the control treatment.

(a)

(b)

(c)
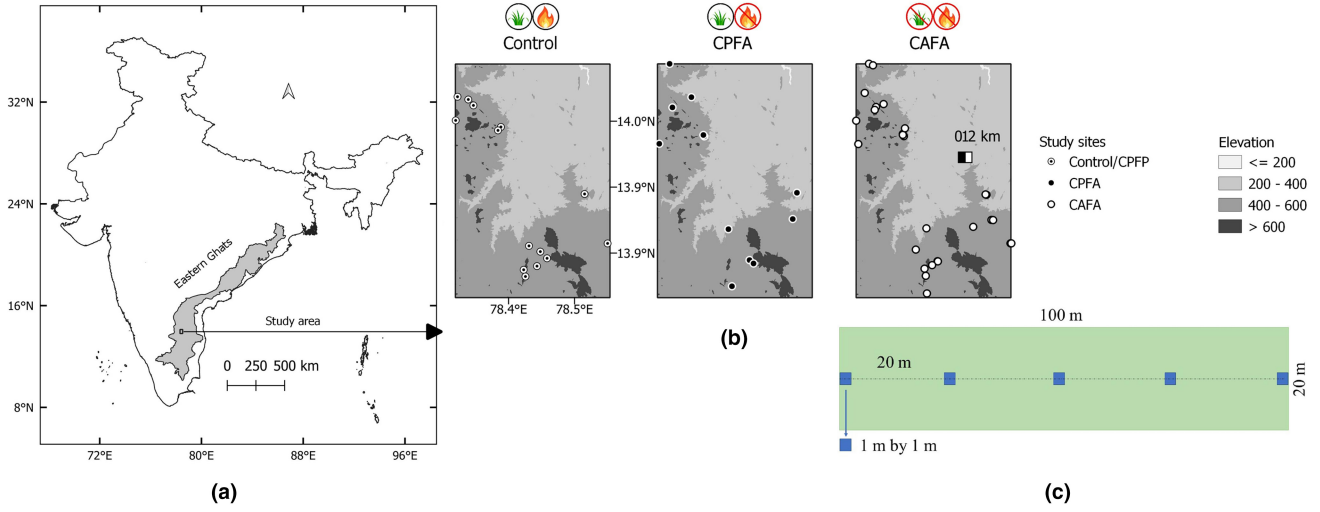
(d)

(e)

(f)

(g)

focal sites (see Figure S1 for location of all sites). We find 86 of these sites burned in all 3 years, 53 burned in two of the 3 years, and 36 burned in a single year, indicating a fire return interval (years/fires) of 1.7 years for our study region.

To create the CAFA treatment, four unskilled labourers worked for about 4.8 h to remove *Cymbopogon* from an area of 4046 m^2 (one acre) and were paid INR 1350 (USD 19.18) as per 2019 Mahatma Gandhi National Rural Employment Guarantee Act (MGNREGA- an Indian labour law that aims to guarantee the right to work) (Published by Authority, 2019).

To prevent fire from the two fire absent treatments (CPFA and CAFA) we created a two-meter-wide fire break around sites in these treatments. Despite the firebreaks, during the 2020 fire season (February to June), fire spread to some of our treatments which were supposed to have fire absent. We lost three replicates from the CAFA treatment and 16 from the CPFA treatment. We also lost three replicates from the control sites, because fire did not spread through the site. The final number of sites remaining was 27 in the CAFA, 14 in the CPFA, and 12 in the control. The locations of the sites for each treatment covered a similar spatial extent in the region (Figure 1b).

In October 2020, toward the end of the peak growing period (Tamilselvan et al., 2021), we set up a 100 m by 1 m long transect in each site that contained five 1 m^2 plots, separated from each other by 20 m (Figure 1c). Live aboveground biomass was clipped from each plot (see photographs in Figure 1d,e). Biomass was sorted to species, dried at 70^{0}C, and weighed to the nearest 1 g. A herbarium specimen was taken of each unique species and identified to the species level using keys (Hajra et al., 1995; Pullaiah, 2018; Pullaiah & Muralidhara Rao, 2002; Pullaiah & Sandhya Rani, 1999; Pullaiah & Sri, 2000; Sharma & Balakrishnan, 1993; Sharma & Sanjappa, 1993; Singh et al., 2015).

2.3 | Quantifying plant biomass

We categorized plants into three categories (hereafter, "palatable category"): *Cymbopogon*, other "unpalatable" herbaceous plants except *Cymbopogon*, and "palatable" herbaceous plants. Institutions from 28 villages requested their members to shortlist local pastoralists with traditional knowledge, specifically those who frequently visited these sites with their livestock. The objective was to gather insights about the local plants' characteristics and their palatability. However, not all village institutions had members with this traditional understanding. Recognizing this gap, and rather than casting a wider net to the entire community, they managed to identify 14 local shepherds from these villages. These shepherds then voluntarily stepped forward to provide the needed information on plant species palatability. These 14 pastoralists were placed in seven teams, each with two persons. We then used participatory rural appraisal (PRA) guidelines to extract knowledge from a group setting (Chambers, 1994; Jackson & Ingles, 1998). The first part of the PRA aimed to create an atmosphere and motivate

participants to share their knowledge. We asked the entire group to prepare a map of their grasslands, and asked each team to share with the group about their observations and concerns about their grasslands and what changes they have observed since the beginning of their pastoral work. After they shared their observations, we explained the method for the second session for categorizing plants. To ensure that all participants share their knowledge, we avoided writing and we asked each team to take a close look at the specimen of the plant and to place a colored sticker on it to indicate its palatability. We used green (score one if eaten by all three types of livestock—sheep, goat, and cattle—for more than 6 months out of the year), yellow (score two if eaten by two of the three types of livestock for more than 3 months out of the year or until awns develop and make it hard for the animals to eat), blue (score three if palatable to one or two types of livestock for up to 3 months), and red (score four if not palatable to any of the livestock) to denote animal preference based on their observations. We then brought the larger group back together to reach a consensus score for each plant species, and to gather more information about which part of the plant is eaten, and for which months in the year they have observed the plant being eaten. This information is given for each plant species in Table S1. For the analysis, we summarized palatability into two categories: palatable plants (score of one or two) and not palatable plants (score of three or four). These two categories best describe the ecosystem service, as plants with a score three (e.g., *Cymbopogon*) are only palatable for a short period (FES, 2009; Sankaran, 2005) and shepherds cannot rely on them to rear livestock.

We quantified the total biomass of each palatable category as the sum across all plots within a site. For each site, we quantified relative biomass for each palatable category as the sum of biomass for each palatable category across all plots within a site, then divided the sum of each category by the total biomass of each site. We used a univariate hierarchical linear mixed effects model to test whether treatments influenced the absolute and relative biomass of palatable and unpalatable plants per site assuming a Student's t-distribution. In this model we used palatable category, treatment and their interaction as categorical fixed effects, and site as a random effect.

To understand the influence of CPFA and CAFA treatments relative to the control treatment on the biomass of species in three palatable categories (*Cymbopogon*, other 'unpalatable' herbaceous plants except *Cymbopogon*, and 'palatable' herbaceous plants), we subtracted the relative biomass of every species in each CPFA and CAFA treatment and site, by the relative biomass of the same species in the nearest neighboring control site (within the same district). Because some CPFA or CAFA sites did not have adjacent control sites, 14 sites were used for this pairwise comparison. To quantify how each treatment changed the biomass of every species compared to the control, we used a hierarchical linear mixed effects model with change in relative biomass per species as the response variable, treatment (CPFA and CAFA), palatability and their interaction as categorical fixed effects, and distinct villages of sites and species as

nested random effects assuming a Student's t-distribution. Posterior predictive plots were used to visually determine how well models reproduced the data (Figures S2 and S3).

2.4 | Quantifying plant diversity and composition

We quantified herbaceous plant diversity at the α-scale and the γ-scale. α-scale, which we define here as the diversity in a site (diversity across five 1m² plots, total area 5 m²) and the γ-scale, which here we define as the total diversity across 10 sites within a treatment (diversity across 50 1m² plots, total area 50 m²). To quantify site-to-site variation within treatment, we used Whittaker's multiplicative β-diversity (γ/α) (Whittaker, 1972). At each spatial scale, we calculated two measures of diversity; (i) species richness, that is, the total number of species observed, and (ii) species evenness, that is, low values mean that one or more species are dominant, while high values mean that relatively equal numbers apply to each species. We used the relative biomass of each species to estimate species evenness using the inverse Simpson's diversity index (Simpson, 1949).

We quantified the effect of experimental treatments on α-scale plant diversity using hierarchical linear mixed effects models. The α-scale species richness was modeled assuming a poisson distribution. We included treatment as a categorical fixed effect, with the site as a nested random effect. The α-scale species evenness was modeled assuming a lognormal distribution. Posterior predictive plots were used to visually determine how well models reproduced the data (Figures S4 and S5).

At the γ-scale, our site sample sizes were unbalanced (12 control, 14 CPFA, and 27 CAFA) and we therefore considered 10 sites to be the γ-scale. We selected 10 sites per treatment randomly. We made comparisons across treatments using sample-based rarefaction (Gotelli & Colwell, 2011). Using bootstrap resamples without replacement, 10 random sites were sampled from each treatment category 200 times. We calculated the species richness and species evenness for each of the bootstrap resamples. The effects of experimental treatments at the γ-scale were examined using the mean and the 95% quantiles of the resamples.

We used the same bootstrap resamples generated for the γ-scale analysis to estimate experimental treatment effects on β-diversity. For each of the resamples, we calculated α-scale means of species
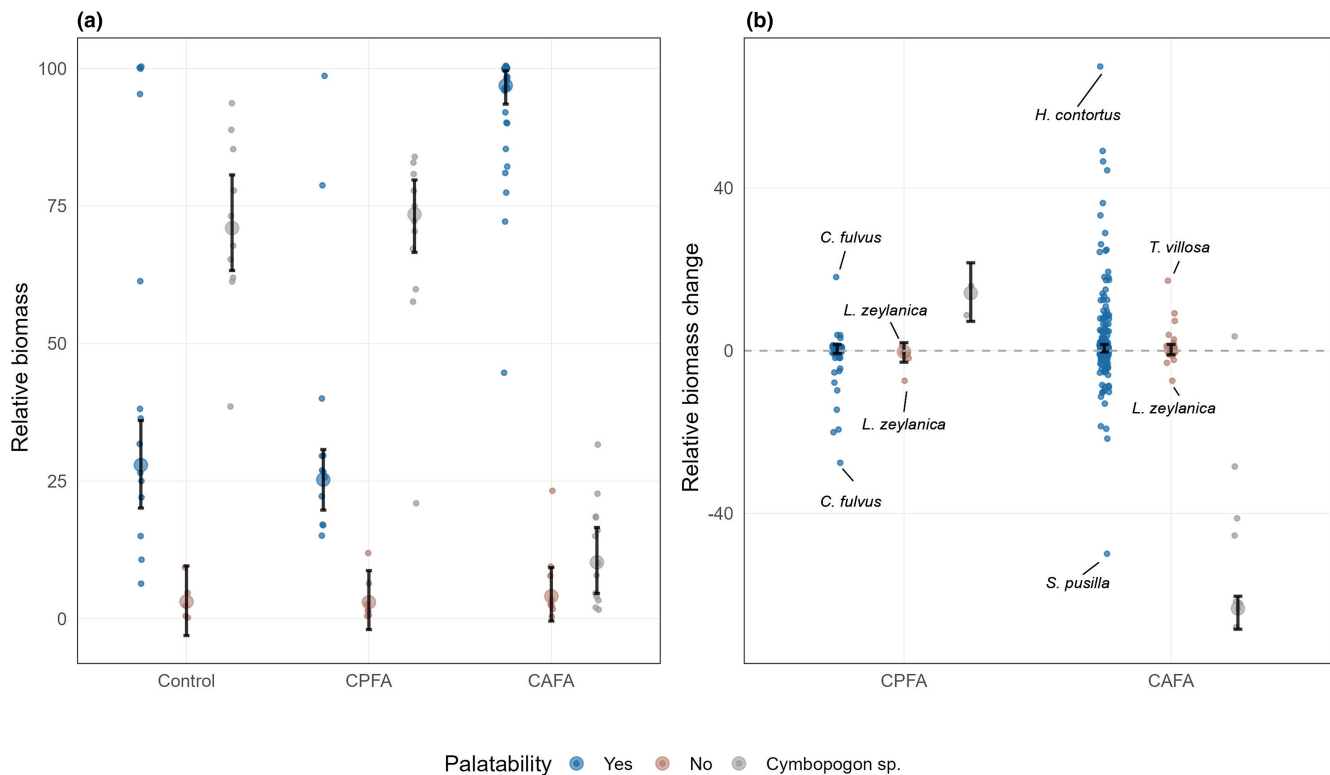

FIGURE 2 (a) Relative biomass (g) at site, each treatment and plant palatable category. Small points are the relative biomass of each palatable category at each site, the data models were fit to. Large points are the conditional effects for each treatment and palatable plant category and lines show the 95% credible intervals. (b) Relative biomass change of species between treatments (CPFA, CAFA) and paired control sites. Small points show the change in biomass in each species and site compared to the nearest control. The dashed line at zero represents relative biomass change of zero. Positive values indicate that the species has higher relative biomass in the treatment compared to control site; whereas negative values show that the species has higher relative biomass in the control site. Large points are the conditional effects for each treatment and palatable category and lines are the 95% credible intervals. Labels show species gained and lost due to treatments. *Cymbopogon* was observed to have higher relative biomass than the control in one of the CAFA sites (absolute weight 50 g in one CAFA site).

evenness and species richness. β-diversity and β-evenness were calculated as the ratio of the resampled γ/α metrics. We estimated the effects of experimental treatments using the mean and 95% quantiles of all the resampled β-diversities.

To examine species accumulation from the site-scale to treatment-scale continuously, we quantified the sample-based accumulation of species. This was calculated for two levels of diversity, using Hill numbers (Hill, 1973). Specifically, we quantified diversity across sampling units for $q=0$ which is equivalent to species richness, and for $q=2$ which is a metric of evenness. We compared rarefied diversity across treatments and sampling levels, and extrapolated expected species richness to 60 sites.

2.4.1 | Quantifying species composition

We quantified the effect of experimental treatments on species composition between sites using nonmetric multidimensional scaling (NMDS) and quantified the distance between site level and treatment biomass abundances using Bray-Curtis (Bray & Curtis, 1957) dissimilarity. We computed dissimilarity between treatments with "bray" as the distance measure with 99 permutations to know the homogenous dispersion, and calculated the beta dispersion. Finally, to test whether treatments varied in their composition, we performed an ANOVA and Tukey HSD tests on distance measures.

Data collection was done using the 'collect' tool in the Open Data Kit (ODK) (Hartung et al., 2010) and transferred to Google Sheets. ODK reduced the errors of input during data collection in the field and saved time from entering data manually. Maps for planning the study and for this article were prepared using QGIS (QGIS Development Team, 2022). All data preparation, manipulation, calculation, analysis, and visualization of results were conducted in R version 4.1.3 (R Core Team, 2022). To calculate metrics, we used "tidyverse" (Wickham et al., 2019) and "vegan" (Oksanen et al., 2020). Graphs were created using the "ggplot2" (Wickham, 2016) and "patchwork" (Pedersen, 2022), tables were made with "gt" (Iannone et al., 2022), and "webshot2". (Chang, 2022). For Bayesian inference and estimates of uncertainty, all hierarchical mixed effects models described above were coded using the "brms" (Bürkner, 2018) and fitted using the Hamiltonian Monte Carlo (HMC) sampler Stan (Carpenter et al., 2017). All models were fitted with four chains and 3000 iterations, with 1000 used as warmup. We used weakly regularizing priors and visual inspection of HMC chains showed convergence. To compute rarefaction and extrapolation of Hill numbers we used the "iNEXT.3D" (Hu & Chao, 2022).

3 | RESULTS

Across all of our sites, we observed 68 herbaceous plant species (12 unpalatable and 56 palatable species) in 19 Families. Out of the 68 species, there were 50 forbs (37 annual and 13 perennial) and

18 graminoids (nine annual and nine perennial). Poaceae was the most diverse family (17 species) followed by Fabaceae (15 species) (Figures S6 and S7). We found that *Boda/Bade* (Telugu and Kannada) is the common name for three different *Cymbopogon* grasses found in our sites: *Cymbopogon martinii* (Roxb.) J. F. Watson, *Cymbopogon caesius* (Hook. & Arn.) Stapf, and *Cymbopogon jwarancusa* (Jones) Schult (Table S1).

The study area of our investigation was primarily composed of native unpalatable *Cymbopogon* grasses, which accounted for 71% of the total plant biomass. The same locations were also included in a 2009 study by FES, which reported a *Cymbopogon* biomass of 0.09 tonnes per hectare (FES, 2009). In our study, we noted an increase in the *Cymbopogon* biomass, amounting to 0.13 t/ha, representing 1.44 times the 2009 levels.

3.1 | Biomass

Upon comparing the control treatment with the exclusion of fire (CPFA), we observed similar absolute (as shown in Table S2) and relative biomass (as seen in Figure 2a) across all palatable plant categories. By contrast, when we excluded both *Cymbopogon* and fire (CAFA), the absolute and relative biomass of *Cymbopogon* decreased. Meanwhile, both the absolute and relative biomass of palatable herbaceous plants increased. These results can be found in Table S3 and Figure 2a. Absolute biomass of palatable grass per hectare was 0.08 tonnes (t), unpalatable was 0.01 t and *Cymbopogon* was 0.22 t in the control.

Many species gained and lost biomass due to treatments. From the exclusion of fire and *Cymbopogon*, particularly palatable species such as *Heteropogon contortus*, *Arundinella nervosa*, and *Lepidagathis cristata* and unpalatable species such as *Acalypha alnifolia*, and *Tephrosia villosa* (Table S4, Figure 2b) benefited favorably.

3.2 | Diversity

Species richness was similar across treatments at both the α- and β-scales. There was a trend for higher species richness in the CAFA treatment at the γ-scale (Table S5, Figure 3). At the α-scale, we found a trend toward higher species evenness (3.44 species) when *Cymbopogon* and fire were experimentally excluded (CAFA). At β-scale and γ-scale, species evenness was significantly higher in the CAFA treatment compared to the other treatments (Table S6, Figure 4). The scale-dependent result can also be visualized with accumulation curves (Figure 5, Table S7). The richness and evenness of treatments at a single site are similar, whereas the CAFA treatment accumulates more species when moving from one to many sites.

3.3 | Composition

Despite the clearly lower biomass of *Cymbopogon* in the CAFA treatment (Figure 2a), an ANOVA test revealed that all treatments
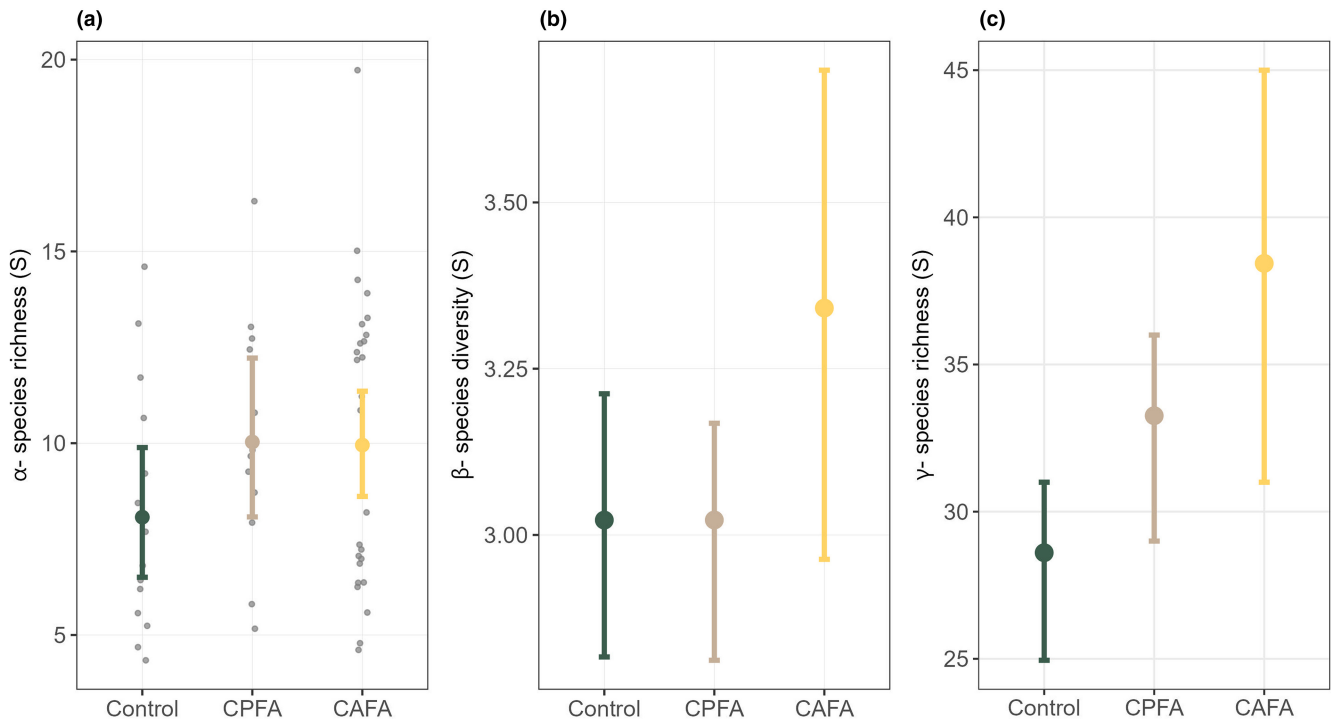
FIGURE 3 (a) α-scale species richness, (b) β-diversity, and (c) γ-scale species richness. For the α-scale (a), small points are the species richness for every site. Large points are the conditional effects and lines show the 95% credible intervals. For the β and γ-scale (b, c) large points show the mean, lines the 95% quantiles of 200 bootstrap resamples of 10 sites (without replacement).

were compositionally indistinguishable from each other ($p > 0.05$). Likewise, Tukey HSD tests did not show any significant pairwise differences across treatments in composition (all $p > 0.05$). The NMDS visualization makes it clear that the nonsignificance is due to the high site-to-site turnover, especially in the CAFA treatment (Figure 6).

4 | DISCUSSION

Globally, savanna ecosystems support high biodiversity as well as the livelihoods of millions of pastoralists, and some savannas are facing increasing dominance of competitive grass species (Bradley et al., 2018; Cochrane 2008; Crimmins & McPherson 2008; D'Antonio & Vitousek 1992; Humphries 1993; Tomat-Kelly et al., 2021) which may reduce their diversity and ecosystem services. In some ecosystems that are invaded by or dominated by fire-tolerant grass species, it is possible to restore the system to a reference state by reducing the frequency of fires (Cury et al., 2020; Moghli et al., 2022; Villarreal et al., 2020). In other systems, additional treatments are necessary to achieve significant increases in the abundance and diversity of native herbaceous species (Davies et al., 2019; Kulpa et al., 2012). We find that fire exclusion and an additional treatment in which grasses were manually removed were both necessary to increase biodiversity and palatable herbaceous biomass of mesic savanna ecosystems in the Eastern Ghats.

In the control treatment, the current high dominance of *Cymbopogon* grasses is likely explained by recent changes in land management. Specifically, local people no longer harvest *Cymbopogon*

for thatching houses (Kukrety, 2011). In addition, it is possible that *Cymbopogon* may have positive feedbacks that have increased its dominance through time. Fire-tolerant and highly flammable *Cymbopogon* grasses create a fuel load and may be increasing fire frequency and/or intensity, which then favors its relative abundance. This grass–fire cycle is studied mostly with invasive grasses (D'Antonio & Vitousek, 1992; Fusco et al., 2019, 2022), but in our case, we suspect that native *Cymbopogon* grasses can also create a similar positive feedback.

Our study highlights the importance of examining multiple spatial grains in biodiversity analysis (Chase et al., 2018; McGlinn et al., 2019). Treatment effects were not seen at the site level; however, the higher site-to-site variation in the CAFA treatment resulted in more diverse communities when multiple sites are considered. We specifically find that different native species became common and dominant at different sites in the CAFA treatment, and that across all sites, the communities were more even in the CAFA treatment. This might be the case in many other ecosystems, in which a single species dominates at the local scale, and site-to-site variability creates higher biodiversity at the landscape scale. Examining patterns at multiple spatial grains might often be helpful to understand responses to treatments and point to potential mechanisms. However, currently most experimental studies examining plant responses to fire consider a single spatial grain when analyzing biodiversity (e.g., Moghli et al., 2022)

We observed that different native palatable species became abundant at different sites within the CAFA treatment, such as *Arundinella nervosa*, *Heteropogon contortus*, *Lepidagathis cristata*, and
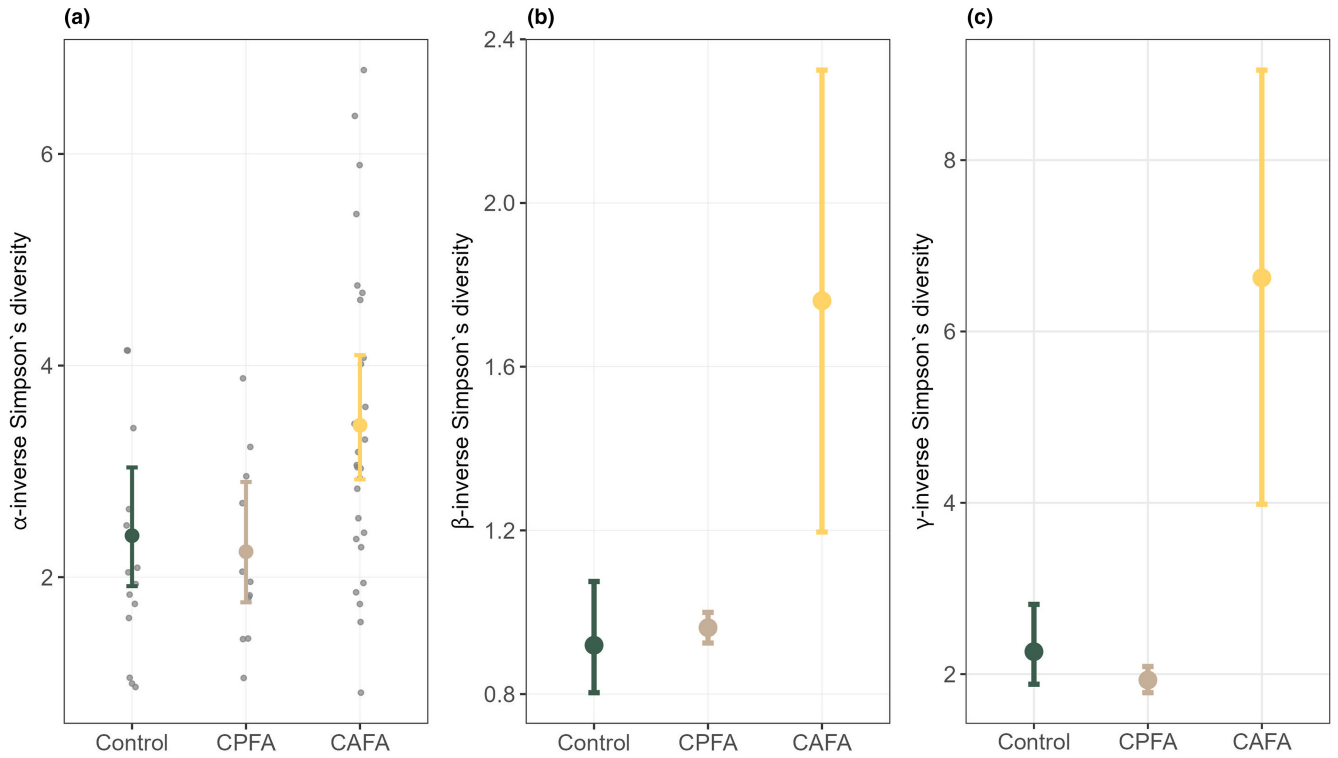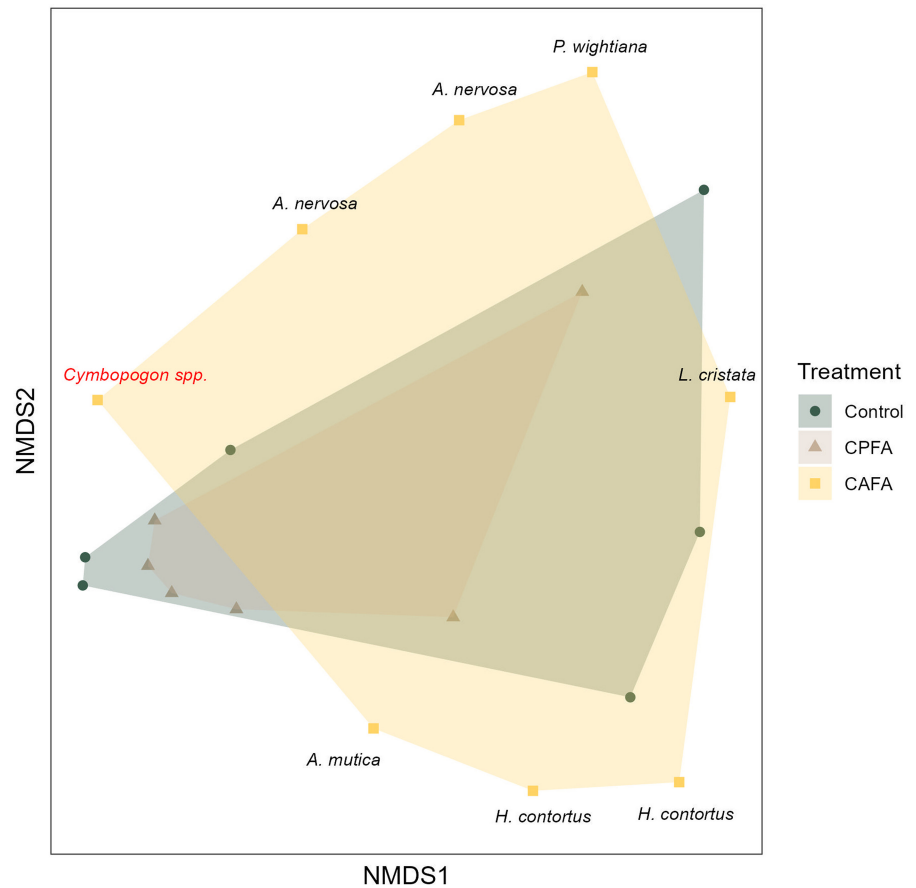
FIGURE 4 (a) α-scale species evenness (inverse Simpson's diversity), (b) β-scale species evenness, and (c) γ-scale species evenness. For the α-scale (a), small points show the data for each site, large points are the conditional effects and lines show the 95% credible intervals. For the β and γ-scale (b, c) large points show the mean, lines the 95% quantiles of 200 bootstrap resamples of 10 sites (without replacement).
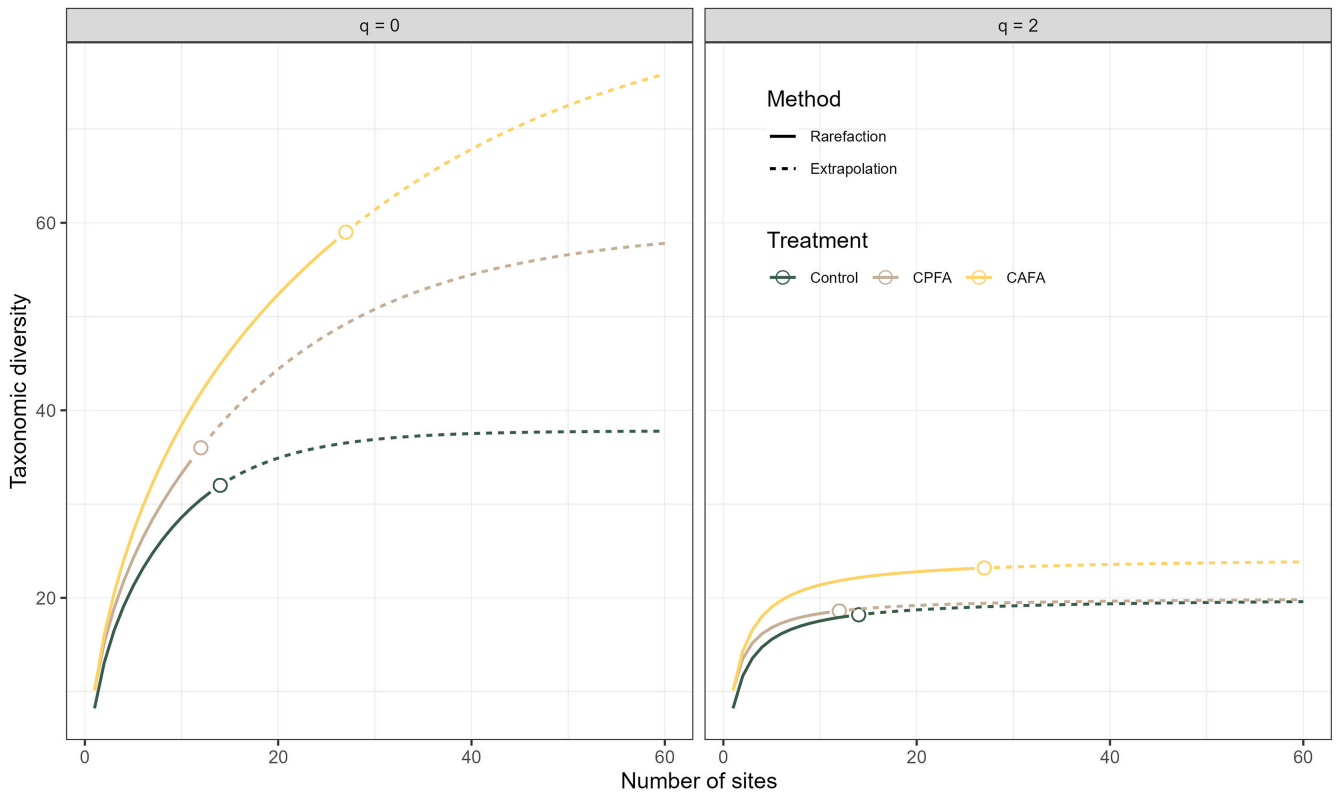
FIGURE 5 Species richness (q = 0) and species evenness (q = 2) accumulation curves across sites for each treatment. Solid lines indicate rarefaction values, points indicate observed values at sampled sites, and dotted lines indicate extrapolated values to 60 sites.

or those lacking soil conservation interventions. The most favorable locations for Cymbopogon grass removal align with areas currently targeted by the Mahatma Gandhi National Rural Employment Guarantee Act 2005 (MGNREGA). This Indian social welfare initiative guarantees the "right to work" and contributes to both rural employment and environmental management, particularly through efforts to conserve soil moisture (SPS, 2007).

Our results suggest that current policies regarding fire in India might not be effective at protecting biodiversity and ecosystem services. Currently, the National Forest Policy (MoEFCC, Government of India, 2018) bans fire with a goal of protecting forest and savanna ecosystems. According to the Indian Forest Act 1927, setting forest fire is a punishable offense. However, as we show here, removing fire alone will not be sufficient to reach goals aimed at protecting biodiversity (Durigan et al., 2020; Smith et al., 2013) or providing palatable fodder for wild and domesticated animals. We show that the management of savanna ecosystems is possible by community governed village institutions and that this management intervention has benefits for livelihood and biodiversity. Such community governed management of commons has also been successful in managing waterbodies in Madhya Pradesh, India (Falk et al., 2020; Galán et al., 2022).

We show that the removal of dominant grass and exclusion of fire provided benefits for livelihood and biodiversity that were seen in a shorter time frame, and thus gave a rapid return of investment (Lehmann & Parr, 2016). However, we note that manual removal of grasses that was conducted for this study is labour intensive and might be difficult to implement at a landscape scale by the users alone (e.g., shepherds in community and village institutions). The large-scale management of tropical mesic savannas in the Eastern Ghats can be accomplished while also creating employment opportunities for villagers if management treatments are supported by programs like MGNREGA and CAMPA (Compensatory Afforestation Fund Management and Planning Authority which is aimed for afforestation for forest land diverted to nonforest uses). Moving forward, it is critical for scientists, the government and NGOs to work collaboratively to arrive at a meaningful and lasting impact on the socioecological systems and their associated services.

AUTHOR CONTRIBUTIONS

TMK and SCS conceived and designed the experiment with inputs from HS; SCS collected data in 2020; SCS, PA, MAK and RPR identified plants; SCS and EL performed the statistical analyses with inputs from TMK; SCS and TMK wrote the first draft of the manuscript; all authors provided inputs and approved the final manuscript version.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available at Zenodo: https://zenodo.org/record/8338233 and code to reproduce results is freely available on GitHub: https://github.com/cssaneesh/biomass_eghats.git.

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REFERENCES

Agee, J. K., Bahro, B., Finney, M. A., Omi, P. N., Sapsis, D. B., Skinner, C. N., van Wagtendonk, J. W., & Phillip Weatherspoon, C. (2000). The use of shaded fuelbreaks in landscape fire management. *Forest Ecology and Management*, *127*, 55–66.

Allchin, F. R. (1963). *Neolithic cattle-keepers of South India: A study of the Deccan ash mounds*. Cambridge University Press.

Archibald, S., Hempson, G. P., & Lehmann, C. (2019). A unified framework for plant life-history strategies shaped by fire and herbivory. *The New Phytologist*, *224*, 1490–1503.

Bachinger, L. M., Brown, L. R., & van Rooyen, M. W. (2016). The effects of fire-breaks on plant diversity and species composition in the grasslands of the Loskop Dam Nature Reserve, South Africa. *African Journal of Range and Forage Science*, *33*, 21–32.

Bessie, W. C., & Johnson, E. A. (1995). The relative importance of fuels and weather on fire behavior in subalpine forests. *Ecology*, *76*, 747–762.

Bianconi, M. E., Hackel, J., Vorontsova, M. S., Alberti, A., Arthan, W., Burke, S. V., Duvall, M. R., Kellogg, E. A., Lavergne, S., McKain, M. R., Meunier, A., Osborne, C. P., Traiperm, P., Christin, P.-A., & Besnard, G. (2020). Continued adaptation of C4 photosynthesis after an initial burst of changes in the Andropogoneae grasses. *Systematic Biology*, *69*, 445–461.

Bond, W. J. (2008). What limits trees in C_4 grasslands and savannas? *Annual Review of Ecology, Evolution, and Systematics*, *39*, 641–659. https://doi.org/10.1146/annurev.ecolsys.39.110707.173411

Bond, W. J., & Parr, C. L. (2010). Beyond the forest edge: Ecology, diversity and conservation of the grassy biomes. *Biological Conservation*, *143*, 2395–2404.

Bond, W. J., & van Wilgen, B. W. (2012). *Fire and plants*. Springer Science & Business Media.

Bradley, B. A., Curtis, C. A., Fusco, E. J., Abatzoglou, J. T., Balch, J. K., Dadashi, S., & Tuanmu, M.-N. (2018). Cheatgrass (Bromus tectorum) distribution in the intermountain Western United States and its relationship to fire frequency, seasonality, and ignitions. *Biological Invasions*, *20*, 1493–1506.

Bray, J. R., & Curtis, J. T. (1957). An ordination of the upland forest communities of southern Wisconsin. *Ecological Monographs*, *27*, 325–349.

Buisson, E., Fidelis, A., Overbeck, G. E., Schmidt, I. B., Durigan, G., Young, T. P., Alvarado, S. T., Arruda, A. J., Boisson, S., Bond, W., Coutinho,

A., Kirkman, K., Oliveira, R. S., Schmitt, M. H., Siebert, F., Siebert, S. J., Thompson, D. I., & Silveira, F. A. O. (2020). A research agenda for the restoration of tropical and subtropical grasslands and savannas. *Restoration Ecology, 29*, e13292. https://doi.org/10.1111/rec.13292

Bürkner, P.-C. (2018). Advanced Bayesian multilevel modeling with the R package brms. *R Journal, 10*, 395.

Carpenter, B., Gelman, A., Hoffman, M. D., Lee, D., Goodrich, B., Betancourt, M., Brubaker, M., Guo, J., Li, P., & Riddell, A. (2017). Stan: A probabilistic programming language. *Journal of Statistical Software, 76*, 1–32. http://www.jstatsoft.org/v76/i01/

Chambers, R. (1994). The origins and practice of participatory rural appraisal. *World Development, 22*, 953–969.

Chang, W. (2022). *webshot2: Take Screenshots of Web Pages*. https://CRAN.R-project.org/package=webshot2

Chase, J. M., McGill, B. J., McGlinn, D. J., May, F., Blowes, S. A., Xiao, X., Knight, T. M., Purschke, O., & Gotelli, N. J. (2018). Embracing scale-dependence to achieve a deeper understanding of biodiversity and its change across communities. *Ecology Letters, 21*, 1737–1751.

Chopra, K., & Dasgupta, P. (2002). *Common Pool resources in India: Evidence, significance and new management initiatives*. Final Report of DFID sponsored project on Policy Implications of Knowledge with respect to Common Pool Resources undertaken jointly with University of Cambridge https://assets.publishing.service.gov.uk/media/57a08d29ed915d622c0017f5/R7973AnnB.pdf

Cochrane, M. C. (2008). *Tropical fire ecology: Climate change, land use and ecosystem dynamics* (2009th ed.). Springer.

Crimmins, T. M., & McPherson, G. R. (2008). Vegetation and seedbank response to Eragrostis lehmanniana removal in semi-desert communities. *Weed Research, 48*, 542–551.

Cury, R. T. d. S., Montibeller-Santos, C., Balch, J. K., Brando, P. M., & Torezan, J. M. D. (2020). Effects of fire frequency on seed sources and regeneration in southeastern Amazonia. *Frontiers in Forests and Global Change, 3*, 82. https://doi.org/10.3389/ffgc.2020.00082

Dabadghao, P. M., & Shankarnarayan, K. A. (1973). *The grass cover of India*. Indian Council of Agricultural Research.

D'Antonio, C. M., & Vitousek, P. M. (1992). Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics, 23*, 63–87.

Davies, K. W., Bates, J. D., & Boyd, C. S. (2019). Postwildfire seeding to restore native vegetation and limit exotic annuals: An evaluation in juniper-dominated sagebrush steppe. *Restoration Ecology, 27*, 120–127.

Durigan, G., Pilon, N. A. L., Abreu, R. C. R., Hoffmann, W. A., Martins, M., Fiorillo, B. F., Antunes, A. Z., Carmignotto, A. P., Maravalhas, J. B., Vieira, J., & Vasconcelos, H. L. (2020). No net loss of species diversity after prescribed fires in the Brazilian savanna. *Frontiers in Forests and Global Change, 3*, 1–62. https://doi.org/10.3389/ffgc.2020.00013/full

Falk, T., Duche, V., Suhas, K. S., Agrawal, I., & Bartels, L. (2020). The role of community institutions in the management of water infrastructure in the context of Madhya Pradesh, India. *Journal of Indian Water Resources Society, 40*, 36–46. http://iwrs.org.in/journal/juloct2020/5jul.pdf

FES. (2003). *Biomass Assessment in Sadhukonda Reserve Forest and Adjoining Areas, Madanapalle, Andhra Pradesh, India*. Foundation for Ecological Security https://fes.org.in/resources/studies-&-reports/working-papers/wp9.pdf

FES. (2009). *Ecological baseline assessment of Kadiri watershed*. Foundation for Ecological Security https://fes.org.in/resources/studies-&-reports/studies/ecological-baseline-assessment-of-kadiri-watershed.pdf

Fusco, E. J., Balch, J. K., Mahood, A. L., Nagy, R. C., Syphard, A. D., & Bradley, B. A. (2022). The human–grass–fire cycle: How people and invasives co-occur to drive fire regimes. *Frontiers in Ecology and the Environment, 20*, 117–126.

Fusco, E. J., Finn, J. T., Balch, J. K., Nagy, R. C., & Bradley, B. A. (2019). Invasive grasses increase fire occurrence and frequency across US ecoregions. *Proceedings of the National Academy of Sciences of the United States of America, 116*, 23594–23599.

Galán, E., Garmendia, E., & García, O. (2022). The contribution of the commons to the persistence of mountain grazing systems under the common agricultural policy. *Land Use Policy, 117*, 106089.

Gibson, D. J., Ely, J. S., & Collins, S. L. (1999). forumThe core-satellite species hypothesis provides a theoretical basis for Grime's classification of dominant, subordinate, and transient species. *Journal of Ecology, 87*, 1064–1067.

Gotelli, N. J., & Colwell, R. K. (2011). Estimating species richness. In A. E. Magurran & B. J. McGill (Eds.), *Biological diversity: Frontiers in measurement and assessment* (pp. 39–54). Oxford University Press.

Hajra, P. K., Rao, R. R., Singh, D. K., & Uniyal, B. P. (1995). *Flora of India*. Botanical Survey of India.

Hartung, C., Lerer, A., Anokwa, Y., Tseng, C., Brunette, W., & Borriello, G. (2010). Open data kit: Tools to build information services for developing regions. In *Proceedings of the 4th ACM/IEEE international conference on information and communication technologies and development. ICTD'10* (pp. 1–12). Association for Computing Machinery.

Hill, M. O. (1973). Diversity and evenness: A unifying notation and its consequences. *Ecology, 54*, 427–432.

Housing Department, Andhra Pradesh. (2013). *Performance Audit of Implementation of INDIRAMMA Housing Scheme. Housing Department*. https://ag.ap.nic.in/GSSA/REPWEB/GSSA201213/English/Chapter3.pdf

Hu, K. S., & Chao, A. (2022). *iNEXT.3D: Interpolation and Extrapolation for three dimensions of diversity*. http://chao.stat.nthu.edu.tw/wordpress/software_download/inext-3d/

Humphries, S. E. (1993). *Environmental impact of weeds. In 10th Australian Weeds Conference/14th Asian-Pacific Weed Science Society Conference*. 1–11.

Iannone, R., Cheng, J., Schloerke, B., Hughes, E., & Seo, J. (2022). *gt: Easily Create Presentation-Ready Display Tables*. https://CRAN.R-project.org/package=gt

Ingalsbee, T. (2005). Fuelbreaks for wildland fire management: A moat or a drawbridge for ecosystem fire restoration? *Fire Ecology, 1*, 85–99.

Jackson, W. J., & Ingles, A. W. (1998). *Participatory techniques for community forestry: A field manual*. International Union for Conservation of Nature.

Kegamba, J. J., Sangha, K. K., Wurm, P., & Garnett, S. T. (2022). A review of conservation-related benefit-sharing mechanisms in Tanzania. *Global Ecology and Conservation, 33*, e01955.

Kellman, M. (1975). Evidence for late glacial age fire in a tropical montane savanna. *Journal of Biogeography, 2*, 57.

Kodandapani, N., Cochrane, M. A., & Sukumar, R. (2004). Conservation threat of increasing fire frequencies in the western ghats, India. *Conservation Biology, 18*, 1553–1561.

Kohli, A., Schmerbeck, J., & Seeland, K. (2011). *Forest fires for Domestic and Commercial Use of Ecosystem Services: A case study in Andhra Pradesh, South India*. https://dlc.dlib.indiana.edu/dlc/bitstream/handle/10535/7374/382.pdf?sequence=1

Kukrety, S. (2011). *Restoration of red sanders (Pterocarpus santalinus L.) forests for ecological and economic benefits*. Degree of Doctor of Philosophy. Graduate School of the University of Florida. http://ufdcimages.uflib.ufl.edu/UF/E0/04/27/70/00001/kukrety_s.pdf

Kulpa, S. M., Leger, E. A., Espeland, E. K., & Goergen, E. M. (2012). Postfire seeding and plant community recovery in the Great Basin. *Rangeland Ecology & Management, 65*, 171–181.

Kumar, R. (2014). Floristic richness in catchment areas of Papagini River—A tributary of river Pennar, Andhra Pradesh, India. *International Journal of Environment, 3*, 177–204.

Lehmann, C. E. R., & Parr, C. L. (2016). Tropical grassy biomes: Linking ecology, human use and conservation. *Philosophical Transactions of*

The Royal Society B Biological Sciences, 371, 20160329. https://doi.org/10.1098/rstb.2016.0329

Li, W., Knops, J. M. H., Png, G. K., Yan, X., Dong, H., Li, J., Zhou, H., & Sierra, R. D. (2020). Six-year removal of co-dominant grasses alleviated competitive pressure on subdominant grasses but dominant shrub removal had neutral effects in a subalpine ecosystem. *Global Ecology and Conservation, 23*, e01167.

Madhusudan, M. D., & Vanak, A. T. (2022). Mapping the distribution and extent of India's semi-arid open natural ecosystems. *Journal of Biogeography, 50*, 1377–1387. https://doi.org/10.1111/jbi.14471

Madhusudan, M. D., Vanak, A. T., & Kulkarni, A. (2021). *Mapping India's Open Natural Ecosystems (ONE). earthengine.app.* https://mdm.users.earthengine.app/view/open-natural-ecosystems

Manakadan, R. (2014). The grassland birds of Rollapadu wildlife sanctuary, Andhra Pradesh, India, with special reference to the impact of grazing-free enclosures. *Journal of the Bombay Natural History Society, 111*, 81.

McCain, K. N. S., Baer, S. G., Blair, J. M., & Wilson, G. W. T. (2010). Dominant grasses suppress local diversity in restored tallgrass prairie. *Restoration Ecology, 18*, 40–49.

McGlinn, D. J., Xiao, X., May, F., Gotelli, N. J., Engel, T., Blowes, S. A., Knight, T. M., Purschke, O., Chase, J. M., & McGill, B. J. (2019). Measurement of biodiversity (MoB): A method to separate the scale-dependent effects of species abundance distribution, density, and aggregation on diversity change. *Methods in Ecology and Evolution, 10*, 258–269.

Meinzen-Dick, R., Chaturvedi, R., Kandikuppa, S., Rao, K., Rao, J. P., Bruns, B., & ElDidi, H. (2021). Securing the commons in India: Mapping polycentric governance. *International Journal of the Commons, 15*, 218.

MoEFCC, Government of India. (2018). *National action plan on forest fire.* https://www.forests.tn.gov.in/app/webroot/img/NAPFF-Final%20Draft%2023.04.2018.pdf

Moghli, A., Santana, V. M., Baeza, M. J., Pastor, E., & Soliveres, S. (2022). Fire recurrence and time since last fire interact to determine the supply of multiple ecosystem services by Mediterranean forests. *Ecosystems, 25*, 1358–1370.

Mohanty, M., Dhal, N. K., Patra, P., Das, B., & Reddy, P. S. R. (2012). Phytostabilisation potential of lemon grass (Cymbopogon flexuosus (Nees ex Stend) wats) on iron ore tailings. *Journal of Environmental Science & Engineering, 54*, 147–152.

Moore, N. A., Camac, J. S., & Morgan, J. W. (2019). Effects of drought and fire on resprouting capacity of 52 temperate Australian perennial native grasses. *The New Phytologist, 221*, 1424–1433.

Mulqueeny, C. M., Goodman, P. S., & O'Connor, T. G. (2010). Landscape-level differences in fire regime between block and patch-mosaic burning strategies in Mkuzi game reserve, South Africa. *African Journal of Range and Forage Science, 27*, 143–150.

Muthumperumal, C., & Parthasarathy, N. (2009). Angiosperms, climbing plants in tropical forests of southern eastern Ghats, Tamil Nadu, India. *Check List, 5*, 92.

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature, 403*, 853–858.

NDDB. (2018). *Dairying in Andhra Pradesh—A Statistical Profile 2018.* National Dairy Development Board. https://www.nddb.coop/sites/default/files/NDDB-AP-04-05.pdf

Neeraja, U. V., Rajendrakumar, S., Saneesh, C. S., Dyda, V., & Knight, T. M. (2021). Fire alters diversity, composition, and structure of dry tropical forests in the eastern Ghats. *Ecology and Evolution, 11*, 6593–6603.

Nerlekar, A. N., Chorghe, A. R., Dalavi, J. V., Kusom, R. K., Karuppusamy, S., Kamath, V., Pokar, R., Rengaian, G., Sardesai, M. M., & Kambale, S. S. (2022). Exponential rise in the discovery of endemic plants underscores the need to conserve the Indian savannas. *Biotropica, 54*, 405–417.

Nerlekar, A. N., & Veldman, J. W. (2020). High plant diversity and slow assembly of old-growth grasslands. *Proceedings of the National Academy of Sciences of the United States of America, 117*, 18550–18556.

Oksanen, J., Guillaume Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P. R., O'Hara, R. B., Simpson, G. L., Solymos, P., Stevens, M. H. H., Szoecs, E., & Wagner, H. (2020). *vegan: Community Ecology Package. Comprehensive R Archive Network (CRAN).* https://CRAN.R-project.org/package=vegan

Paddayya, K. (1968). *Investigations into the Neolithic Culture of the Shorapur Doab, South India.* Ph.D. Deccan College.

Pedersen, T. L. (2022). Patchwork: The composer of plots. https://CRAN.R-project.org/package=patchwork

Planning Officer, C. (2018). *Handbook of statistics.* https://core.ap.gov.in/cmdashboard/Download/Publications/DHB/Chittoor%20-%202018.pdf

Planning Officer, C. (2022). *Hand Book of Statistics-Ananthapuramu District.* http://des.ap.gov.in/jsp/social/HBS%20Ananthapuramu%20District%202013.pdf

Prichard, S. J., Stevens-Rumann, C. S., & Hessburg, P. F. (2017). Tamm review: Shifting global fire regimes: Lessons from reburns and research needs. *Forest Ecology and Management, 396*, 217–233.

Published by Authority. (2019). *Extraordinary. The Gazette of India Part II-Section 3-Sub-Section (ii): 1–4.*

Pullaiah, T. (2018). *Flora of Andhra Pradesh.* Scientific Publishers.

Pullaiah, T., & Muralidhara Rao, D. (2002). *Flora of eastern Ghats Hill ranges of Southeast India.* Regency Publications.

Pullaiah, T., & Sandhya Rani, S. (1999). *Trees of Andhra Pradesh.* Scientific Publishers.

Pullaiah, T., & Sri, R. K. (2000). *Flora of Eastern Ghats Hill Ranges of Southeast India.* Regency Publications.

QGIS Development Team. (2022). *QGIS Geographic Information System. QGIS Association.* https://www.qgis.org

R Core Team. (2022). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing https://www.R-project.org/

Rajeevan, M., Unnikrishnan, C. K., Bhate, J., Niranjan Kumar, K., & Sreekala, P. P. (2012). Northeast monsoon over India: Variability and prediction. *Meteorological Applications, 19*, 226–236.

Rajendrudu, G., & Das, V. S. (1981). C_4 photosynthetic carbon metabolism in the leaves of aromatic tropical grasses—I. Leaf anatomy, CO_2 compensation point and CO_2 assimilation. *Photosynthesis Research, 2*, 225–233.

Ratnam, J., Bond, W. J., Fensham, R. J., Hoffmann, W. A., Archibald, S., Lehmann, C. E. R., Anderson, M. T., Higgins, S. I., & Sankaran, M. (2011). When is a "forest" a savanna, and why does it matter? *Global Ecology and Biogeography, 20*, 653–660.

Ratnam, J., Chengappa, S. K., Machado, S. J., Nataraj, N., Osuri, A. M., & Sankaran, M. (2019). Functional traits of trees from dry deciduous "forests" of southern India suggest seasonal drought and fire are important drivers. *Frontiers in Ecology and Evolution, 7*, 1–6. https://doi.org/10.3389/fevo.2019.00008

Ratnam, J., Tomlinson, K. W., Rasquinha, D. N., & Sankaran, M. (2016). Savannahs of Asia: Antiquity, biogeography, and an uncertain future. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences, 371*, 20150305. https://doi.org/10.1098/rstb.2015.0305

Reddy, C. S., Jha, C. S., Manaswini, G., Alekhya, V. V. L. P., Pasha, S. V., Satish, K. V., Diwakar, P. G., & Dadhwal, V. K. (2017). Nationwide assessment of forest burnt area in India using Resourcesat-2 AWiFS data. *Current Science, 112*, 1521–1532.

Riedel, N., Fuller, D. Q., Marwan, N., Poretschkin, C., Basavaiah, N., Menzel, P., Ratnam, J., Prasad, S., Sachse, D., Sankaran, M., Sarkar, S., & Stebich, M. (2021). Monsoon forced evolution of savanna and the spread of agro-pastoralism in peninsular India. *Scientific Reports, 11*, 9032.

Ripley, B., Visser, V., Christin, P.-A., Archibald, S., Martin, T., & Osborne, C. (2015). Fire ecology of C_3 and C_4 grasses depends on evolutionary history and frequency of burning but not photosynthetic type. *Ecology*, *96*, 2679–2691. https://doi.org/10.1890/14-1495.1

Sankaran, M. (2005). Fire, grazing and the dynamics of tall-grass savannas in the Kalakad-Mundanthurai Tiger Reserve, South India. *Conservation and Society*, *3*, 4–25.

Satendra, & Kaushik, A. D. (2014). *Forest fire disaster management*. National Institute of Disaster Management.

Schmerbeck, J., & Seeland, K. (2007). Fire supported forest utilisation of a degraded dry forest as a means of sustainable local forest management in Tamil Nadu/South India. *Land Use Policy*, *24*, 62–71.

Senande-Rivera, M., Insua-Costa, D., & Miguez-Macho, G. (2022). Spatial and temporal expansion of global wildland fire activity in response to climate change. *Nature Communications*, *13*, 1208.

Sharma, B. D., & Balakrishnan, N. P. (1993). *Flora of India*. Botanical Survey of India.

Sharma, B. D., & Sanjappa, M. (1993). *Flora of India*. Botanical Survey of India.

Simpson, E. H. (1949). Measurement of diversity. *Nature*, *163*, 688.

Singh, A. K. (2013). Probable agricultural biodiversity heritage sites in India: XVII. The south-central region of Eastern Ghats. *Asian Agri-History*, *17*, 199–220.

Singh, P., Karthigeyan, K., Lakshminarasimhan, P., & Dash, S. S. (2015). *Endemic vascular plants of India*. Botanical Survey of India.

Smith, M. D., van Wilgen, B. W., Burns, C. E., Govender, N., Potgieter, A. L. F., Andelman, S., Biggs, H. C., Botha, J., & Trollope, W. S. W. (2013). Long-term effects of fire frequency and season on herbaceous vegetation in savannas of the Kruger National Park, South Africa. *Journal of Plant Ecology*, *6*, 71–83.

Snyman, H. A. (2015). Fire and the dynamics of two unpalatable grass species (Cymbopogon pospischilii and Elionurus muticus) in a semi-arid climate. *African Journal of Range and Forage Science*, *32*, 1–12.

Soenarko, S. (1977). *The genus Cymbopogon Sprengel (Gramineae)*. *Reinwardtia*. http://e-journal.biologi.lipi.go.id/index.php/reinwardtia/article/view/914

Solbrig, O. T., Medina, E., & Silva, J. F. (2013). *Biodiversity and savanna ecosystem processes: A global perspective*. Springer Science & Business Media.

SPS. (2007). *MGNREGA Watershed Works Manual. Samaj Pragati Sahayog*. https://www.cse.iitb.ac.in/~karjat/waterdocs/watershed_manual_english.pdf

Staver, A. C., & Hempson, G. P. (2020). Seasonal dietary changes increase the abundances of savanna herbivore species. *Science Advances*, *6*, eabd2848. https://doi.org/10.1126/sciadv.abd2848

Strohbach, B. J., & Walters, W. (2015). An overview of grass species used for thatching in the Zambezi, Kavango east and Kavango west regions, Namibia. *Dinteria*, *35*, 13–42.

Superintendent of Census Operations. (1964). In C. Sekhar (Ed.), *District census handbook*. Government of Andhra Pradesh.

Superintendent of Census Operations. (1965). In C. Sekhar (Ed.), *District census handbook*. Government of Andhra Pradesh.

Tamilselvan, B., Sekar, T., & Anbarashan, M. (2021). Short-term girth increment and biomass changes in tree species of Javadhu hills, Eastern Ghats, Tamil Nadu, India. *Trees, Forests and People*, *4*, 100081.

Tomat-Kelly, G., Dillon, W. W., & Flory, S. L. (2021). Invasive grass fuel loads suppress native species by increasing fire intensity and soil heating. *Journal of Applied Ecology*, *58*, 2220–2230.

Vanak, A. T., Kulkarni, A., Gode, A., & Krishnaswamy, J. (2015). Extent and status of semi-arid savanna grasslands in peninsular India. In G. S. Rawat & B. S. Adhikari (Eds.), *Ecology and management of grassland habitats in India* (pp. 192–201). Wildlife Institute of India.

Vandvik, V., Klanderud, K., Meineri, E., Måren, I. E., & Töpper, J. (2015). Seed banks are biodiversity reservoirs: species-area relationships above versus below ground. *Oikos*, *125*, 218–228. http://datadryad.org/stash/dataset/doi%253A10.5061%252Fdryad.q401b

Vidaller, C., Dutoit, T., Ramone, H., & Bischoff, A. (2019). Fire increases the reproduction of the dominant grass Brachypodium retusum and Mediterranean steppe diversity in a combined burning and grazing experiment. *Applied Vegetation Science*, *22*, 127–137.

Villarreal, M. L., Iniguez, J. M., Flesch, A. D., Sanderlin, J. S., Cortés Montaño, C., Conrad, C. R., & Haire, S. L. (2020). Contemporary fire regimes provide a critical perspective on restoration needs in the Mexico-United States borderlands. *Air, Soil and Water Research*, *13*, 1178622120969191.

Whittaker, R. H. (1972). Evolution and measurement of species diversity. *Taxon*, *21*, 213–251.

Wickham, H. (2016). *ggplot2: Elegant Graphics for Data Analysis*. https://ggplot2.tidyverse.org

Wickham, H., Averick, M., Bryan, J., Chang, W., D'Agostino McGowan, L., François, R., Grolemund, G., Hayes, A., Henry, L., Hester, J., Kuhn, M., LinPedersen, T., Miller, E., Bache, S. M., Müller, K., Ooms, J., Robinson, D., Seidel, D. P., Spinu, V., … Yutani, H. (2019). Welcome to the tidyverse. *Journal of Open Source Software*, *4*, 1686. https://doi.org/10.21105/joss.01686

Wolford, W., Borras, S. M., Jr., Hall, R., Scoones, I., & White, B. (2013). Governing global land deals: The role of the state in the rush for land. *Development and Change*, *44*, 189–210.

SUPPORTING INFORMATION

Additional supporting information can be found online in the Supporting Information section at the end of this article.