

POST-FIRE REGENERATION STATUS OF TREE SPECIES IN A TROPICAL DRY DECIDUOUS FOREST OF SOUTHERN INDIA

M Sathya & S Jayakumar*

Environmental Informatics and Spatial Modelling Laboratory, Department of Ecology and Environmental Sciences, School of Life Sciences, Pondicherry University, Pondicherry 605 014, India

*jayakumar.eco@pondiuni.edu.in

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Recurrent fires in forested areas are a leading disturbance that can strongly affect the vegetation dynamics, structure and regeneration. In this study, we examined tree species regeneration in response to fire in Sathyamangalam Tiger Reserve. Fire frequency map was prepared using Landsat 5, 7 and 8 satellite images for the period between 1999 and 2015 (17 years). The study resulted in the occurrence of ten fire frequency classes along with an unburned class. For each class, three plots were randomly laid in the field, and data were collected on the three growth forms such as seedlings, saplings and trees. A total of 62 species belonging to 50 genera and 23 families were recorded. There was a negative linear correlation between the numbers of species in all three growth forms and fire frequencies. Lower fire frequency classes (F1–F4) sustained the individual density of seedlings and saplings. Friedman's two-way analysis of variance by rank showed that the number of individuals of three dominant species (*Anogeissus latifolia*, *Phyllanthus emblica* and *Tectona grandis*), in seedling and sapling, exhibited a significant difference among each other in fire frequency classes, whereas in tree form they were not significant. The species with more seedlings were *P. emblica*, *Terminalia chebula* and *Ziziphus rugosa*. This study found that even in the short fire intervals *P. emblica* species could sustain in all three life forms, thus this species can be recommended for afforestation in high fire frequency areas.

Keywords: Fire frequency, fire return interval, Sathyamangalam tiger reserve, species diversity

INTRODUCTION

Fire, a natural event, has been recognised as an important component of ecosystem structure and function for a wide variety of natural systems (Thanos et al. 1996). In recent years many studies have reported an increase in the number of wildfires and areas burned across the globe (Kodandapani et al. 2004, Dimitrakopoulos et al. 2011). Among the global issues, forest fires emerge as a major problem in the tropics. In 2000, about 350 million ha of forest land, which equals 6% of the world's geographical area, were burned (Bahuguna & Singh 2002). In India, about 9% (4713400 ha) of the total forest area was burned (Bahuguna & Singh 2002). Forest fires are a common phenomenon in India where tropical deciduous forest make up 75% of forested areas. Deciduous forests face large, highly intense wildfires because of higher total and dead fuel loads and greater horizontal fuel continuity. Most forest fires are anthropogenic in origin, i.e. deliberately started to promote agriculture, silviculture, fodder for livestock grazing and to facilitate collection of non-timber products (Murthy et al. 2006). Forest fires impact biodiversity, productivity,

regeneration and soil erosion. Damage caused by forest wildfires include plant tissue death, seed mortality, soil heating and burned bark (Van & Sheil 2005, Satyam & Jayakumar 2012). Monitoring and management of forest fire is important for India because 55% of total forest cover is prone to fire annually (Gubbi 2003). Knowledge about pre-fire, during fire and post-fire forest stand conditions is important requirements for management (Bahuguna & Singh 2002). Acquiring information about post-fire tree mortality will be helpful for ecological damage assessment and future mitigation planning.

Over centuries, humans have caused extensive damage to natural forest through anthropogenic fires (Shijo et al. 2009). The size, frequency, intensity, timing, ignition pattern and ecological effects of fires and post-fire plant regeneration patterns are key features in ecological damage assessments. In India, fire-based regeneration studies have been conducted in the Western Ghats (Mahesh 2005, Kodandapani 2013, Nandita & Sukumar 2015), Central India (Sonali 2002, Sonali & Howe 2003) and Northern

India (Munesh et al. 2012, Ashish et al. 2013). In recent years, using satellite data, studies on fire monitoring have also been carried out in the Mudumalai Tiger Reserve (Satyam & Jayakumar 2015, Satyam et al. 2015), Kerala (Badarinath et al. 2012), Bandipur National Park (Somashekar et al. 2009), Bhadra Wildlife Sanctuary (Somashekar et al. 2008), Andhra Pradesh (Reddy et al. 2012), Satna (Rajpoot & Kumar 2013) and Rajasthan (Reddy et al. 2009). Advancements in remote sensing have contributed towards finding timely, cost-effective solutions by specifying fire locations, fire intensity and area burned (Chuvienco et al. 2004, Hari & Sudhakar 2012). Sathyamangalam Tiger Reserve (STR) is a fire-prone region with a long fire history. However, the complexity of fire patterns and forest stand dynamics is still unknown in STR. Hence, the study aimed to

assess fire frequency using satellite images and to investigate the effects of fire on species diversity, stand structure and regeneration status of the tree species in STR.

MATERIALS AND METHODS

Study area

Established in March 2013 as part of the Tiger Project, Sathyamangalam Tiger Reserve (STR) is one of the largest tiger reserves in the state of Tamil Nadu. The study area is located geographically between 10° 29' to 11° 43' N latitude and 76° 50' to 77° 27' E longitude (Figure 1). STR includes 10 forest reserves belonging to two administrative forest divisions, namely the Sathyamangalam division and the Hasanur division (FMP 2010).

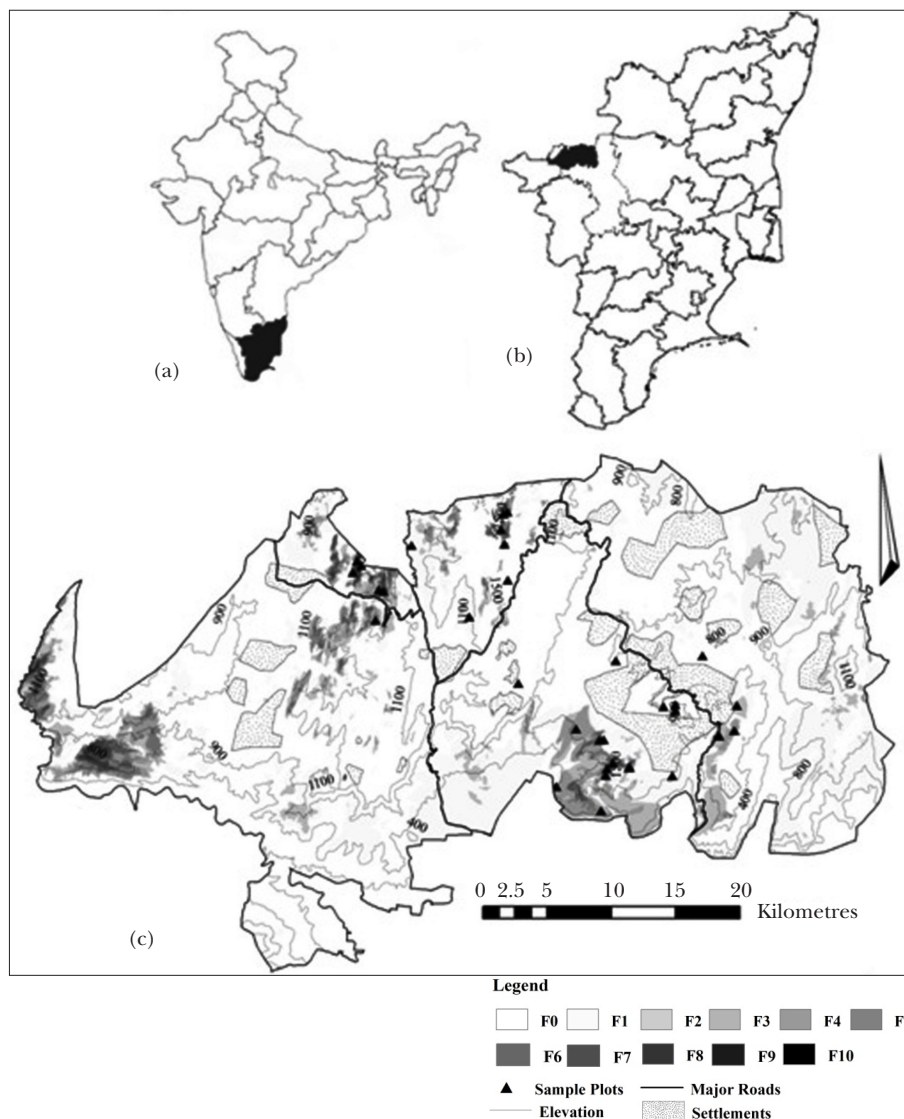


Figure 1 (a) India, (b) Tamil Nadu and (c) fire frequency map of study area for 17 years (1999–2015)

Topography of the reserve is highly variable with plains, slopes, hills, streams and rivers. It receives an average annual rainfall of 850 mm and mean temperature ranges from 21°C to 28 °C. The Bhavani and Moyar rivers run year-round through the reserve (Nishanth et al. 2012). Though the reserve lies in a rain shadow region, the major source of rain (70%) is received from north-east monsoon (October–December). It also experiences a dry spell between January and April, which coincides with the accumulation of dry leaf litter and more fire incidences. Intermittent showers occur between May and September. The STR is predominantly a tropical dry forest which includes dry thorn, dry deciduous, semi-evergreen and savanna forests. The reserve is very rich in flora and fauna. There are 34 settlements inside the reserve, of which 16 are non-tribal hamlets and 18 are tribal villages that depend on forests and forest products for their livelihood. They practice agriculture alongside the streams, rivers and plains (Balasubramanian et al. 2011).

Fire frequency mapping

Fire maps were prepared between 1999 and 2015, using satellite images. Landsat 5 Thematic Mapper (TM), Landsat 7 Enhanced Thematic Mapper (ETM+) and Landsat 8 Operational Land Imager (OLI) imagery pertaining to the above period (Figure 2) were obtained from the United States Geological Survey's (USGS) Earth Explorer. Based on fire history, download of satellite data was restricted to January–May in each year. The study area was extracted from the images and classified into burned (F1 – F10) and unburned (F0) classes using supervised classification with maximum likelihood algorithm. Further, the maps were cross-checked with known fire locations, found in the Sathyamangalam district forest office and NASA-FIRMS (Fire Information for Resource Management System) data. All the raster maps were converted to vector, assigned a unique value for burned and unburned areas, and again converted to raster. Finally, all the raster images were combined to form a fire frequency map using raster calculator in ERDAS Imagine 2011 software.

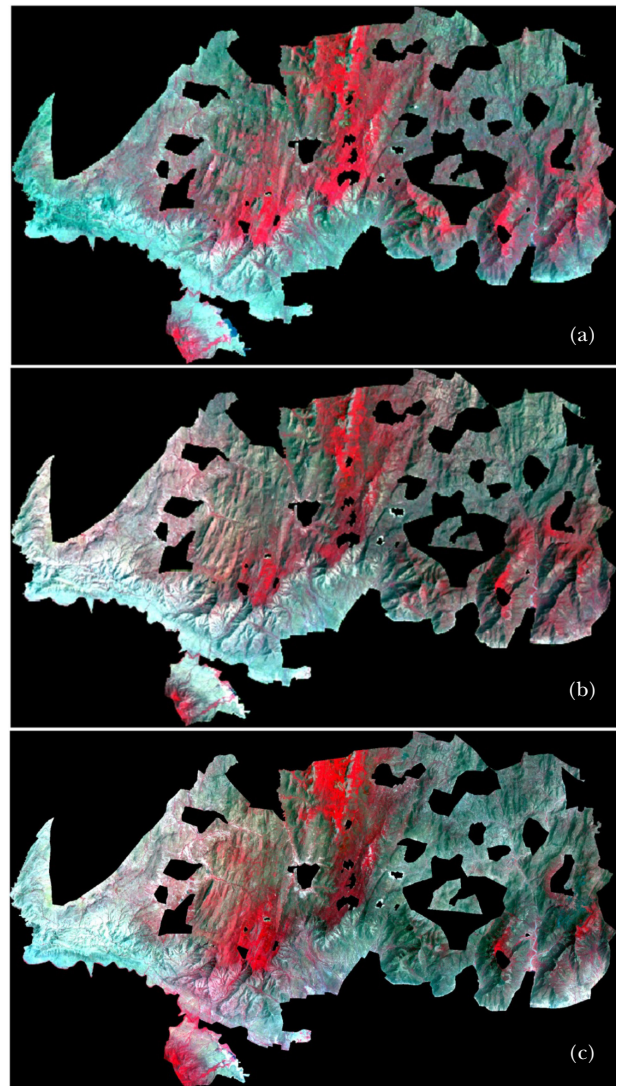


Figure 2 Standard false-colour composite images of the study area, (a) Landsat 5 Thematic Mapper 7 April 1999, (b) Landsat 7 Enhanced Thematic Mapper plus 21 February 2003 and (c) Landsat 8 Operational Land Imager 31 March 2014

Floristic data collection

Floristic data were collected from 33 plots (0.1 ha each) that were equally distributed in all fire frequency classes (F1–F10) and control (F0) class (Figure 1). Data was collected from three 0.1 ha square plots for each class ($n = 33$). Thus, the location for each plot was randomly distributed on the fire frequency map with respect to each fire frequency and control class. Each plot was located in the field based on geographical

coordinates and identified with the help of Global Positioning System (GPS). Plants were identified to species following Fyson (1915), Nair and Henry (1983) and Matthew (1983, 1999). In each plot, all trees with > 10 cm girth at breast height (GBH) were measured at 1.3 m above ground. For saplings, four 5 m × 5 m subplots were laid on all four corners of the 0.1 ha plot. For seedlings, eight 1 m × 1 m subplots were laid on opposite corners of the four 5 m × 5 m subplots. All trees with > 1 and ≤ 10 cm GBH were categorised as saplings. Trees with ≤ 1 cm GBH or < 1.3 m height were defined as seedlings. Finally, the taxonomic ranks of each species were revised using Angiosperm Phylogeny Group (APG) III Classification (Bremer et al. 2009).

Data analysis

Fire return interval was calculated using fire frequency maps. The total numbers of species in each frequency class and in the three growth stages (tree, sapling and seedling) were calculated. Stem density and basal area were calculated for the trees and subjected to one-way ANOVA test. Tukey's test was used to compare the differences in means. The Shannon-Wiener index and Simpson's dominance index were used to calculate diversity. Regression analysis was conducted to explore the basal area, stem density and number of species for seedlings, in relation to fire frequency. GBH was converted to diameter at breast height (DBH). Stem size class analysis was also performed for all fire frequencies. Percentage of abundance was calculated for the 10 most common tree, sapling and seedling species. Friedman's two-way analysis of variance by rank was used to analyse significant differences in the distribution of three dominant species of the forest in tree, sapling and seedling with fire frequencies. The significance threshold was set at 0.05.

RESULTS

Fire frequency

Fire frequency mapping revealed that the study site was burned a maximum of 10 times during the 17 years (1999 to 2015). The mean fire-return interval for STR was 4.7 years.

Species richness

A total of 62 tree species were recorded from 50 genera and 23 families from all fire frequency classes and control plot. Fabaceae was the dominant family with 14 species, followed by Euphorbiaceae and Rutaceae with five species each, and Rubiaceae with four species. The dominant genera were *Ficus*, *Terminalia* and *Zizyphus* with three species each followed by *Carissa*, *Cassine*, *Dalbergia*, *Grewia*, *Phyllanthus* and *Senegalia* with two species each. *Phyllanthus emblica* and *Pterocarpus marsupium* were the two species found in all 10 fire frequency classes and in control class. *Anogeissus latifolia*, *Tectona grandis* and *Terminalia chebula* were present in control as well as in seven frequency classes. Of the 62 species, 30 species were found only in one frequency class.

Species diversity

Shannon-Wiener and Simpson's dominance indices followed a decreasing trend (Figure 3) among the control and other fire frequency classes. The Shannon-Wiener and Simpson's dominance values for the control plot were 2.92 and 0.93 respectively. The diversity values for all fire frequency classes were lower than that of the control class. An uneven trend was noticed in both diversity values between F1 and F6 while F7 to F10 displayed an evenly decreased trend in both Shannon-Wiener and Simpson's dominance indices.

Population structure

Plots that experienced one, two and three fires had basal areas greater than control plot, 2.52 m³ ha⁻¹ (Table 1), however these mean values were not notably different (Figure 4a). Plots burned more than six times over the study period showed decreasing basal areas, dropping to zero in F10. The basal area per ha for F0 was 2.52 m². There was an uneven trend between F1 and F6 (Table 1). The unburned plots recorded more plant species present at seedling (12) and tree (30) stages than that in plots with fire frequencies above 1. However, the highest number of species in the sapling stage was recorded for F2 (15 species), followed by F1 (12). Among the burned plots, for the seedling stage, F1 had the most species (8), while F8 and F10 had

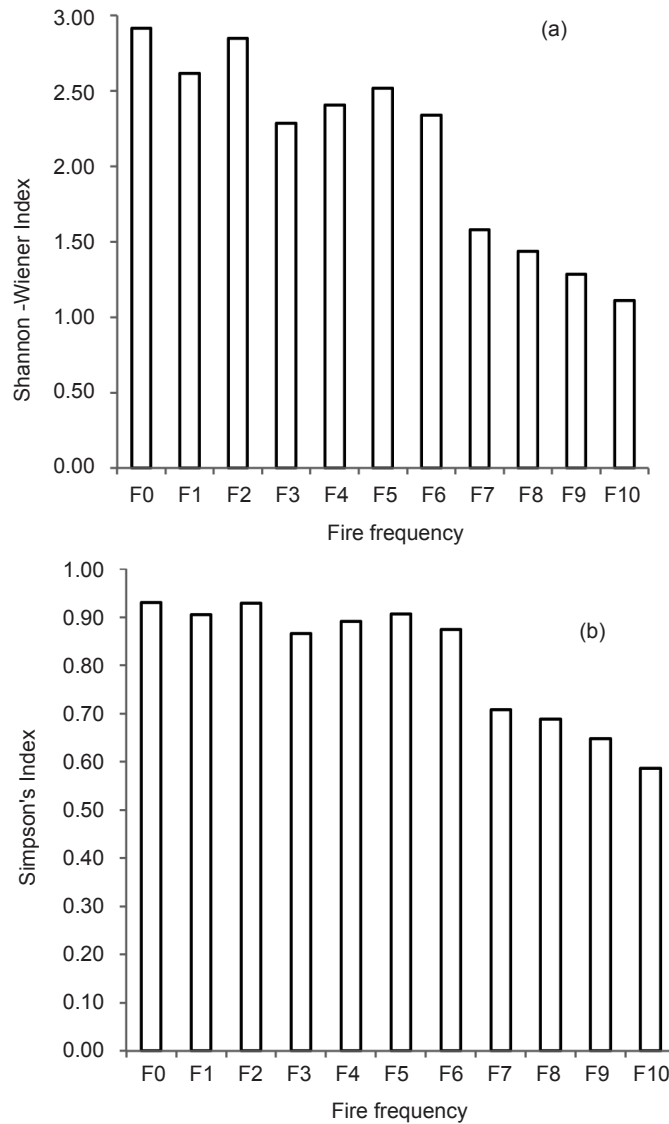


Figure 3 (a) Shannon-Wiener index and (b) Simpson's dominance index for stems present in different burned frequency and control class

Table 1 Fire return interval for species richness, basal area and tree density in relation to fire frequency from 1999 to 2015 in Sathyamangalam Tiger Reserve

Fi	Year of fire occurrence	FRI (year)	Species richness	Basal area (m ² ha ⁻¹)	Tree density (ha ⁻¹)
F0	-	0	30	2.52 ± 1.29	433 (55.08)
F1	2009	16	21	3.20 ± 3.33	210 (137.48)
F2	2005, 2014	8	21	5.68 ± 5.45	220 (30)
F3	2005, 2009, 2014	5.5	17	5.40 ± 7.37	150 (98.49)
F4	1999, 2005, 2007, 2012	4	16	1.91 ± 1.85	183 (60.28)
F5	1999, 2000, 2005, 2007, 2011	3.5	17	1.66 ± 0.67	113(11.54)
F6	1999, 2000, 2004, 2006, 2007, 2011	2.5	16	2.15 ± 1.09	107(15.28)
F7	1999, 2000, 2003, 2005, 2006, 2007, 2011	2.2	8	1.14 ± 0.21	67(5.77)
F8	1999, 2001, 2003, 2005, 2007, 2008, 2009, 2012	2	6	0.51 ± 0.18	27(20.82)
F9	1999, 2001, 2002, 2004, 2005, 2007, 2009, 2010, 2011	1.7	5	0.38 ± 0.24	27(15.28)
F10	1999, 2001, 2002, 2004, 2005, 2007, 2009, 2010, 2012, 2013	1.5	4	0.0	0.0

Fi = fire frequency class, where i denotes the number of fire occurrences from 1999 to 2015 (averaged over three plots), data is expressed as mean values ± standard deviation (SD)

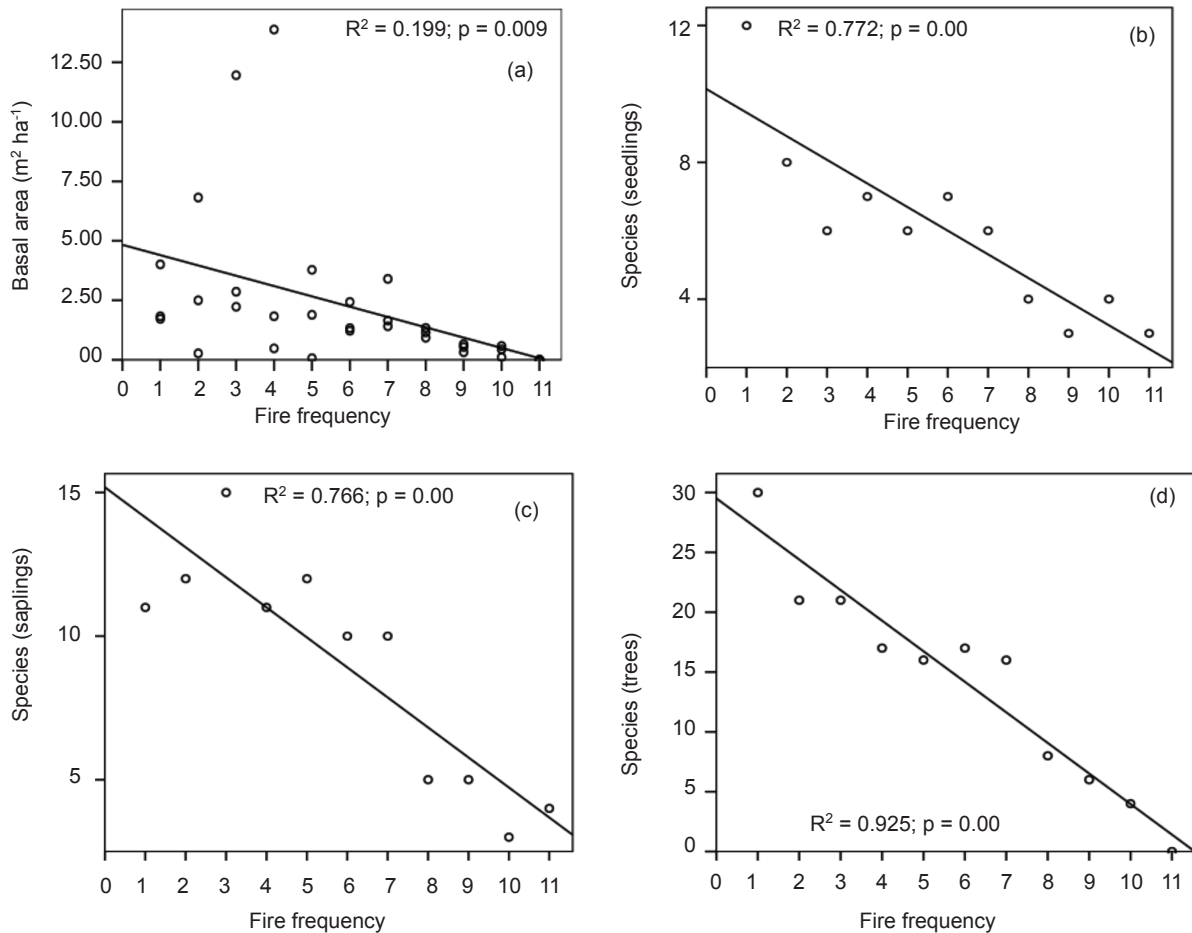


Figure 4 Forest fire frequency in relation to (a) basal area and species richness, (b) seedlings, (c) saplings and (d) trees

the fewest species (3 each). Among the burned plots, for the tree stage, F1 and F2 had the most species (21 each) while F9 had the fewest species (4). The total number of species in each fire frequency class for all three forms (seedlings, saplings and trees) displayed a strong negative linear relationship ($R^2 = 0.925$, $p = 0.000$) with fire frequencies (Figures 4b–d).

Girth class

In F0, trees with DBH < 20 cm comprised 95% of the stand, with stand density recorded at 92 trees ha^{-1} for the 1–10 cm DBH class and 25 trees ha^{-1} for the 10–20 cm DBH class. There were no individuals larger than 50 cm DBH in these unburned plots (Figure 5). F1, F2 and F4 recorded higher stand density at 1–10 cm DBH and the remaining frequencies recorded maximum density at 10–20 cm DBH.

Regeneration

The highest numbers of seedlings (29583), saplings (4233) and trees (433) ha^{-1} were recorded in F0 (Figure 6). Among the burned plots, the highest number of seedlings (21250 ha^{-1}) was recorded for F3 followed by F1 and F4 (14583 ha^{-1} each). F9 recorded the lowest density of seedlings (1667 ha^{-1}). With regard to saplings, maximum density (3700 ha^{-1}) was recorded in F2, followed by F4 (3433 ha^{-1}) and F3 (2900 ha^{-1}). The fewest saplings were recorded in F8 (300 ha^{-1}). The highest number of trees was recorded in F2 (220 ha^{-1}) followed by F1 (210 ha^{-1}) and F4 (183 ha^{-1}). F8 and F9 had the fewest trees (27 ha^{-1} each). There were no trees in F10.

Anogeissus latifolia, *P. emblica* and *T. grandis* were the three most common species. *Anogeissus latifolia*, the predominant species contributed 1290 trees and 3267 saplings per ha, but recorded

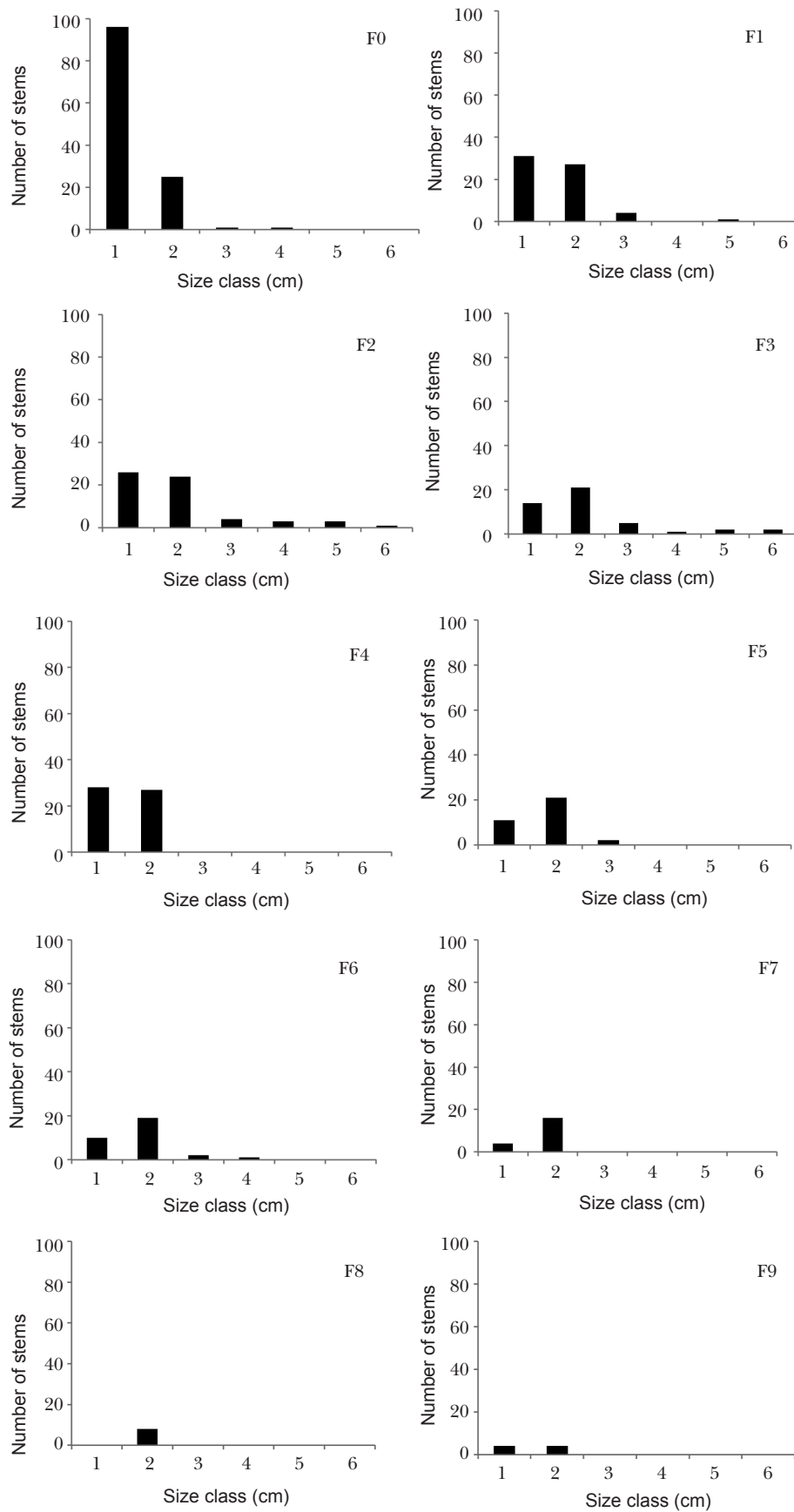


Figure 5 Stem diameter distribution in unburned (F0) and burned classes (F1–F9); the minimum diameter class was 1–10 cm DBH, increased by 10 cm interval

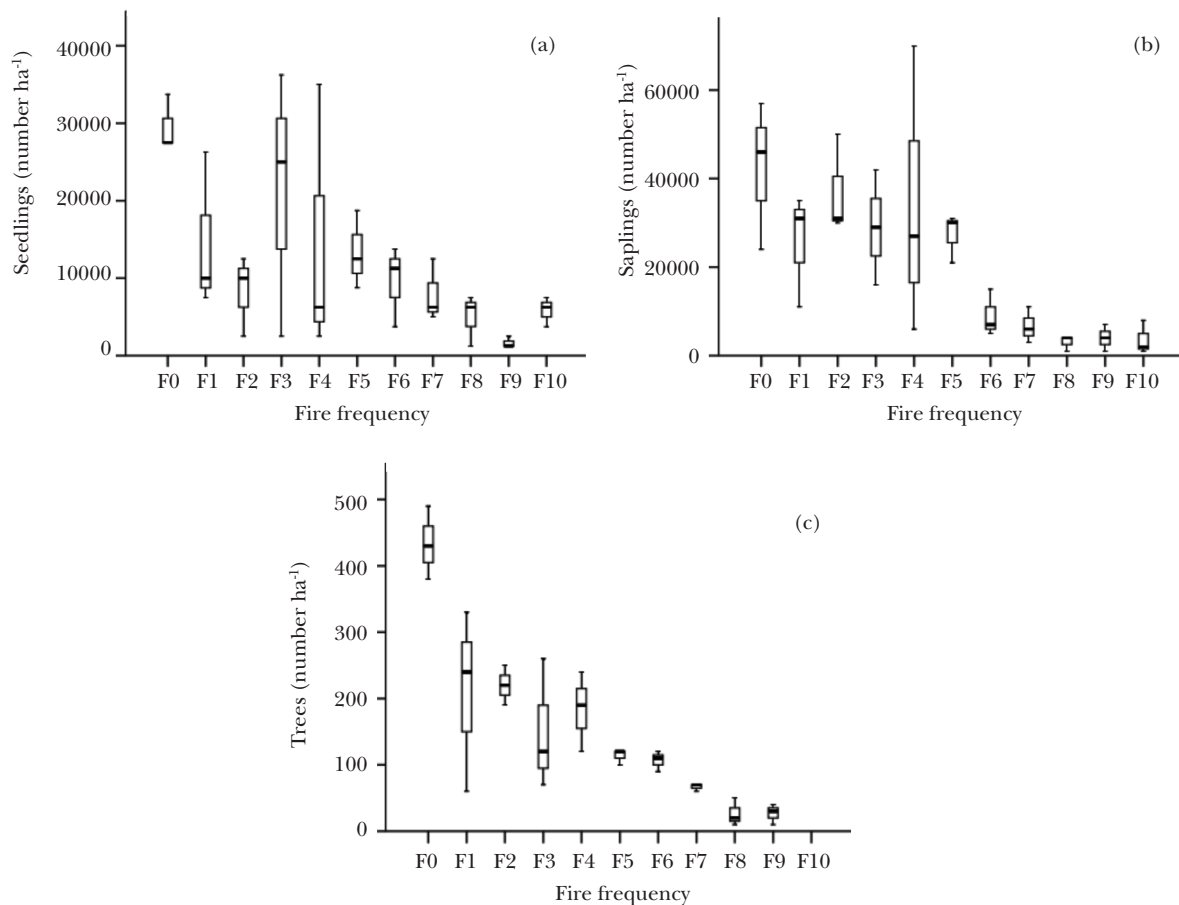


Figure 6 Box plot showing number of individuals recorded per hectare, (a) seedlings, (b) saplings and (c) trees with error bars representing standard deviations

only 100 seedlings ha⁻¹ (Figure 7). For *P. emblica*, 5133 seedlings, 5900 saplings and 900 trees ha⁻¹ were recorded. For *T. grandis*, 767 seedlings, 6067 saplings and 860 trees ha⁻¹ were recorded. Seedlings of *A. latifolia* were completely absent in all burned plots except F2 (Figure 7a). *Anogeissus latifolia* saplings were absent in F0, F3 and F8–F10. Trees of this species were absent in the higher fire frequency classes (F8–F10). *Phyllanthus emblica* seedlings were absent in F2 and F5, and present in all other fire frequency classes. The saplings and trees of *P. emblica* were absent in F1 and F10 respectively. Seedlings of *T. grandis* were absent in F2, F3 and F7–F10. *Tectona grandis* saplings were absent in F2, F3 and F10 while trees of this species were absent in F2, F8 and F10. For all three growth stages, these three most common species did not show any trend in relation to fire frequency.

Anogeissus latifolia contributed the highest number of saplings in F4 (1000) followed by F1 (833) and the fewest at F7 (100). In the case of trees, *A. latifolia* were found more in F4 (370)

followed by F0 (330) and the fewest were found at F3 (30). The most *P. emblica* seedlings were present at F7 (1100) while the fewest seedlings were recorded at F8 and F10 (233). The highest number of trees for *P. emblica* were recorded at F2, F4 and F8 (133 each) while the fewest were found at F1 and F9 (33 each). *Tectona grandis* contributed the most seedlings (233) in F0 followed by F4 and F5. For the sapling stage, *T. grandis* contributed an equal number (1100) in F0, F4, F5 and F6. For the tree stage, the highest number of individuals was recorded for *T. grandis* at F6 (200) followed by F1, F4 (170 each) while the fewest trees were found at F3 and F7 (30 each).

The regeneration pattern of these dominant species through Friedman's two-way analysis of variance by rank revealed that the three dominant species were significantly different from each other in seedlings and saplings form,

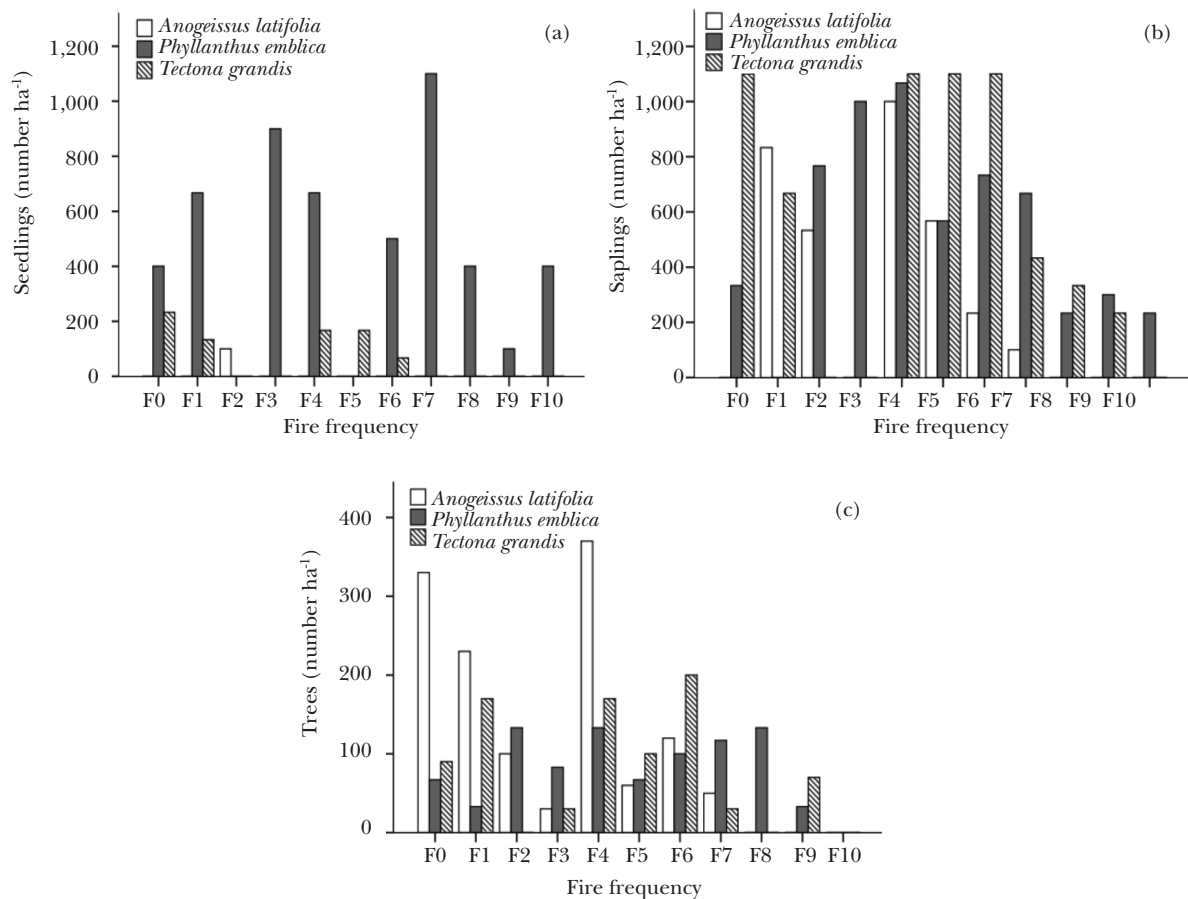


Figure 7 Comparison of top three dominant species in different growth forms, (a) seedlings, (b) saplings and (c) trees

but not significantly different in tree form in the fire frequency classes at 0.05 significance level (Figures 4b–d). When compared with control class, the tree form of *A. latifolia* was significant at higher fire frequency classes (F5, F7–F10), while the other two dominant species were not significant. In seedling and sapling forms, all three dominant species were not significant in the control class. While *A. latifolia*, and *T. grandis* were dominant in total number of individuals, their contribution in seedlings was very low (Table 2.). The seedling growth form was dominated by *P. emblica*, *T. chebula* and *Z. rugosa* (Figure 8) and their contribution of individuals in seedlings were in the order of *P. emblica* > *T. chebula* > *Z. rugosa*. *Terminalia chebula* individuals were maximum at F3 (778) followed by F4 (667) and F5 (444) and minimum at F8–F10 (111 each). Seedlings of *T. chebula* were completely absent at F1, F2 and F7. *Ziziphus rugosa* was absent at F2, F3 and F8–F10. It contributed more in F4 (500) followed by F5, F7 with 444 each and the lowest at F0 (222).

DISCUSSION

The fire frequency maps of STR for 17 years revealed that the mean fire return interval for STR was 4.7 years. Previous studies have reported the fire return interval for STR as 44.1 years (Kodandapani et al. 2012) and 2–16 years (Kodandapani 2013).

The floristic composition of STR is dominated by *A. latifolia*, *P. emblica*, *T. grandis* and *P. marsupium*, which is similar to that reported for other tropical dry deciduous forests (Balch et al. 2013) including Mudumalai Tiger Reserve (MTR) (Kodandapani et al. 2008a, Satyam & Jayakumar 2015). The tree stand density and species richness were highest at F0, gradually decreasing between F1 to F4 to low numbers from F6 to F9. There was no stand at F10. This observation substantiates the point that repeated fires generally cause a reduction in species richness and stand density (Kodandapani et al. 2008b, Jhariya et al. 2012).

Table 2 Ten most common plant species in the seedling, sapling and tree stage in plots experiencing fire frequencies of 0–10 times (F0–F10) during 1999–2015

Species	Family	Plant density (number of individuals ha ⁻¹) in fire frequency plots										
		F0	F1	F2	F3	F4	F5	F6	F7	F8	F9	F10
Seedling												
<i>Phyllanthus emblica</i>	Phyllanthaceae	400	667	n/a	900	667	n/a	500	1100	400	100	400
<i>Terminalia chebula</i>	Combretaceae	230	n/a	n/a	778	667	444	230	n/a	111	111	111
<i>Ziziphus rugosa</i>	Rhamnaceae	222	320	n/a	n/a	500	444	250	444	n/a	n/a	n/a
<i>Tectona grandis</i>	Lamiaceae	233	133	n/a	n/a	167	167	67	n/a	n/a	n/a	n/a
<i>Canthium dicoccum</i>	Rubiaceae	n/a	315	400	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
<i>Chloroxylon swietenia</i>	Rutaceae	n/a	n/a	420	n/a	n/a	420	n/a	n/a	n/a	n/a	n/a
<i>Ziziphus jujuba</i>	Rhamnaceae	n/a	300	n/a	n/a	n/a	n/a	245	n/a	n/a	n/a	n/a
<i>Pterocarpus marsupium</i>	Fabaceae	n/a	n/a	n/a	200	n/a	220	n/a	n/a	100	100	n/a
<i>Syzygium cumini</i>	Myrtaceae	n/a	n/a	n/a	600	n/a	n/a	n/a	n/a	n/a	n/a	n/a
<i>Cassia fistula</i>	Fabaceae	256	n/a	150	n/a	n/a	330	n/a	n/a	n/a	n/a	n/a
Sapling												
<i>Phyllanthus emblica</i>	Phyllanthaceae	333	n/a	767	1000	1067	567	733	667	233	300	233
<i>Anogeissus latifolia</i>	Combretaceae	n/a	833	533	n/a	1000	567	233	100	n/a	n/a	n/a
<i>Tectona grandis</i>	Lamiaceae	1100	667	n/a	n/a	1100	1100	1100	433	333	233	n/a
<i>Terminalia chebula</i>	Combretaceae	n/a	n/a	n/a	950	423	n/a	350	n/a	190	n/a	190
<i>Ziziphus rugosa</i>	Rhamnaceae	450	320	n/a	n/a	495	623	325	233	n/a	n/a	n/a
<i>Albizia amara</i>	Fabaceae	n/a	n/a	315	800	547	368	254	n/a	n/a	n/a	n/a
<i>Erythroxylum monogynum</i>	Erythroxylaceae	950	547	430	667	n/a	760	n/a	n/a	n/a	n/a	n/a
<i>Catunaregam spinosa</i>	Rubiaceae	820	350	n/a	n/a	1100	n/a	n/a	n/a	n/a	n/a	n/a
<i>Hardwickia binata</i>	Fabaceae	n/a	n/a	n/a	n/a	410	560	362	n/a	n/a	n/a	n/a
<i>Canthium dicoccum</i>	Rubiaceae	n/a	530	767	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Tree												
<i>Anogeissus latifolia</i>	Combretaceae	330	230	100	30	370	60	120	50	n/a	n/a	n/a
<i>Phyllanthus emblica</i>	Phyllanthaceae	67	33	133	83	133	67	100	117	133	33	n/a
<i>Tectona grandis</i>	Lamiaceae	90	170	n/a	30	170	100	200	30	n/a	70	n/a
<i>Pterocarpus marsupium</i>	Fabaceae	100	100	150	150	100	100	200	167	100	100	n/a
<i>Canthium dicoccum</i>	Rubiaceae	100	450	300	n/a	n/a	n/a	100	n/a	n/a	n/a	n/a
<i>Albizia amara</i>	Fabaceae	n/a	n/a	200	400	500	200	n/a	n/a	n/a	n/a	n/a
<i>Ziziphus rugosa</i>	Rhamnaceae	350	200	n/a	150	400	100	n/a	100	n/a	n/a	n/a
<i>Terminalia chebula</i>	Combretaceae	400	n/a	n/a	250	n/a	100	200	n/a	100	200	n/a
<i>Erythroxylum monogynum</i>	Erythroxylaceae	400	300	150	300	n/a	100	n/a	n/a	n/a	n/a	n/a
<i>Hardwickia binata</i>	Fabaceae	n/a	100	n/a	n/a	400	250	100	n/a	n/a	n/a	n/a

n/a = not applicable

The Shannon-Wiener and Simpson's diversity indices also follow a similar pattern in different fire frequency classes. This observation is similar to other studies conducted in a dry deciduous forest of Bolivia and in MTR, which reported a reduction in species diversity with increasing fire frequencies (Kennard et al. 2002, Satyam and Jayakumar 2015). The stem size class distribution also showed that high-intensity fires led to complete deforestation. The stand density and species composition of seedlings and saplings follow a similar trend in all fire frequencies. West et al. (1981) and Jhariya et al. (2012)

reported that increasing fire frequency might reduce adaptive species. The higher density of seedlings and saplings in low fire frequency sites may be due to the positive effects of fire, i.e. releasing chemicals and nutrients to the soil and stimulating the germination of certain species (Goldammer 1988, Joshi 1990, Sonali 2002, Kodandapani 2012).

The top three dominant species *A. latifolia*, *P. emblica* and *T. grandis* retained their position at both the tree and sapling stage but at the seedling stage, *T. chebula* and *Z. rugosa* replaced *A. latifolia* and *T. grandis*. *Phyllanthus emblica* was

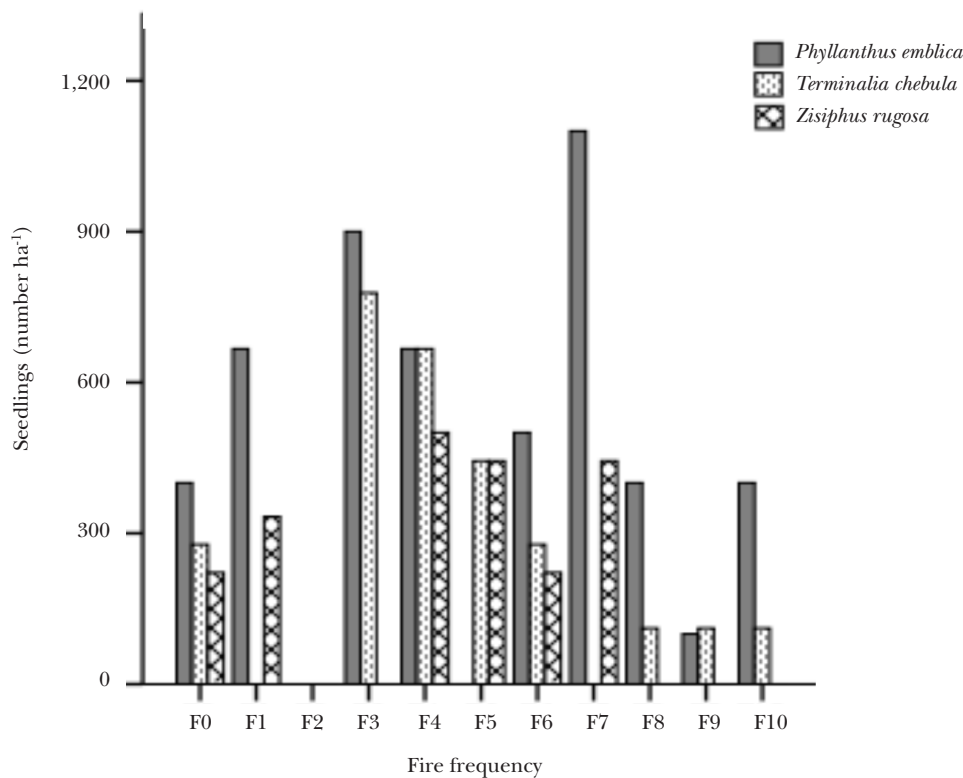


Figure 8 Comparison of top three species contributing highest number of seedlings

the dominant species in all three growth forms. The long dormancy period of *P. emblica* seeds may be the reason for withstanding high frequency fires, as its seeds were naturally dispersed by ruminants (Mawalagedera et al. 2014). The results differed from that of a previous report for MTR where *A. latifolia*, *Temnora crenulata*, and *T. grandis* were the top three dominant species in all three growth forms (Satyam & Jayakumar 2015). However, Munesh et al. (2012) reported that the seedlings and saplings of *A. latifolia* were severely affected by fire in Garhwal Himalayas. The deep and spreading root system of this species can withstand drought and other adverse conditions. Most tropical fires are surface fires (Shijo et al. 2009, Balch et al. 2013) and this will cause more damage to small-statured trees. However, in STR, a notable recruitment trend was seen in small-statured trees such as *Z. rugosa*, *Ziziphus jujuba*, *Erythroxylum monogynum* and *Carissa carandus* (Table 2).

CONCLUSIONS

The Landsat satellite data were found to be useful for mapping fire-burned areas. In STR, fire occurred a maximum of 10 times during 1999–2015 (17 years). The mean fire return interval

was 4.7 years. Increasing fire frequency decreased species diversity, stand structure, basal area and regeneration status of the tree species. The number of seedlings, saplings and trees recorded in control was higher than the fire-burned areas. However, areas with low fire frequencies (F1–F4) showed enhanced regeneration compared to control. Frequent fires (F10) totally removed trees. *Anogeissus latifolia*, *P. emblica* and *T. grandis* were the dominant tree species. Nevertheless, higher numbers of seedlings were contributed by *P. emblica*, *T. chebula* and *Z. rugosa*. Although *A. latifolia* was the dominant species having more tree individuals, its regeneration capacity was very low in STR. Planting saplings of *A. latifolia* and *P. emblica* may help increase tree density in high fire frequency areas. The present study forms a baseline data with regard to fire and tree dynamics that may facilitate future in-depth studies.

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