

Tree Diversity Changes over a Decade (2003-2013) in Four Inland Tropical Dry Evergreen Forest Sites on the Coromandel Coast of India

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Abstract

Forest tree diversity inventory and its periodical monitoring are important to understand changes in tree population structure and to provide information useful for biodiversity conservation and reserve management. In a long-term forest dynamics program in Indian tropical dry evergreen forest, this communication deals with tree diversity changes at decadal interval. The initial inventory of tree diversity was carried out in 2003, in four tropical dry evergreen forest sites - (much disturbed sites Shanmuganathapuram - SP and Araiypatti - AP and moderately disturbed sites - Karisakkadu - KR and Maramadakki - MM) on the Coromandel Coast of peninsular India, by establishing four 1ha permanent plots, one in each site. In 2013, the four plots were re-inventoried for tree diversity (≥ 10 cm gbh) changes which yielded 56 species from 46 genera and 26 families. The studied forest sites are threatened by disturbance due to multiple reasons; cutting of trees inside of the forest, grazing by goats, construction of temple approach road, and some aspects cultural attachment of local people like constructing new, additional structures of temple by denuding a portion of forest etc.. Tree species richness over a decade increased by four species in site SP, two species in site AP, and one species in site KR, but decreased by one species in site MM. Tree density decreased drastically by 480 (28.92%) and 102 (12.63%) stems ha^{-1} respectively in sites SP and AP, but moderately increased by 82 (12.09%) stems ha^{-1} in site KR and 26 (3.46%) stems ha^{-1} in site MM. Tree basal area declined in site KR from 21.6 m^2 to 20.26 $\text{m}^2 \text{ha}^{-1}$ and in site SP from 21.1 to 20.38 $\text{m}^2 \text{ha}^{-1}$, but increased from 19.1 m^2 to 19.43 m^2 and from 15.5 to 18.63 $\text{m}^2 \text{ha}^{-1}$ in sites AP and MM respectively. Three tree species (*Allophylus serratus*, *Maytenus emarginata* and *Ehretia pubescens*) were lost out of the 57 species recorded in 2003, and two species (*Jatropha gossypifolia* and *Streblus asper*) were new additions in ten years. The long-term forest monitoring data will be valuable to understand forest dynamics and for conservation and management of this and similar tropical forests.

Key Words: decadal change, permanent plot, re-census, tree species density, tropical dry evergreen forest

Introduction

Tropical forests cover just ten per cent of the earth's surface but contain more than 50% of plant species (Mayaux et

al. 2005). They act as reservoirs of biodiversity, timber, medicinal plants, and oxygen, and play a critical role in watershed protection (Richards 1996). As in all other ecosystems, tropical forests too face a wide range of disturbances

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of variable duration, intensity and frequency. Long-term vegetation monitoring has been undertaken in various tropical forests around the world (Rees et al. 2001; Laurance et al. 2004; Weckel et al. 2006; Marimon et al. 2012; van den Berg et al. 2012; Ge et al. 2013), and an increasing number of studies have shown that forests have undergone dynamic widespread directional shifts in composition and structure (Enquist and Enquist 2011; Feeley et al. 2011; Peng et al. 2011; Fauset et al. 2012; Kucbel et al. 2012). Long-term monitoring of permanent forest plots provides data to assess aboveground standing biomass stock and C dynamics and assesses the response of the forest to the environmental drivers such as elevated temperature, CO₂ fertilization, increase in incoming solar radiation and nitrogen enrichment (Bhat and Ravindranath 2011). Monitoring can also aid in collecting reliable scientific information on habitat composition, structure and dynamics and in evaluating existing management approaches and their impacts on forest ecosystems (Baithalu et al. 2013). Long-term studies of tree population dynamics are critical for understanding the conservation need of tropical forest ecosystems (Hubbell and Foster 1990; Condit 1995; Sheil et al. 2000). Several long-term studies of tree population dynamics are available in both tropical and temperate forests (Lieberman and Lieberman 1987; Kochummen and Manokaran 1987; Foster and Hubbell 1992; Sukumar et al. 1992; Balslev and Korning 1994; Bhat et al. 2000; Lwanga et al. 2000; Battles et al. 2003; Takahashi et al. 2003; Fashing et al. 2004; Kariuki et al. 2006; Baithalu et al. 2012, 2013). However, most studies on forest composition worldwide have been conducted on rain forest vegetation, and there are relatively few data available on tropical dry forest communities (Sussman and Rakotozafy 1994).

Tropical forests in peninsular India are disappearing at an alarming rate due to anthropogenic activities and are replaced by forests composed of inferior species or their land use pattern is changed (Chittibabu and Parthasarathy 2000; Parthasarathy 1999). Human interferences ultimately affect the structure, composition, species diversity and ecosystem services of the forests (William-Linera and Lorea 2009) and natural disturbances determine forest dynamics and tree diversity at the local and regional scales (Clark 1992; Young and Hubbell 1991; Attiwill 1994; Burslem and Whitmore 1999; Hubbell et al. 1999; Masaki et al.

1999; Sheil 1999; Ramírez-Marcial et al. 2001). The knowledge on forest structure, species composition and diversity at different levels of human disturbance would facilitate the caution and implementation of more effective conservation measures (Htun et al. 2011; Anbarashan and Parthasarathy 2012). Understanding the response of forest vegetation to different intensities of human disturbance would help to identify where conservation efforts should be given priority, and thereby enable the efficient use of limited conservation funds (James et al. 2001; Bhuyan et al. 2003; Htun et al. 2011).

Tropical forests contain as much as 40% of the C stored as terrestrial biomass (Dixon et al. 1994) and account for 30 to 50% of terrestrial productivity (Grace et al. 1994). Therefore, a small perturbation in a biome could result in a significant change in the global C cycle. Aboveground biomass (AGB) is a useful tool for assessing changes in forest structure (Brown et al. 1999) and an essential aspect of studies on carbon cycle (Cairns et al. 2003; Keller et al. 2001; Ketterings et al. 2001). In the last few decades, many aspects of tree inventory such as pervasive increases in biomass, tree turnover and growth rates of tropical forests have been widely reported (Murphy et al. 2013).

In a long-term forest environmental monitoring program in Indian tropical dry evergreen forest (TDEF), this publication is concerned with changes in tree diversity at the first decadal interval. Tropical dry evergreen forest is a tree-dominated, closed canopy forest type, which is restricted to south Asia on the Coromandel Coast of India, north-eastern Sri Lanka, southwest China and north-eastern Thailand and in Caribbean Islands and south coast of Jamaica (Parthasarathy et al. 2008). The tropical dry evergreen forests on the Coromandel Coast of India, composed of indigenous species, are mostly sacred groves preserved as a result of the religious belief of the local people (Anbarashan and Parthasarathy 2012). Various anthropogenic pressures due to developmental activities, deforestation, exploitation of forest resources and increase in human population have threatened many sacred groves. In last two decades many studies have been conducted on biodiversity and ecology of TDEF sites but long-term studies are very few (Baithalu et al. 2012, 2013; Mani and Parthasarathy 2009; Parthasarathy and Venkateswaran 2005) as compared to biodiversity assessment. The main objectives of the pres-

ent study are to determine changes in tree population and tree AGB in four inland tropical dry evergreen forest sites on the Coromandel Coast of India, a decade after the initial inventory.

Materials and Methods

Study area

The research was carried out in four inland tropical dry evergreen forest sites - (Araiypatti - AP ($10^{\circ}18'12''$ N latitude and $78^{\circ}58'14''$ E longitude), Karisakkadu - KR ($10^{\circ}27'2.44''$ N

and $79^{\circ}0'27''$ E), Maramadakki - MM ($10^{\circ}29'1.40''$ N and $79^{\circ}0'56''$ E) and Shanmuganathapuram - SP ($10^{\circ}18'3.33''$ N and $78^{\circ}56'15.26''$ E), which are located 40 km west of the eastern coast in Pudukottai district ($9^{\circ}50''$ and $10^{\circ}40''$, $78^{\circ}25''$ and $79^{\circ}15''$) of Tamil Nadu, south India (Fig. 1). The forest area of the four study sites ranges from 1.5 ha (in sites AP and SP) to 2.5 ha (in KR and MM). All the four study sites are sacred groves or temple forests and are protected on the religious belief of the local people. Sites MM and SP are located near human settlement, while AP and KR are nearly 300 m away from habitation. Agricultural fi

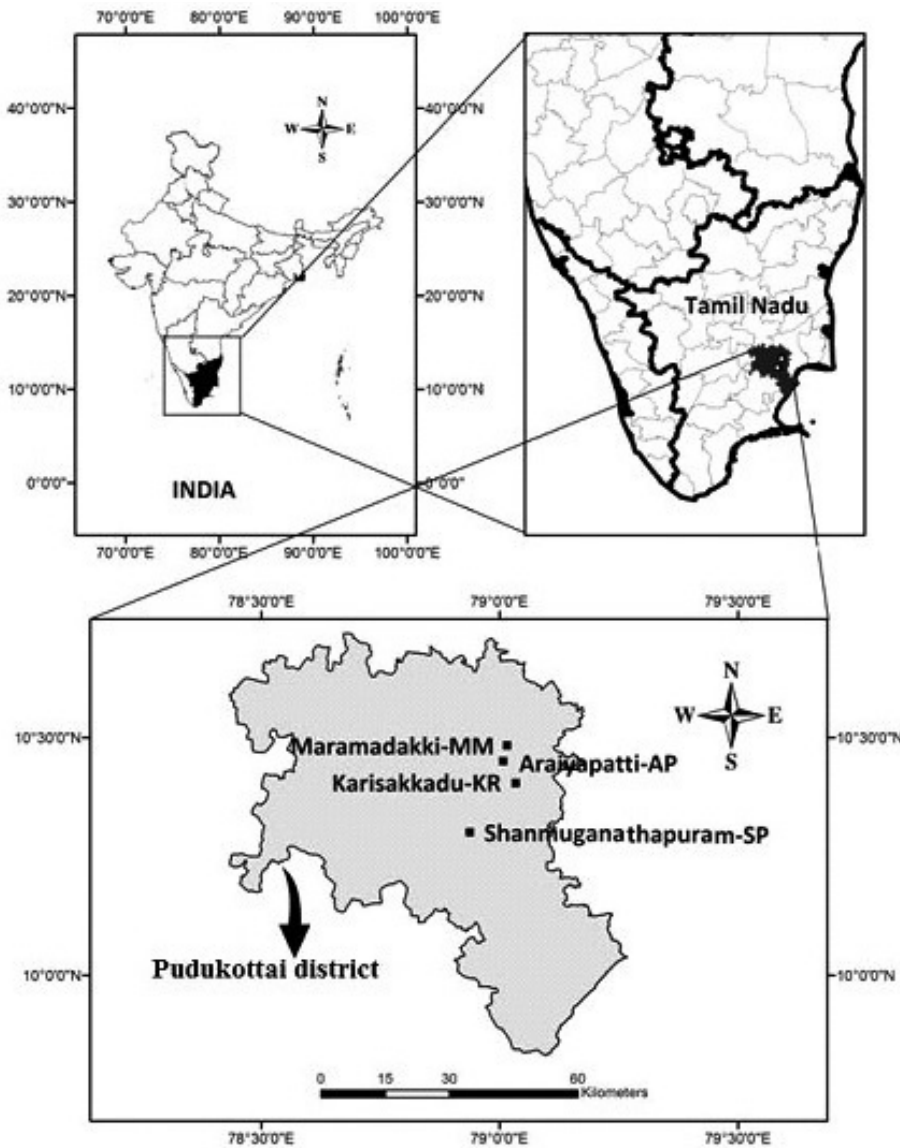


Fig. 1. Map showing the location of the four tropical dry evergreen forest sites on the Coromandel Coast of India Maramadakki (MM), Karisakkadu (KR), Araiypatti (AP) and Shanmuganathapuram (SP).

elds of paddy, maize, millet, *Casuarina*, sugarcane and vegetable plantations surround all the study sites. Species such as *Memecylon umbellatum* and *Glycosmis mauritiana* are often cut for fuel wood and other purposes. The inter-distance between the four study sites ranges from 4 to 25 km. The four study sites experience various levels of anthropogenic disturbance. Sites AP and SP are much disturbed due to heavy anthropogenic activities. The disturbed sites are selectively logged (tree felling for poles, timber and fuel wood extraction) by local people, whereas site MM is least disturbed and KR site is moderately disturbed. The climate data available for 1992–2012 from the nearest station (located 15 km away from our study sites) reveal mean annual rainfall of 1033 mm per year received largely during the northeast monsoon (October–December) with a mean annual temperature of 29.5°C. The mean annual maximum and minimum temperature ranges from 33.4°C to 25.4°C for the same period (Anbarashan and Parthasarathy 2012). The vegetation of all our study sites is described as tropical dry evergreen forests (Type 7/CI of Champion and Seth

(1968); Mani and Parthasarathy (2005). Soils are basically lateritic and sandy to sandy loam in texture in all the sites.

Anthropogenic disturbance was estimated in the four study sites – MM, KR, AP and SP and the disturbance levels were qualitatively classified (as in Veblen et al. 1992 with modifications; Table 1) and were compared with that of 2003 scores (Mani and Parthasarathy 2005). High scores denote high level of site disturbance and low ranks indicate low level. The sum of all scores provides an overall state of anthropogenic disturbance. Sites AP and SP are more highly disturbed than those of MM and KR sites. Over a decade, human disturbance increased by three fold in site AP, one and a half fold in site SP, but there were only moderate changes in sites MM and KR.

Data collection

During the initial inventory of 2003, four 1 ha permanent plots (100 m × 100 m) were established in four tropical dry evergreen forest sites - AP, MM, KR and SP. All trees ≥ 10 cm girth at breast height (gbh) were measured at 1.3

Table 1. Past and present site disturbance score status (1- ranked as rare: 2- occasional: 3- frequent) of four inland tropical dry evergreen forest sites, Araiypatti (AP), Karisakkadu (KR), Maramadakki (MM) and Shanmuganathapuram (SP) in peninsular India

Attributes	AP		KR		MM		SP	
	2003	2013	2003	2013	2003	2013	2003	2013
1 Site encroachment (land use within the forest)								
i. temple construction	1	2	2	3	3	3	2	2
2 Bridle path use	1	3	2	2	3	3	3	3
3 Temple visitors impact: area used for								
i. vehicle parking (area occupied)	1	2	3	3	2	2	3	3
ii. cooking	1	2	3	3	3	2	2	3
iii. festive occasion use	1	3	2	2	3	3	2	3
4 Grazing (cattle/goat)	2	3	2	2	3	3	2	2
5 Culture attachment of local people	1	2	2	2	3	3	2	3
6 Resource removal								
i. firewood	1	1	1	1	2	1	1	2
ii. timber	1	2	1	1	1	1	1	2
iii. other: medicinal plants, edible fruits and soil	1	2	1	1	2	2	2	2
7 Nearest habitation & people's forest dependence	1	2	2	2	3	3	3	3
8 Approach road to temple (width)	1	2	2	2	3	3	3	3
9 Fodder (native forest species) collection	0	3	0	2	0	2	0	2
10 Habitat fragmentation	0	2	0	1	0	1	0	2
11 Solid waste dumping (plastic, polythene, glass, rexin)	0	2	0	1	0	1	0	2
12 Ground clearing for temple related work	0	1	0	1	0	1	0	1
Total score	13	34	23	29	31	33	26	38

m from the ground level (Mani and Parthasarathy 2005). In the present census of 2013, the 1 ha plot at each site was subdivided into one-hundred 10 m×10 m quadrats. For multi-stemmed trees, bole girths were measured separately. The census (2013) results were compared with initial inventories (2003) to analyse the changes in tree density over a decade (2003-2013) at species and stand levels and for aboveground biomass - ABG (Chave et al. 2005). For species diversity and evenness, the Shannon (H') Simpson (λ) and Fisher's (α) diversity indices were calculated (Magurran 2004). The differences in diameter distribution of trees between the two inventories (2003-2013) were tested using Kolmogorov-Smirnov two sample test (Zarr 2006), and we used paired t-tests to check for the significant differences in tree variables across four forest sites using SPSS software. Above ground biomass was calculated using regression equation given by Chave et al. (2005): $AGB_{est} = \rho \times \exp(-1.499 + 2.148 \ln(D) + 0.207 (\ln(D))^2 - 0.0281 (\ln(D))^3)$ where, ρ is wood specific gravity and D is diameter at breast height.

Results

Decadal change in tree diversity

The re-inventory of trees in four tropical dry evergreen forest sites (Maramadakki, Karisakkadu, Araiypatti and Shanmuganathapuram) on the Coromandal Coast of peninsular India showed a significant change in species richness (*df*=99, *p*<0.05; except in SP), density (*df*=99, *p*<0.05) and basal area (*df*=99, *p*<0.05; except in KR) in the ten-year interval (2003-2013). The initial inventories in 2003 recorded 57 species, whereas in 2013, a total of 56 species were recorded with a loss of three species and recruitment of two new species. Species richness in 2013 ranged from a low of 27 species in site MM to a high of 37 species in site AP, through intermediate values of 30 and 29 species in sites KR and SP respectively, (Table 2). In the ten-year interval the tree species richness reduced by 3% (one species) in site MM whereas in site SP species richness increased by 10% (four species), by 5% (two species) in AP and 3% (one species) in site KR. There was an addition of three and two genera in sites SP and AP respectively; one genus disappeared from site MM, but in site KR there was no change. In AP, KR and MM one family

Table 2. Decadal changes (2003-2013) in tree diversity in four tropical dry evergreen forest sites - Araiypatti (AP), Karisakkadu (KR), Maramadakki (MM) and Shanmuganathapuram (SP) on the Coromandel Coast of peninsular India

Variable	AP		KR		MM		SP		Net change
	2003	2013	2003	2013	2003	2013	2003	2013	
Species richness	35	37	30	31	28	27	26	29	3
Number of genera	30	32	27	27	25	24	24	27	3
Number of families	20	21	17	17	18	16	18	19	1
Shannon	2.44	2.561	2.24	2.527	2.01	2.115	1.29	1.549	0.259
Simpson	0.139	0.12	0.212	0.132	0.239	0.197	0.498	0.395	-0.103
Fisher's a	7.198	8.315	6.659	6.705	5.789	5.487	4.375	5.376	1.01
Stand density (stem ha ⁻¹)	807	705	596	678	724	750	1663	1182	-481
Stand basal area (m ² ha ⁻¹)	19.1	19.43	21.6	20.26	15.5	18.63	22.1	20.38	-1.72
Aboveground biomass (Mg ha ⁻¹)	424.74	460.25	373.33	352.26	305.43	405.53	349.86	333.33	-16.33
			Net change	Net change	Net change	Net change	Net change	Net change	Net change
			2	1	1	0	0	0	0
			0.121	0.287	0.287	0.105	0.105	0.105	0.105
			-0.019	-0.08	-0.08	-0.042	-0.042	-0.042	-0.042
			1.117	0.046	0.046	-0.302	-0.302	-0.302	-0.302
			-102	82	82	26	26	26	26
			0.42	-1.34	-1.34	3.13	3.13	3.13	3.13
			35.51	-21.07	-21.07	100.04	100.04	100.04	100.04

was lost, whereas in SP family loss was none. The Shannon index increased in all the study sites: from 2.44 to 2.56 in site AP, 2.24 to 2.52 in site KR, 2.01 to 2.11 in site MM and 1.29 to 1.54 in site SP; while the Simpson index decreased in all the study sites (Table 2). The stand density decreased in the ten-year interval by 480 and 102 stems in SP and AP respectively, but increased moderately by 82 stems in KR and 26 stems in MM (Table 2). The tree basal area per hectare declined in site KR from 21.6 m² to 20.26 m² and in site SP from 22.1 m² to 20.38 m² but increased

from 19.1 m² to 19.43 m² in AP and from 15.5 m² to 18.63 m² in MM (Table 2).

Decadal changes in species richness and density by tree girth class

The changes in species richness and density of trees across girth classes in all the study sites varied considerably (Fig. 2). In MM, tree species richness declined in the lower girth class (10-30 cm), whereas in other three sites species richness increased in lower girth class. In the 61-90 cm girth class site KR lost one species and AP lost two species, but in other two sites it remained unchanged. While in the middle girth class (121-150 cm), the species richness showed a considerable variation in all the four sites. In the highest girth class (>210), there was no change in species richness in all the sites except in KR. The maximum stem density loss was in the lower girth class (10-30 cm) in SP (435 stems), whereas in KR tree density increased by 136 stems. In MM the stem density increased in all the girth classes, while in the highest girth class (>210 cm) the stem density increased in MM and KR, decreased in AP, but remained unchanged in SP (Table 2). Kolmogorov-Smirnov two sample test (K-S) did not show a significant difference across the studied sites AP, KR, MM and SP ($d=0.0623$, $p>0.05$; $d=0.157$, $p>0.05$; $d=0.0128$, $p>0.05$; $d=0.0206$, $p>0.05$ respectively).

Change in size-class distribution of dominant tree species over a decade

Over a decade the stem size-class distribution of dominant tree species changed considerably in all the study sites. Tree density decreased in lower girth class (10-30 cm) for species such as *Memecylon umbellatum*, *Pterospermum canescens*, *Albizia amara*, *Lepisanthes tetraphylla*, *Drypetes sepiaria*, *Strychnos nux-vomica* and *Pongamia pinnata*. Whereas the density of *Euphorbia antiquorum* and *Canthium dicoccum* marginally increased in the lower girth class. Notably, for *Glycosmis mauritiana* the stem density doubled in the lower girth class. Tree density of 30-60 cm girth class increased for *Memecylon umbellatum*, *Strychnos nux-vomica*, *Canthium dicoccum* and *Lepisanthes tetraphylla*, while for *Pterospermum canescens* it remained unchanged over a decade. Over all, the tree density decreased with increasing girth class for all the species (Fig. 3).

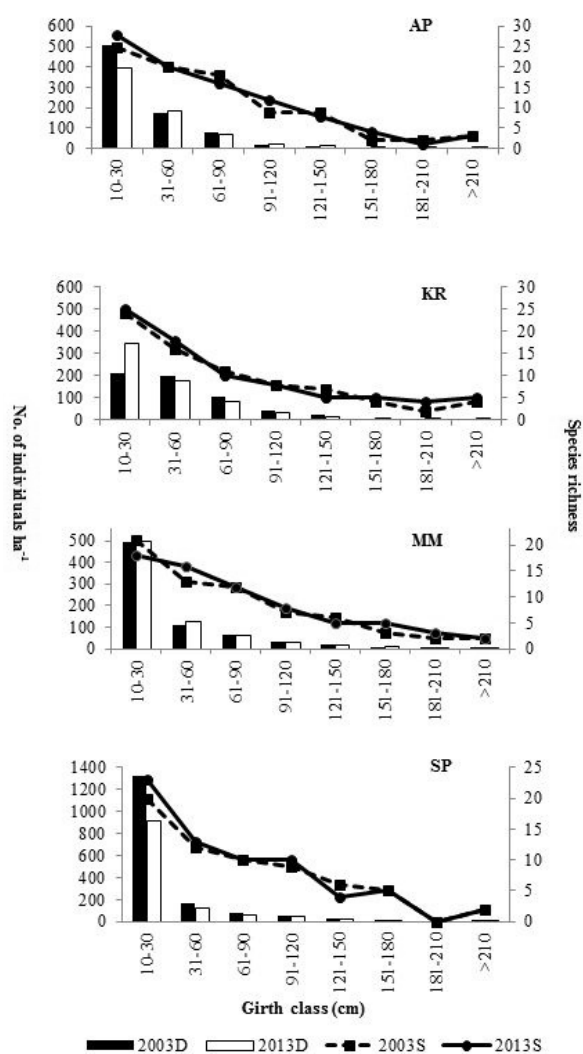


Fig. 2. Decadal changes (2003-2013) in species richness (SR) and tree density (D) by tree girth class in Araiypatti (AP), Karisakkadu (KR), Maramadakki (MM) and Shanmuganathapuram (SP) tropical dry evergreen forest sites.

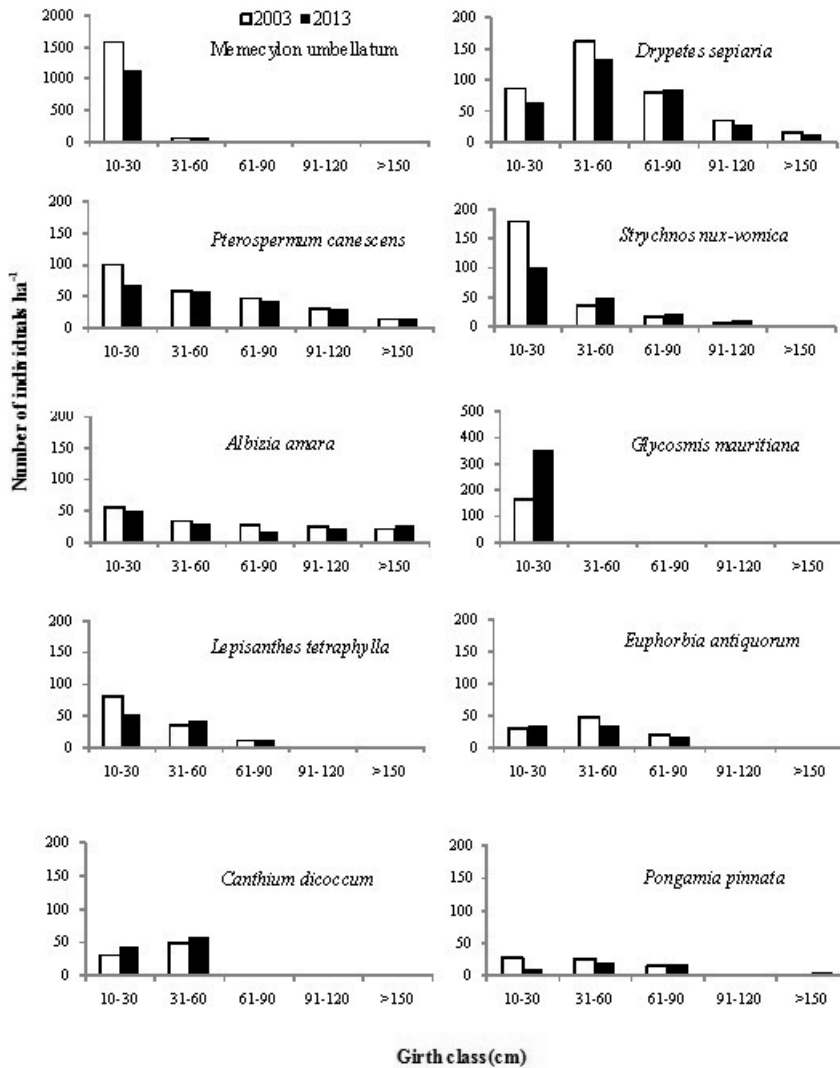


Fig. 3. Decadal changes (2003-2013) in tree density of dominant species at Araiypatti (AP), Karisakadu (KR), Maramadakki (MM) and Shanmuganathapuram (SP). Data were pooled for all four sites.

Species composition changes over a decade

The species density of many trees has changed in all the study sites (AP, KR, MM and SP) over the decade (Table 3). In AP, one tree species (*Ehretia pubescens*) was lost, three species (*Benkara malabarica*, *Lannea coromandelica* and *Streblus asper*) were newly added and 35 species survived. In KR, there was an addition of one species (*Diospyros montana*) without any loss and 31 species survived; in MM, two species (*Jatropha gossypifolia* and *Clausena dentata*) were new additions, three species (*Flacourtia indica*, *Allophylus serratus* and *Maytenus emarginata*) were lost and 27 species survived; while in SP, two species (*Manilkara hexandra*,

Albizia lebbek) were lost and five species (*Canthium dicoccum*, *Cadaba trifoliata*, *Gmelina asiatica*, *Dalbergia paniculata* and *Cassia siamea*) were newly added and 29 species survived. Tree density of two species (*Strychnos nux-vomica*-62 stems and *Lepisanthes tetraphylla*-22 stems) decreased in AP, and those of *Drypetes septaria* (51 stems in site KR) and *Memecylon umbellatum* (41 stems) decreased in site MM. The dominant tree *Memecylon umbellatum* suffered a great stem loss (435 stems), followed by *Pterospermum canescens* (40 stems) in SP; while a moderate gain of 38, 56, 48 and 45 stems of *Glycosmis mauritiana* was recorded in sites AP, KR, MM and SP respectively. In the 2013 re-inventory there was an addition of just one individuals for a

Table 3. Comparison of species density (no. of stems ha⁻¹) and its net change (NC) by sites and total net change (TNC) for all trees ≥ 10 cm gbh in four tropical dry evergreen forest sites-Arayapatti (AP), Karisakkadu (KK), Maramadakki (MM) and Shanmuganathapuram (SP) on the Coromandel Coast of India, along with details of family, arranged in decreasing order of net tree density change between 2003-2013

Tree species (and family)	AP		NC		KK		NC		MM		SP		TNC	
	2003		2013		2003		2013		2003		2013		NC	
	2003	2013	2003	2013	2003	2013	2003	2013	2003	2013	2003	2013	2003	2013
<i>Glycosmis mauritiana</i> (Lam.) Yuich. Tanaka (Rutaceae)	48	86	38	11	67	56	80	128	48	26	71	45	187	
<i>Canthium dicoccum</i> (Gaertn.) Teijsm. & Binn. (Rubiaceae)	0	0	0	11	27	16	69	69	0	0	4	4	20	
<i>Tarema asiatica</i> (L.) Kuntze ex Schumann (Rubiaceae)	7	11	4	4	12	8	3	8	5	22	23	1	18	
<i>Cadaba trifoliata</i> (Roxb.) Wight & Arn. (Capparaceae)	0	0	0	12	28	16	10	8	-2	0	2	2	16	
<i>Gardenia resinifera</i> Roth (Rubiaceae)	0	0	0	19	26	7	0	0	0	0	0	0	7	
<i>Benkara malabarica</i> (Lam.) Tirven. (Rubiaceae)	0	1	1	1	6	5	2	3	1	0	0	0	7	
<i>Clausena dentata</i> (Willd.) Roemer (Rutaceae)	3	6	3	0	0	0	0	1	1	6	7	1	5	
<i>Dalbergia paniculata</i> Roxb. (Papilionaceae)	1	5	4	0	0	0	0	0	0	0	1	1	5	
<i>Flacourtia indica</i> (Burm.f.) Merr. (Salicaceae)	1	6	5	0	0	0	1	0	-1	0	0	0	4	
<i>Lamnea coromandelica</i> (Houtt.) Merr.(Anacardiaceae)	0	1	1	21	23	2	1	1	0	1	1	0	3	
<i>Diospyros ebenum</i> Koen.(Ebenaceae)	11	13	2	3	4	1	0	0	0	0	0	0	3	
<i>Morinda pubescens</i> J.E. Smith (Rubiaceae)	4	4	0	5	4	-1	0	0	0	5	8	3	2	
<i>Diospyros ferrea</i> (Willd.) Bakh. (Ebenaceae)	0	0	0	2	4	2	0	0	0	0	0	0	2	
<i>Chloroxylon swietenia</i> DC. (Flindersiaceae)	40	30	-10	16	27	11	10	11	1	26	25	-1	1	
<i>Gmelina asiatica</i> L. (Verbenaceae)	5	4	-1	0	0	0	1	1	0	0	2	2	1	
<i>Ficus benghalensis</i> L. (Moraceae)	0	0	0	2	3	1	2	2	0	1	1	0	1	
<i>Acacia leucophloea</i> (Roxb.) Willd. (Mimosaceae)	1	2	1	0	0	0	0	0	0	2	2	0	1	
<i>Ficus amplissima</i> J.E.Smith (Moraceae)	0	0	0	1	1	0	2	3	1	0	0	0	1	
<i>Jatropha gossypifolia</i> L. (Euphorbiaceae)	0	0	0	0	0	0	0	1	1	0	0	0	1	
<i>Strobilus asper</i> Lour. (Moraceae)	0	1	1	0	0	0	0	0	0	0	0	0	1	
<i>Cassia fistula</i> L. (Caesalpinaceae)	6	6	0	3	4	1	6	6	0	5	4	-1	0	
<i>Acalantia monophylla</i> (L.) Correa (Rutaceae)	4	5	1	0	0	0	0	0	0	6	5	-1	0	
<i>Sapium insigne</i> (Royle) Trimen (Euphorbiaceae)	0	0	0	9	9	0	0	0	0	0	0	0	0	
<i>Pleiospermium alatum</i> (Wall. ex Wight. & Arn.) Swingle (Rutaceae)	2	4	2	0	0	0	0	0	0	3	1	-2	0	
<i>Premna serratifolia</i> L. (Verbenaceae)	0	0	0	0	0	0	4	4	0	0	0	0	0	
<i>Wrightia tinctoria</i> (Roxb.) R.Br. (Apocyanaceae)	0	0	0	0	0	0	1	3	2	3	1	-2	0	
<i>Ficus microcarpa</i> L.f. (Moraceae)	0	0	0	0	0	0	3	3	0	0	0	0	0	
<i>Commiphora candate</i> (Wight & Arn.) Engle. (Bursaceae)	0	0	0	2	2	0	0	0	0	0	0	0	0	

Table 3. Continued

Tree species (and family)	AP		KK		MIM		SP		TNC	
	NC		NC		NC		NC			
	2003	2013	2003	2013	2003	2013	2003	2013		
<i>Ficus religiosa</i> L. (Moraceae)	1	1	0	0	0	0	1	1	0	0
<i>Phyllanthus polyphtyllus</i> Willd. (Euphorbiaceae)	0	0	0	2	0	0	0	0	0	0
<i>Cantium coromandelicum</i> (Burm.f.) Alston (Rubiaceae)	1	1	0	0	0	0	0	0	0	0
<i>Walsura trifolia</i> (A. Juss.) Harms (Meliaceae)	0	0	0	1	0	0	0	0	0	0
<i>Prosopis juliflora</i> (Sw.) DC. (Mimosaceae)	11	12	1	0	0	6	0	0	0	-1
<i>Diospyros montana</i> Roxb. (Ebenaceae)	9	7	-2	0	1	0	0	0	0	-1
<i>Cassia roxburghii</i> DC. (Caesalpinaceae)	3	2	-1	0	0	1	0	0	0	-1
<i>Cassia siamea</i> Lam. (Caesalpinaceae)	4	2	-2	0	0	0	0	1	1	-1
<i>Craeva magna</i> (Lour.) DC. (Capparaceae)	3	2	-1	1	0	0	0	0	0	-1
<i>Albizia odoratissima</i> (L.f.) Benth. (Mimosaceae)	0	0	0	3	2	0	0	0	0	-1
<i>Sapindus emarginatus</i> Vahl (Sapindaceae)	0	0	0	2	1	0	0	0	0	-1
<i>Terminalia bellirica</i> (Gaertner) Kurz (Combretaceae)	2	1	-1	0	0	0	0	0	0	-1
<i>Allophylus serrates</i> (Roxb.) Kurz (Sapindaceae)	0	0	0	0	0	0	0	0	0	-1
<i>Maytenus emarginata</i> (Willd.) Ding Hou (Celastraceae)	0	0	0	0	0	0	0	0	0	-1
<i>Ehretia pubescens</i> Benth. (Cordiaceae)	2	0	-2	0	0	0	0	0	0	-2
<i>Syzygium cumini</i> (L.) Skeels (Myrtaceae)	9	9	0	0	0	0	0	21	18	-3
<i>Borassus flabellifer</i> L. (Arecaceae)	0	0	0	0	0	0	0	11	8	-3
<i>Strychnos potatorum</i> L. f. (Loganiaceae)	9	6	-3	0	0	0	0	0	0	-3
<i>Albizia lebbek</i> (L.) Benth. (Mimosaceae)	5	2	-3	0	0	0	0	0	0	-4
<i>Vitex altissima</i> L. f. (Verbenaceae)	0	0	0	10	11	1	17	23	16	-5
<i>Azadirachta indica</i> A. Juss. (Meliaceae)	4	1	-3	6	6	0	1	6	3	-6
<i>Cordia oblique</i> Willd. (Cordiaceae)	18	11	-7	0	0	0	0	0	0	-7
<i>Mamlikara hexandra</i> (Roxb.) Dubard (Sapotaceae)	7	7	0	26	22	-4	37	28	1	-14
<i>Euphorbia antiquorum</i> L. (Euphorbiaceae)	5	3	-2	56	48	-8	11	22	25	-15
<i>Albizia amara</i> (Roxb.) Boivin (Mimosaceae)	61	44	-17	30	30	0	25	38	48	-20
<i>Pongamia pinnata</i> (L.) Pierre (Papilionaceae)	72	51	-21	0	0	0	0	0	0	-21
<i>Lepisanthes tetraphylla</i> (Vahl.) Radlk. (Sapindaceae)	121	99	-22	3	3	0	4	3	1	-23
<i>Persea pernum canescens</i> Roxb. (Sterculiaceae)	0	0	0	8	8	0	14	12	228	-42
<i>Drypete sepiaria</i> (Wight & Arn.) Pax & Hof. (Euphorbiaceae)	3	3	0	253	202	-51	86	87	30	-58
<i>Strychnos nux-vomica</i> L. (Loganiaceae)	235	173	-62	0	0	0	0	0	5	-62
<i>Memecylon umbellatum</i> Burm.f. (Melastomataceae)	89	83	-6	73	93	20	323	282	1148	-462
Total	807	705	-102	596	678	82	724	750	1663	-480

total of 7, 5, 7 and 8 species respectively, in sites AP, KR, MM and (Table 3).

Family composition changes over a decade

During the re-inventory, 26 families were enumerated. Tree diversity changes at the family-level ranged from a stem loss of as high as 435 stems of Melastomataceae to a gain of 43 stems of Rutaceae in SP, to as low as a stem decline of 41 trees of Melastomataceae and an addition of 49 stems of Rutaceae in site MM (Table 4). Whereas in AP and KR a moderate decrease of 65 stems of Loganiaceae, 59 stems of Euphorbiaceae were recorded, and the stem density gain was 44 and 56 stems of Rutaceae respectively, for the same sites. Maximum stem loss occurred in Melastomataceae (435 stems), Sterculiaceae (40 stems) and Euphorbiaceae (25 stems) in site SP. Two families (Celastraceae and Salicaceae) were lost from the site MM and no family was added, whereas two families (Capparaceae and Papilionaceae) were added and one family (Sapotaceae) was lost in site SP, one family (Anacardiaceae) was added in site AP and no change occurred in KR. During the initial census, 26 families were enumerated in all sites (AP, KR, MM and SP), of which 12 families were common to all the sites. Rubiaceae was the most speciose family in all the study sites, but showed maximum changes in tree density in sites AP and SP, with one species each newly added (*Benkara malabarica* and *Canthium dicoccum* respectively in sites AP and SP), Whereas in KR and MM there was no change in species richness of Rubiaceae. The tree density in Rubiaceae increased by 5, 35, 6 and 8 stems in AP, KR, MM and SP respectively (Table 4).

Decadal changes in aboveground biomass (AGB)

During the decade of interval (2003-2013) the AGB increased by 100.09 Mg ha⁻¹ and 35.51 Mg ha⁻¹ in two sites (MM and AP) but in the other two sites (KR and SP) it has decreased by 21.07 and 16.53 Mg ha⁻¹ respectively. A Kolmogorov-Smirnov two sample test (K-S) did not show a significant difference in AGB across the study sites AP, KR, MM and SP ($d=0.0511$, $p>0.05$; $d=0.0435$, $p>0.05$; $d=0.0698$, $p>0.05$; $d=0.04$, $p>0.05$ respectively). The total AGB changed considerably in small and middle girth classes. The AGB for lower girth class (10-30 cm) decreased in SP (by 2.13 Mg ha⁻¹), AP (4.77 Mg ha⁻¹) and

KR (0.46 Mg ha⁻¹), but increased in MM (by 2.03 Mg ha⁻¹; Fig. 4). In middle girth class, AGB declined by 5.28 Mg ha⁻¹ and 9.78 in KR and SP, whereas in other two sites it increased by 9.56 Mg ha⁻¹ and 21.94 at AP and MM respectively. In higher girth class (151-180 and >210 cm) the AGB increased in all the sites (Fig. 4).

Discussion

Re-census of tree density in four Indian tropical dry evergreen forest sites revealed a considerable change in species richness, stem density and basal area of trees over a decade (2003-2013). During this decadal interval, total tree species richness increased by 10% in site SP, 5% in AP and 1% in KR, whereas in site MM tree species richness declined by 3%. In the present study, loss of three tree species (*Allophylus serratus*, *Maytenus emarginata* and *Ehretia pubescens*), which were represented by single stem in the decade-back initial inventory, could be attributed either to natural cause of tree death or to human interference, during the ten-year gap. Species of small populations are vulnerable to local extinction as death of few individuals can result in loss of species representation (Primack and Hall 1992). Globally, species loss in tropical forest has been driven by a complex interplay of various forces. These include anthropogenic disturbances like logging and land expansion for agriculture, probably coupled with climate changes (Condit et al. 1996; Kolb and Diekmann 2004; Stork 2010; Appiah 2013). In several forest areas, fragmentation as well as repeated forest destruction have resulted in the elimination of tree species from forests. Lalfakawma et al. (2009) also reported that species richness declined in disturbed semi-evergreen forests of Lunglei district of Mizoram, north-east India, and also tree species richness was reported to decrease with increasing stand disturbance in tropical dry forest of Nicaragua (Sabogal 1992). Fashing et al. (2004) reported that tree species richness declined by 21% over an 18-year interval in Kakamega forest, Kenya. In our sites at a decadal interval two species (*Jatropha gossypifolia* and *Streblus asper*) were newly recorded in sites MM and AP respectively. Sheil et al. (2000) also reported increased species richness over a time period (1944-1992) in Budongo rain forest, Uganda. In the present study the Shannon diversity index increased in all the sites, while the Simpson in-

Table 4. Family diversity and density (stems ha⁻¹) changes in 2003 and 2013, arranged in alphabetic order of tree families in four tropical dry evergreen forest sites - Araiyapatti (AP), Karisakadu (KR), Maramadakki (MM) and Shanmuganathapuram (SP)

Family	Species richness												Tree density														
	AP			KR			MM			SP			AP			KR			MM			SP					
	2003	2013	NC	2003	2013	NC	2003	2013	NC	2003	2013	NC	2003	2013	NC	2003	2013	NC	2003	2013	NC	2003	2013	NC			
Anacardiaceae	0	1	1	1	1	0	1	0	1	0	1	1	0	0	1	1	21	23	2	1	1	0	1	0	1	1	0
Apocynaceae	0	0	0	0	0	0	1	1	0	1	1	1	0	0	0	0	0	0	0	1	3	2	3	1	0	1	-2
Arecaceae	0	0	0	0	0	0	0	0	1	1	1	1	0	0	0	0	0	0	0	0	0	0	0	11	8	-3	-3
Burseraceae	0	0	0	1	1	0	0	0	0	0	0	0	0	0	0	0	2	2	2	0	0	0	0	0	0	0	0
Capparaceae	1	1	0	2	2	0	1	1	0	0	1	1	3	10	7	13	4	-9	10	7	8	1	5	5	0	2	2
Cesalpiniaceae	3	3	0	1	1	0	2	2	0	1	2	1	13	2	-11	3	29	26	7	8	1	0	-1	0	0	0	
Celastraceae	0	0	0	0	0	0	1	0	-1	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	0	
Combretaceae	1	1	0	0	0	0	0	0	0	0	0	2	1	-1	0	0	0	0	0	0	0	0	0	0	0	0	
Cordiaceae	2	1	-1	0	0	0	0	0	0	0	0	20	11	-9	0	0	0	0	0	0	0	0	0	0	0	0	
Ebenaceae	2	2	0	2	3	1	0	0	0	0	0	20	20	0	0	5	9	4	0	0	0	0	0	0	0	0	
Euphorbiaceae	2	2	0	4	4	0	2	3	1	2	2	0	8	6	-2	320	261	-59	97	110	13	63	38	-25	0	0	
Salicaceae	1	1	0	0	0	0	1	0	0	0	0	1	6	5	0	0	0	0	1	0	-1	0	0	0	0	0	
Flindersiaceae	1	1	0	1	1	0	1	1	0	1	1	0	40	30	-10	16	27	11	10	11	1	1	26	25	-1	-1	
Loganiaceae	2	2	0	0	0	0	0	0	0	1	1	0	244	179	-65	0	0	0	0	0	0	0	5	5	0	0	
Melastomateaceae	1	1	0	1	1	0	1	1	0	1	1	0	89	83	-6	73	93	20	323	282	-41	1148	713	-435	0	0	
Meliaceae	1	1	0	2	2	0	1	1	0	1	1	0	4	1	-3	7	7	0	1	1	0	6	3	-3	-3		
Mimosaceae	4	4	0	2	2	0	2	2	0	3	2	-1	78	60	-18	33	32	-1	33	44	11	51	34	-17	-17		
Moraceae	1	2	1	2	2	0	3	3	0	2	2	0	1	2	1	3	4	1	7	8	1	2	2	2	0		
Myrtaceae	1	1	0	0	0	0	0	0	0	1	1	0	9	9	0	0	0	0	0	0	0	0	21	18	-3		
Papilionaceae	2	2	0	0	0	0	0	0	0	0	1	73	56	-17	0	0	0	0	0	0	0	0	0	1	1		
Rubiaceae	3	4	1	5	5	0	3	3	0	2	3	1	12	17	5	40	75	35	74	80	6	27	35	8	8		
Rutaceae	4	4	0	1	1	0	1	2	1	4	4	0	57	101	44	11	67	56	80	129	49	41	84	43	43		
Sapindaceae	1	1	0	2	2	0	2	1	-1	1	1	0	121	99	-22	5	4	-1	5	3	-2	1	1	0	0		
Sapotaceae	1	1	0	1	1	0	1	1	0	1	0	-1	7	7	0	26	22	-4	37	28	-9	1	0	-1	-1		
Sterculiaceae	0	0	0	1	1	0	1	1	0	1	1	0	0	0	0	8	8	0	14	12	-2	228	188	-40	-40		
Verbenaceae	1	1	0	1	1	0	3	3	0	1	2	1	5	4	-1	10	11	1	22	23	1	23	18	-5	-5		
Total	35	37	2	30	31	1	28	27	-1	26	29	3	807	705	-102	596	678	82	724	750	26	1663	1182	-481	-481		

dex decreased in all the four study sites, and this result agrees with that of the recent long-term study conducted in a site at Puthupet, an Indian coastal tropical dry evergreen forest (Baithalu et al. 2013).

The stand density decreased in highly disturbed sites SP (28.9%) and AP (14.5%), but increased moderately in less disturbed sites KR (12.1%) and MM (3.47%). Reduction in stem density can be attributed to removal of stems, browsing and trampling by livestock and human pressure

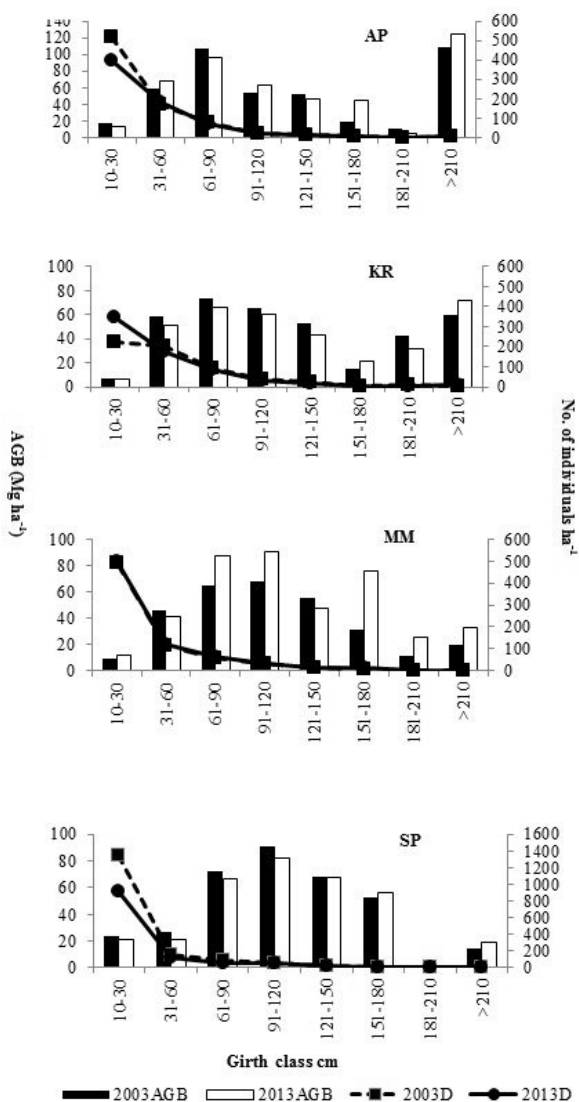


Fig. 4. Decadal changes (2003-2013) in aboveground biomass (AGB) and tree density (SD) by tree girth class in Araiypatti (AP), Karisakkadu (KR), Maramadakki (MM) and Shanmuganathapuram (SP) tropical dry evergreen forest sites.

including land use change such as expanding agriculture, village temple construction, and human settlement close to forests. The coastal tropical dry evergreen sites also showed sharp reductions in stem density over decades (Baithalu et al. 2012, 2013 and Mani and Parthasarathy 2009). Felfili (1995) reported a reduction of 2% in density over a six-year period in the gallery forest of central Brazil. In another Brazilian site Felfili et al. (2000) reported that the stem density decreased by 4.5% over a nine-year period. Decrease in stem density has been attributed to disturbance intensity (Ramirez-Marcial et al. 2001) in Mexico, tree cutting for domestic use in Indonesia (Smiet 1992), and selective and commercial extraction of timber (Elouard et al. 1997; Pomeroy et al. 2003) in Western Ghats. Several studies have reported more intensive exploitation of forests close to human settlements than those located in remote areas (Acharya 1999; Sagar et al. 2003; Ramacharitra 2006). Bhat et al. (2011) also observed stem density decreased by (45.5%) in different study sites over a period 25 years in Uttara Kannada district, Western Ghats, south India. Stem density declined notably between the first (1971) and the last (2012) survey in the tropical rain forests of Australia (Murphy et al. 2013). On the other hand, the stand density increased by 15.1% from 375 trees ha⁻¹ to 411 trees ha⁻¹ over a 4.3 year period in a study conducted by Krisnawati et al. (2011) at Samboja research forest east Kalimantan. Silva et al. (1995) found an addition of 13% in density over 11-years in a logged forest area in Brazil.

In lower girth class (10-30 cm), tree density decreased drastically in the highly disturbed sites AP and SP when compared to other two less disturbed study sites KR and MM. The most dominant understory tree species *Memecylon umbellatum* decreased in density over a decade in the study sites. This can be attributed to combined effect of stem removal as fuel wood by the local people and also possibly due to progression to the next girth classes. Cutting of a greater number of lower-girthed trees indicates that local people frequently need them in bulk quantity. Baithalu et al. (2013) and Mani and Parthasarathy (2009) also reported substantial reduction in the lower girth trees in coastal tropical dry evergreen forest sites and so also many studies have reported a decline in small stems over a time in other tropical forests of the world (Primack and Hall 1992; Newbery et al.1999; Janzen 1970; Clark and Clark 1984; Connell et

al. 1984). Several studies have reported dependency of rural population on forest for meeting their biomass requirement such as fuel wood, fodder, fencing poles and small timber for agricultural implements etc., leading to forest degradation (Singhal et al. 2003; Arjunan et al. 2005; Chandrashekara et al. 2005; Davidar et al. 2007; Bhat and Ravindranath 2011). In the less disturbed sites KR and MM, stem density increased over a decade in lower girth class (10-30 cm gbh) which agrees with the findings of previously studied Puthupet coastal tropical dry evergreen forest site (Parthasarathy and Venkateswaran 2005). The tree density distribution among different girth classes with high density in the lower girth classes and low density as in the highest girth class showed the decreasing tendency of the forest using the site resources, as Hitimana et al. (2004) also found a correlation between tree distribution along diameter classes and site resource utilization efficiency of forest. The basal area of trees declined over a decade in highly disturbed sites AP and SP, indicating high anthropogenic pressure. The frequent tree cutting was evident from the cut sign and the multiple branching of stems at ground level as authenticated by the one and half to three-fold increased disturbance score recorded in sites AP and SP over a decade. Bhat et al. (2000) also found decreased basal area with increasing human disturbance in tropical forests of Western Ghats, India. Baithalu et al. (2013) and Mani and Parthasarathy (2009) also reported that the basal area decreased in other tropical dry evergreen forest sites AK and TM with increasing disturbance, while Hitimana et al. (2004) reported that the basal area in larger diameter classes has become reduced in western Kenya. In contrast, basal area of trees increased due to the less disturbed condition of sites MM and KR. Bhat and Ravindranath (2011) reported basal area increased over 25 years, in tropical rain forest of Uttara Kannada District, Western Ghats, India. Laurance et al. (2009) in central Amazonian forests and Krisnawati et al. (2011) in Samboja research forest, east Kalimantan also reported an increase in basal area over a 4.3-year time period.

During our census interval (2003-2013), the total above-ground biomass increased by 100.09 Mg ha⁻¹ in site MM and 35.51 Mg ha⁻¹ in AP, which may be due to an increase in biomass due to increased tree density and tree diameter. The storage of carbon in a forest is related to multiple rea-

sons including climate, soils structure, nutrient availability and disturbance, and many sites also reported a high correlation of biomass with dbh (Brown et al. 1989; Cairns et al. 2003; Kale et al. 2004; Ketterings et al. 2001; Lodhiyal and Lodhiyal 2003; Saldarriaga et al. 1988; Telenius and Verwijst 1999; Mani and Parthasarathy 2009) and also the present study revealed a high r² value. According to Baker et al. (2004) and Malhi et al. (2004) AGB increased over a decade in Neotropical forest plots and Chave et al. (2008) also found increased ABG in tropical rain forest of eastern South America. The IPCC (2003&2006) reported increased (2 Mg ha⁻¹ yr⁻¹) biomass of primary tropical forests in South America and 1.3 Mg ha⁻¹ yr⁻¹ in Africa (Lewis et al. 2009). According to West et al. (2014) aboveground biomass increased in plots of conventional logging (disturbed) and reduced-impact logging (undisturbed) over the 16-year monitoring period in Paragominas, Pará state, Brazil. In contrast, two of our study sites, KR and SP showed a marginal decrease in aboveground biomass by 21.07 Mg ha⁻¹ and 16.53 Mg ha⁻¹ respectively, which could be due to decreased tree density and anthropogenic disturbance. In many instances, the carbon stocks in forests may change without a change in forest area. Examples include losses of biomass associated with selective wood harvest, forest fragmentation, ground fires, shifting cultivation, browsing, and grazing (e.g., Laurance et al. 1998; Nepstad et al. 1999; Laurance et al. 2000; Barlow et al. 2003 and Houghton 2005), and accumulation of biomass in growing and recovering (or secondary) forests. Over a decade, biomass of trees increased in higher girth class (151-180 and ≥210 cm) in all the study sites, but decreased in lower girth class (10-30 cm) except in SP. This result agrees with that of West et al. (2014) who found that biomass of trees increased in the higher girth classes, but decreased in lower girth classes.

Conclusion

The four permanent research plots of tropical dry evergreen forest on the Coromandel Coast of peninsular India monitored a decade after their establishment revealed that sites located close to human settlement showed reduced tree density, basal area and aboveground biomass owing to resource extraction facilitated by easy access to forests. In par-

ticular, greater number of trees were removed in lower size classes indicating the forest dependence of local people for their livelihood. Whatsoever the causes be the changes in forest composition and structure will have important implications on vital ecosystem services they render (Korner 2004; Bunker et al. 2005). Therefore an integrated plan is required to support recovery of the forest species and ecosystem by making compensatory planting of native plant species with the involvement of the local community while planning for site conservation. Further research like continuous decadal interval monitoring of permanent plots would help understanding forest dynamics and such long-term data will be valuable for sustainable management of forest and their bioresources and for biological conservation of species in such under-studied tropical dry evergreen and similar forests in a long run. The present study revealed a loss of 1475 tree individuals in a decade interval which attests the extent of anthropogenic pressure prevailing in the study sites. Further, this alarms the immediate need for framing action plans and conservation strategies to protect this forest type itself which is fast vanishing in many sites.

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