II. REVIEW OF LITERATURE
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Species diversity

The number of species in a community is not enough to describe diversity. Equally important is the relative abundance of the species. Higher diversity is found where equitability is higher and tropical forests exhibit greater equitability and species richness as compared to temperate forests. Whittaker (1977) and Whitmore (1984) distinguished three types of diversity: within community (alpha) diversity, between community (beta) diversity and landscape or total (gamma) diversity. The alpha diversity in tropical forests is greater than the temperate forests (Black et al., 1950; Cain et al., 1956).

Diversity in tropical forests is not homogeneous; diversity decreases as the elevation increases and maximum diversity is observed at middle elevation (Whittaker, 1969). High tree species diversity and low population density of individuals is a common property in tropical forests (Davis and Richards, 1934; Black et al., 1950). So far many studies have attempted to address,

1. Why are there so many species in the tropics? (MacArthur, 1965)
2. How do these different species manage to co-exist? (Whitmore, 1984)
3. Why components of diversity increase with decreasing latitude? (Murdoch et al., 1972), but a satisfactory answer is not obtained.

Various models have been studied to explain the species richness. A dynamic equilibrium model of species richness (Huston, 1980) and non-equilibrium or community drift hypothesis (Hubbell, 1979; Connell, 1978) concluded that high diversity in the rainforest is a non-equilibrium state. The maximum diversity occurs in a region of intermediate disturbance of frequency and intensity (Tilman, 1988), or
in resource-poor habitats (Hubbell, 1979). It assumes that species are in competitive equipoise and the diversity expresses as the balance between speciation and extinction due to chance fluctuation in number.

These models are unable to explain the exact reason for higher species richness in tropical forest. Real tropical forests are neither equilibrium nor non-equilibrium communities. Species richness is mainly influenced by time scale, environmental stability, habitat favourableness and characteristics of community (Whittaker, 1969).

**Tree diversity in various tropical forests:**

Davis and Richards (1934) described the vegetation of rainforest region in British Guyana, on the north eastern coast of South America. Trees were measured in two plot sizes 61 x 7.6 m and 122 x 7.6 m in the forests of mixed type. All trees ≥ 4.6 m height measured in one large plot. The stand density was 426 ha⁻¹. The trees formed only two distinct strata, an irregular canopy of about 24m average height and undergrowth trees up to about 14 m height. Black *et al.* (1950) made a comparative study of one hectare plot in Teje and Belém in the Brazilian Amazon forest. All tree species with a minimum gbh of 10 cm were enumerated and found that periodically inundated Igapo forest contained fewer species with strong species dominance than the dry never-inundated terra firme forests, where the distribution of individuals among species was more regular and equal. In terra firme a total density of 423 individuals separating 87 species were found, whereas 564 individuals belonging to 60 species were encountered in Igapo forest. Pires *et al.* (1953) enumerated tree diversity in terra firme forest of Para, Brazil. All trees ≥ 10 cm dbh total to 1482 belonging to 179 species on a 3.5 ha scale. Grubb *et al.* (1963) made a comparative
study of forest structure, physiognomy and floristics in montane and lowland forests, Ecuador, South America. All trees ≥ 10 cm dbh were measured in five 930 m² plots. There were 74, 82, 88, 108 and 126 trees and 27, 31, 29, 30 and 42 species respectively in the five plots.

Ashton (1964) made a quantitative phytosociological study in Brunei mixed dipterocarp forest, Andulau F.R., at Kuala Belalong in south east Asia. All trees ≥ 10 cm diameter measured in 100 acres sample totalled 760 species and represented 30,000 individuals. Poore (1968) made a pioneering study in the Jengka forest reserve, Malaysia, considering the woody species ≥ 30 cm gbh, in an area of 24.48 ha. He enumerated 2773 trees belonging to 375 species representing 139 genera and 52 families. The top ten families accounted 1970 individuals and represented 174 species. The total basal area was 558.16 m².

Paijmans (1970) studied tree species (≥ 30 cm gbh) diversity in the rainforests of Papua New Guinea by belt transect (40 x 200m or 80 x 100m) method. Tree density of four sites was 264, 280, 213 and 174 respectively. Species richness for the same area was 122, 147, 145 and 116 respectively. Among the four plots he found 392 species, of which 146 species had single representatives. The basal area of the four sites was 29.2, 56.7, 31.2 and 30.8 m² respectively. Wong and Whitmore (1970) investigated the tree flora in ten 0.4 ha plots within a 500 ha block across Pasoh Forest Reserve, Malaysia. In the 4 ha sample, 2280 trees of ≥ 30 cm gbh were encountered, with 1820 trees accounting 328 species.

Knight (1975) investigated tree species composition and population structure in the Barro Colorado Island, Panama. All woody species ≥ 2.5 cm dbh were sampled in 13 stands, each measuring 10 x 20 m area. There were 300 species, with stand
average density of 1189 stems. The Shannon index with a mean for all stands was about 4.8. Species diversity increased most rapidly during the first 15 years of succession, but continued to increase slowly after 65 years. The species-area curve showed a slight plateauing at the largest sample size (1.5 ha). Forman and Hahn (1980) studied species diversity and spatial patterns of trees ≥ 10cm dbh in a 4 ha plot of semi-evergreen forest in the Caribbean island, United States. They enumerated 132 individuals belonging to 28 species, with a higher Shannon index (H' = 4.37).

Uhl and Murphy (1981) investigated the composition, structure and regeneration in terra firme rainforest of the San Carlos region, southern Venezuela. All woody species ≥ 10 cm dbh were measured and identified in 2.5 ha area. A 1-ha plot and three 0.5 ha plots accounted for tree densities of 744, 387, 335 and 259 respectively. The species richness of the four sites was 83, 79, 63 and 68 respectively. Shannon diversity values ranged from 4.8 to 5.4. The mean basal area of the three sites was 27.8 m²ha⁻¹.

Thorington et al. (1982) studied the distribution patterns of trees and its importance in Barro Colorado Island, Panama. All trees ≥ 60 cm gbh at 1.5 m above the ground were measured, which totalled to 856 trees representing 112 species in their five 1-ha plots. Prance and Schaller (1982) studied the various types of vegetation in Pantanal region, Brazil-Bolivia. In the five 1-ha plots of cerrado vegetation and semi-deciduous forests, all woody flora ≥ 15cm dbh was enumerated. The cerrado plot contained 189 trees representing 22 species. The 129 species of the 189 trees belonged to only six species. Semi-deciduous forest contained 224 trees of 35 species.
Proctor et al. (1983) enumerated species diversity in four regions of lowland rainforest in Gunung Mulu National Park, Sarawak. All four sites were marked out and sub-divided into twenty-one (20 x 20 m) sites. Tree density (≥ 10 cm dbh) ranged from 615 to 778 ha⁻¹. Species richness ranged from 73 - 223 ha⁻¹ and the basal area from 37 to 57 m²ha⁻¹. Hubbell and Foster (1983) made a detailed study on a 50 ha plot of semi-evergreen forest in Barro Colorado Island, Panama. They found 2,38,000 trees and estimated 28,500 lianas and epiphytes belonging to 300 species with a minimum dbh of 1 cm. Of these, trees ≥ 20cm dbh which were considered as large trees had 7614 representatives and belonged to 186 species. These large trees exhibited dominance with 33 species having more than 50 individuals.

Singh et al. (1984a) made a comparative study of riparian, mesic upland and less mesic upland rainforests in Silent Valley, Kerala considering all woody plants in a 0.1, 0.8 and 0.1 ha respectively. Species richness was 36, 34 and 37 respectively. Shannon diversity index for the three plots was 4.15, 4.08 and 3.52. The total basal area of trees for the three plots was 47.7, 102.7 and 29.7 m²ha⁻¹.

Brown et al. (1985) studied the floristics and structure of a semi-evergreen forest in northwestern Argentina. All woody species ≥ 10 cm dbh were measured in a 0.5 ha area. The density ranged from 264 - 337 trees ha⁻¹, representing 40 tree species. The basal area was 21 to 31 m²ha⁻¹. Whitmore et al. (1986) made total species count in lowland tropical rainforest of Costa Rica. In a 100m² area, they recorded 233 vascular plant species and 32 bryophytes. Murphy and Lugo (1986) made a species diversity study in a sub-tropical dry forest of Puerto Rico, USA. All woody species ≥ 2.5 cm dbh were measured in an area of 1.4 ha (sub-gridded into 144 (10 x 10 m
The mean density was 12173.3 trees ha$^{-1}$ totalling 33 species and 21 families. The basal area was 17.8 m$^2$ha$^{-1}$.

Rai and Proctor (1986) made a floristic study in four evergreen rainforest sites in Karnataka, Western Ghats. The tree density ($\geq 10$ cm dbh) was 795, 333, 515 and 462 ha$^{-1}$ recorded in 0.44 ha (220 x 20m), 1-ha (500 x 20m), 1-ha (500 x 20m) and 0.5 ha (250 x 20m) respectively. The basal area of the trees $\geq 5$ cm dbh was 33.7, 33.5, 35.6 and 48.6 m$^2$ha$^{-1}$.

Itow (1986) studied species diversity of equatorial insular forest on Ponape and Kosrae, Micronesia, using the minimum programme method, (sampling of 100 or more trees with dbh $\geq 10$ cm). Diversity indices such as Shannon ($H'$), the complement of Simpson's index ($d'$), Fischer's alpha $S_{50}$, $S_{100}$ and b were calculated.

There was paucity of species with index value of $H' = 0.95 - 3.30$, $d' = 0.29 - 0.87$, alpha = 2 - 5, $S_{100} = 8 - 15$ and b = 3 - 11. Species-area relationship showed that the curves are gentle in the insular communities and steep in the continental forests.

Campbell et al. (1986) conducted a quantitative ecological inventory in a lowland Amazonian tropical moist forest, selecting three 1-ha primary terra firme forest and 0.5 ha of an adjacent Varzea forest near O'Deset on the eastern bank of Rio Xingu, Para, Brazil. Tree density was 1420 individuals for trees $\geq 10$ cm dbh representing 127 genera and 265 species in the primary terra firme forest. In Varzea (0.5 ha) forest, there were 220 trees representing 29 genera and 40 species. The total basal area varied from 28.87 to 32.14 m$^2$ha$^{-1}$ in terra firme forest and in Varzea forest it was 15.69 m$^2$ha$^{-1}$. Among the 1-ha plots of the total terra firme sample, the number of trees ranged from 393 - 460. The number of families was constant $i.e.$ 33 whereas the number of species ranged from 118 to 162.
Boom (1986) studied tree diversity, density and dominance by transect sample method (10 x 100 m) in terra firme Amazonian rainforest, north eastern Bolivia. The density of trees \( \geq 10 \) cm dbh was 649, accounting 94 species and 30 families. The total basal area was 21.48 m\(^2\)ha\(^{-1}\) and the average basal area per tree was around 0.0331 m\(^2\).

Manokaran and Kochummen (1987) determined recruitment, growth and mortality of tree species \( \geq 10.1 \) cm dbh on a 2.02 ha plot, in primary lowland tropical rainforest, Peninsular Malaysia. Over a 34- year period from 1947 - 1981, species richness was identical, in 1947 and 1981 (243 and 244 species respectively in 45 families; but showed a decline to 229 species between 1953 to 1971. Tree density averaged 492 trees ha\(^{-1}\) between 1947 - 1981 but showed a decline of about 10 \%. The basal area changed little, averaging 32.4 m\(^2\)ha\(^{-1}\).

Lott et al. (1987) made a comparative study of diversity and size class distribution of stems and their relationships in two upland and Arroyo neotropical forests of coastal Jalisco. The Arroyo forest was not only species-rich (105 spp. of woody plants of \( \geq 2.5 \) cm dbh) but also had a higher basal area (0.524m\(^2\)) when compared to the upland forests. The species richness and basal area of the 1000 m\(^2\) area was 92 species and 0.261 m\(^2\); and 83 species and 0.218 m\(^2\) respectively. Their 0.3 ha area, 30 transects (20 x 50m each) contained 1352 individuals representing 176 species.

Swaine et al. (1987) in 18 different sites of 0.1 to 97 ha area-studied dynamics of tree population in tropical forest of Kade, Ghana for 5- to 36-year period. Species richness ranged from 52 to 141 ha\(^{-1}\) (of \( \geq 10 \) cm dbh), basal area from 28 to 52 m\(^2\)ha\(^{-1}\) and tree density from 395 to 734 ha\(^{-1}\). Ho et al. (1987) studied forest composition and
dynamics in Segamet soil series, Jengka forest reserve, Malaysia. All trees, with a minimum of 0.91 cm dbh in non-contiguous plots totalling 11.7 ha were enumerated. There were 1211 woody stems belonging to 261 species, 122 genera and 42 identified families. The mean basal area of the twenty most common species was 23.3 m²ha⁻¹.

Balslev et al. (1987) made plant biodiversity inventories in moist tropical unflooded and flood plain forests in Yasuni National Park, Amazonian Ecuador. All woody species ≥ 10 cm dbh totalled to 228 species (728 individuals) and 149 species (417 individuals) in unflooded and flooded plain forests respectively. The total basal area was 33.7 m²ha⁻¹. Lieberman and Lieberman (1987) enumerated all trees ≥ 10 cm dbh on a 12.4 ha plot, in three locations, measuring 4.4, 4.0 and 4.0 ha of primary tropical wet forest at La Selva Biological station, Costa Rica. The trees were tagged and measured. The density was 1838, 1578 and 2114 individuals, species richness was 172, 171 and 165 respectively. It totals to 5530 trees representing 269 species in the 162 genera and 62 families. Basal area was 27.82 m²ha⁻¹. The steepest plot harboured the greatest density (529 ha⁻¹) and lowest basal area (25.54 m²ha⁻¹).

Pascal et al. (1988) reported a density and diversity of 103 and 22, 35 and 17, 47 and 15, 24 and 13 for trees ≥ 10 cm dbh from their four plots measuring 0.16, 0.05, 0.05 and 0.05 ha respectively in ‘Kan’ forest of Karnataka. The Shannon index ranged from 3.71 to 4.5 and the basal area ranged from 70.1 m²ha⁻¹ to 34.2 m²ha⁻¹.

Proctor et al. (1988) measured all trees ≥ 10 cm dbh in 10 plots of 0.04 - 0.4 ha, situated between 280 - 870 m in a primary rainforest of Gunung Silam, Sabah, Malaysia. They found a total of 374 species in a cumulative area of 2.6 ha representing 47 families. The species richness further increased to 412 when trees with a minimum of 5-cm dbh were considered.
Swan Jr. (1988) made a comparative vegetation analysis in two 0.24 ha plots (total of 0.48 ha) along an elevational gradient in the Fern valley and Jungle valley of Bukit Timah Nature reserve, Singapore. All stems ≥ 2 cm dbh were measured, identified and mapped at a scale of 1 cm = 1 m. Stand density was 564 and 1000 stems and species richness was 171 and 214 species respectively. The basal area was 22.3 and 28.9 m²ha⁻¹ respectively.

Bongers et al. (1988) made floristic inventory in a lowland tropical rainforest at Los Tuxtlos, Mexico, measuring all woody individuals ≥ 1.0 cm dbh on a 1-ha plot. There were 2976 individuals representing 185 species and 72 families. The basal area was 38.1 m². Trees (≥ 10 cm dbh) totalled 346 stems with 34.9 m² of basal area.

Whittaker et al. (1989) studied various sample (900 to 2500 m²) plots on the Krakatau islands, Indonesia. A total of 67 species (≥ 30 cm gbh) was recorded from 19 plots on Rakata, 13 on Sertang and 5 on R.Kecil. Within each plot the composition and relative abundance of the ground flora, epiphytes and climbers were also recorded.

Van der Moezel and Bell (1989) sampled the species richness in 16 sites (area 1000 m²) of the mallee region of southern-Western Australia. Richness varied from 17 species to 48 species per 1000 m². Sites with the highest species richness occurred on soils with the lowest nutrient content.

Jha and Singh (1990) studied the vegetation types of dry tropical forest in the Vindhyan region, central India selecting 42 sites in the entire region (ca. 500 km²). Ten randomly placed 10 x 10m quadrats were analysed phytosociologically. All woody species ≥ 30 cm gbh were measured. The stand density varied from 294 - 539 stems ha⁻¹ and the stand basal area from 6.6 m²ha⁻¹ to 23.2 m²ha⁻¹.
Manokaran and LaFrankie (1990) analysed population structure and spatial patterns of all woody species attaining $\geq 1$ cm dbh on a 50-ha plot at Pasoh forest reserve, Negeri Sembilan, Malaysia. Mean density was 6769 trees ha$^{-1}$ for tree stems $\geq 1$ cm dbh and 530 stems ha$^{-1}$ for $\geq 10$ cm dbh representing overall 817 species (Condit et al., 1996).

Williams-Linera’s (1990) study focused on micro-environmental conditions, vegetation structure and tree mortality in a ten month to twelve-year-old tropical premontane wet forest of Panama. In his 0.8 ha study on four randomly located 10 x 20 m strips, the density was 659 stems ha$^{-1}$. The basal area averaged 35.9 m$^2$ha$^{-1}$. There was a total of 2738 stems, 47 species and 16 families of woody plants $\geq 2$ m tall.

Heaney and Proctor (1990) studied a series of six 1-ha plots at different altitudes from 100 m - 2600 m of an unbroken sequence of rainforest in Volcan Barva, Costa Rica. The stand density (trees $\geq 10$ cm dbh) in the six 1-ha plots was 494, 391, 546, 553, 448 and 617, species richness was 111, 135, 109, 65, 69 and 35 species respectively. The species richness was highest in the plot at 500 m with 135 species and least at 2600-m site. The basal area was least at 100 m (22.7 m$^2$) and highest (51.2 m$^2$) at 2600 m.

Rankin-de-Merona et al. (1990) studied forest structure, diversity and interactions of fauna in the rainforest of North Manaus, upland Amazonia. All woody species $\geq 10$ cm dbh in 5 - ha scale contained 176 species and 3125 trees with a mean of 625 stems ha$^{-1}$. Gentry and Terborgh (1990) inventoried 1- ha of an alluvial flood plain forest in Cocha Cashu, Manu National Park, Peru. All individuals $\geq 10$ cm dbh totalled to 673 trees and 201 species comprising 47 families. Rashmi Lal (1990) studied three 0.5 ha plots in the tropical rainforest of South Andamans. The density
and richness of the 3 plots were 590 and 96; 496 and 85 and 441 and 61 respectively. Shannon index for the 3 plots was 3.9, 3.795, and 3.832 giving a value of 4.122 for the whole 1.5 ha study area.

Jeffrè and Veillon (1990) made a comparative floristic account of slope forest and alluvial forest in south of New Calédonia. They enumerated in the slope forest 309 species on 2.79 ha and having 1533 stems of ≥ 10 cm dbh. It was more diverse and denser than the forest on alluvium, which contained 219 species and 1183 stems in 2.68 ha. The basal area was greater on the slope forest (49.54 m²ha⁻¹) than alluvial forest (47.01 m²ha⁻¹). The alluvial forest harboured 142 genera and 67 families compared to that of 166 genera and 70 families in the slope forest. The two forests had 178 species in common. Weaver and Murphy (1990) conducted diversity studies in three 0.4 ha plots of lower montane wet forests of Luquillo mountains, Puerto Rico. The species richness was 50, 50 and 90, the stand density was 1750, 1620 and 1850 with basal area contribution of 40, 30 and 40 m²ha⁻¹ respectively.

Mosango (1991) studied the diversity, density and biogeochemical aspects of an equatorial forest of Kongolo Island, Zaire. Trees ≥ 30 cm gbh were measured in 2.5 ha. The density and diversity of trees ranged from 440 - 553 trees ha⁻¹ and 207 to 332 species ha⁻¹ respectively. The basal area ranged from 10 - 45 m²ha⁻¹.

Williams-Linera (1991) analysed tree stratum structure and floristic composition of a cloud forest site located between 1850 - 2150 m altitude in the El Triunfo biosphere reserve, Chiapas, Mexico. Density and basal area of trees ≥ 10 cm dbh were 590 ha⁻¹ and 53.0 m²ha⁻¹ respectively, whereas for trees ≥ 5 cm dbh, it was 960 ha⁻¹ and the basal area was 54.4 m²ha⁻¹.
Sukumar et al. (1992) as a part of long-term vegetation monitoring in a tropical deciduous forest in Mudumalai, southern India enumerated all woody plants of ≥ 1 cm dbh. They recorded 25,929 live stems, belonging to 71 species in a 50 ha plot. Trees of ≥ 10 cm dbh totalled 15,417 individuals with 63 species.

Campbell et al. (1992) compared phytosociology and dynamics of three flood plain (Vârzea) forests of known ages in Rio Juruá, Western Brazilian Amazon. Linear transects (500 x 20 m) of 1-ha were constructed in three places. All trees ≥ 10 cm dbh were measured, mapped and identified. There were 523, 420, 777 trees and 106, 73 and 20 species respectively in plots 1, 2 and 3. Shannon diversity index of 3.6, 3.51 and 1.82 and stand basal area of 25.46 m², 27.01 m² and 25.72 m² were obtained for the three plots. Thompson et al. (1992) studied species diversity in lowland evergreen forest of Mara, Brazil on six 0.25 ha plots. The richness of woody species ≥ 10 cm dbh was 77 species ha⁻¹; mean density was 419 trees ha⁻¹ and the basal area 23.8 m²ha⁻¹.

Medley (1992) made an inventory of the woody flora and ecological study of 12 forest areas (of 122 m²) in Tana river National Primate reserve, Kenya. He described the composition and structure of the riverine forest. There were 173 species ha⁻¹ (for trees ≥ 10 cm dbh) and 49 families. Tree density was low, 409 ha⁻¹ and the basal area was 23.1 m²ha⁻¹. Geldenhuys (1992) studied floristic richness, size, composition and inter relationships of the floras of two different (from mountain to coast) geographic regions in South Africa. The forest harboured 1438 species (woody and herbaceous) that belonged to 155 families and 661 genera.

Newbery et al. (1992) worked out the structure, relative abundance and family composition of two 4-ha plots at primary lowland dipterocarp forest at Danum Valley, Sabah, Malaysia. Trees ≥ 10 cm gdbh were measured, mapped and identified. A total
of 511 species (164 genera and 59 families) were recorded in a total sample of 17,985 individuals. Trees of ≥ 30cm ghn comprised 484 and 455 ha⁻¹ in 247 and 242 species in plot 1 and plot 2 respectively. The mean basal area for trees ≥ 30 cm ghn was 26.36 m² and 26.79 m²ha⁻¹ respectively.

Parthasarathy et al. (1992) investigated woody species diversity and human impacts in a tropical evergreen forest in Kalakad National Park, southern Western Ghats, India. In their six 1-ha plots species diversity and stand density (≥ 30cm ghn) were 77 and 852,64 and 856,85 and 915,84 and 725,82 and 885, 80 and 574 trees ha⁻¹ respectively. The stand basal area was 59.73, 77.38, 55.3, 94.64, 64.87 and 61.7m² ha⁻¹.

Primack and Hall (1992) made a quantitative inventory of 0.6 ha in the rainforest of Sarawak, Malaysia. Woody species ≥ 10cm dbh accounted for density range of 250 - 500 ha⁻¹ and the species richness ranged from 22 to 37 ha⁻¹. Felfili and de Silva (1993) made a comparative study of the floristics and phytosociology of woody vegetation in central Brazil. All trees ≥ 30 cm dbh were sampled in six 1-ha plots. A total of 6971 trees belonging to 139 species and 42 families were encountered. Each 1-ha plot had a combination of 50 - 80 species and approximately 30 families. The basal area ranged from 5.79 to 11.3 m²ha⁻¹.

Keel et al. (1993) investigated tree and liana density in primary forests of eastern Paraguay. All individuals ≥ 2.5 cm dbh measured in three 0.1 ha plots (100 x 10 m) totalled to 85, 50 and 64 tree species in 30, 24 and 28 families. There were 92, 27 and 50 liana individuals representing 29, 13 and 12 species in the three plots respectively. Geldenhuys and Murray (1993) studied floristic and structural composition of Hanglip forest in Soutpansberg, North Transvaal, South Africa. All trees ≥ 10 cm dbh were measured on 363 ha of evergreen forest which was gridded.
into 62 circular plots of 0.04 ha in six selected stands. The mean stand density ha⁻¹ was 582, 664, 494, 590, 449 and 565 and the number of species totalled to 33, 29, 16, 29, 28 and 17 respectively. Among the six stands 46 species were enumerated. The mean stand basal area in m²ha⁻¹ was 21.2, 35.9, 56.6, 67.6, 58.1 and 72.1 respectively.

Jayasingham and Vivekanantharaja (1994) studied high-disturbed and open forest types in tropical monsoon forest at the Wasgonuva National Park, Sri Lanka. All plants ≥ 10 cm dbh were measured in 0.3 ha (three 100 x 10 m) area. The mean tree density per 0.1 ha was 34.8, 28.5 and 26 respectively, with mean richness of 10.2, 9.3 and 7.9. The total mean basal area was 7.071, 5.638 and 0.2988 m² per 0.1 ha respectively. Valencia et al (1994) enumerated a total of 693 trees (≥10 cm dbh), 307 species, 138 genera and 46 families in a 1-ha square plot of terrafirme forest, Amazonian Ecuador. This is the highest record of tree species richness for a tropical rain forest sample of this size. The most speciose families were Fabaceae sensu lato, Lauraceae and Sapotaceae.

Sussmen and Rakotozafy (1994) studied diversity, density and floristic composition of 2 distinct habitats of gallery forest in the Beza Mahafaly reserve, southwestern Madagascar. All woody plants ≥ 2.5 dbh were measured in 25 transects (50 x 2 m) totalling to 0.25 ha. There were 923 individuals, 69 species and 43 genera. Johnston and Gillman (1995) studied floristic composition and stand structure in four 1-ha plots in lowland tropical forests of Kurupukari, Guyana. The density was 2035 trees in 120 species and 30 families for trees ≥ 10 cm dbh threshold, whereas in the girth of ≥ 5 cm dbh there were 3897 trees in 153 species and 31 families. Total basal area varied from 32.39 to 34.63 m²ha⁻¹.
Visalakshi (1995) made vegetation analysis of two tropical dry evergreen forests, Marakanam reserve forest (MRF) and Puthupet sacred grove (PSG) on the Coromandel coast of southern India. Three 0.1 ha plots of MRF harboured a mean stand density of 280 individuals ha\(^{-1}\) for all woody plants (≥ 20 cm gbh) and the basal area was 11.1 m\(^2\)ha\(^{-1}\), whereas in PSG stand density was 1130 individuals ha\(^{-1}\) and the basal area was 36.9 m\(^2\)ha\(^{-1}\). Shannon index for MRF and PSG sites was 2.19 and 1.78 respectively.

Nadkarni et al. (1995) described the structure and composition of the primary montane forest in Monteverde, Costa Rica. All stems of both ≥ 2 cm and ≥ 10 cm dbh were measured. There were 555 individuals, 33 genera and 114 species. The total basal area was 62 m\(^2\)ha\(^{-1}\). Zhang and Cao (1995) enumerated tree species diversity (≥3 cm dbh) in two plots (50x50m; 40x40m) that totalled to 43 species and 142 stems and 14 species and 72 stems respectively in a tropical forest at Xishuangbanna, China.

Ojo and Ola-Adams, (1996) inventoried more than 315 species of trees of ≥ 5 cm dbh on 5 sites in a Nigerian rain forest. There were 80, 226, 163, 137 and 159 species in the five sites namely Gambari, Oban, Omo, Owan and Sapoba. Strasberg (1996) enumerated the floristic richness and structure of a 1-ha permanent lowland tropical rainforest on the Oceanic island of La Reunion. All woody species ≥25 cm gbh were measured. A total of 1270 individuals and 43 species were inventoried respectively. The total basal area was 82.67 m\(^2\).

Ganesh et al. (1996) enumerated 2708 trees (>10 cm dbh) belonging 90 species on a 3.2 ha, in an undisturbed middle elevation evergreen forest in Kalakad-Mundanthurai Tiger reserve, southern Western Ghats. Pascal and Pelissier (1996) studied the structural characteristics and ecosystem functioning of Uppangala.
(Karnataka) evergreen forest in the Western Ghats, India. All trees \( \geq 30 \text{ cm gbh} \) were enumerated in five strips, totalling 3.12 ha. The stem density was 635 ha\(^{-1}\) and the basal area 39.7 m\(^2\)ha\(^{-1}\). There were 91 species belonging to 31 families.

Hare et al. (1997) investigated the structure and composition of trees (\( >2.5\text{ cm dbh} \)) in a sub-tropical dry forest of Dominican Republic, Puerto Rico. In five 50x50 plots had 3646 stems ha\(^{-1}\) belonging to 43 species (29 identified and 14 unidentified species). The total basal area of live stems was 7.5 m\(^2\) ha\(^{-1}\). Cao and Zhang (1997) enumerated 72 species (257 stems) and 52 species (184 stems) (of \( \geq 10\text{ cm dbh} \)) in two 2500m\(^2\) plots in a tropical seasonal rainforest of Xishuangbanna, China.

Parthasarathy and Sethi (1997) determined diversity and population structure of trees and lianas (\( \geq 10\text{ cm gbh} \)) in a tropical dry evergreen forest at Puthupet on the Coromandel coast, south India, totalling 2675 stems on a 2-ha scale (1338 stems ha\(^{-1}\)) in 51 species, with mean basal area of 32.8 m\(^2\) ha\(^{-1}\). Parthasarathy and Karthikeyan (1997 a) made an inventory of all woody species (\( \geq 10 \text{ cm gbh} \)) in two tropical dry evergreen forests at Kuzhanthaikuppam (KK) and Thirumanikkuzhi (TM), on the Coromandel coast of southern India, selecting 1-ha square plot in each of the sites. Species richness was 42 in KK and 38 in TM. A total of 2413 trees, 54 species in 31 families were encountered in the two plots. The stand density was 1367 in KK and 964 in TM. Stand basal area was 15.44 and 29.48 m\(^2\)ha\(^{-1}\) for the two sites respectively.

Parthasarathy and Karthikeyan (1997 b) made an inventory of tree diversity, density and population structure of a tropical wet evergreen forest in Mylodai Courtallum R.F., Western Ghats, India. There was a total of 57 woody species (\( \geq 30 \text{ cm gbh} \)) in 47 genera and 32 families. The Shannon's index was 3.069. The stand density was 482 stems ha\(^{-1}\).
Ghate et al. (1998) explored the patterns of tree diversity in relation to vegetation type in the whole chain of the Western Ghats of India. There were 20,785 individuals (≥10 cm dbh), belonging to 398 species in 108 belt transects covering a total area of 75-ha, comprising 7 vegetation types with distinctive species composition.

Buttress

Buttresses are prominent in some forest formations. They differ in shape, size and thickness between species and families (Whitmore, 1990). Buttresses are tension structure, resonating when struck with an axe and are mainly found on uphill sides of trees and counter balancing asymmetric or epiphyte laden crowns. Buttresses are reported to be of structural importance helping to support trees. Flying buttress occurs in some families and is common in swamp forests (Whitmore, 1990).

Buttresses have developed the reaction wood in only five of 48 species examined (Fisher, 1982 as in Whitmore, 1984). The different incidence of buttresses in different species can be ascribed to intrinsic differences in construction of the root wood. The incidence and size of buttresses diminishes with altitude and outside the tropics. Part of explanation must be that smaller trees have less need for support (Whitmore, 1984). The presence of buttressed trees is a common feature in tropical rainforests.

In the lowest tree storey except among young trees which when mature will belong to the higher storeys, the buttressing is almost absent. There is a general correlation between the height of trees and the production of buttresses. The shape of buttresses is often highly characteristic of particular species and even within the
species show plasticity in this regard with some individuals having large buttresses with others of the same having none (Richards, 1996).

In fact the distribution of the buttressed habit shows that the climatic condition under which it is most strongly developed, over a high rainfall combined with a high mean temperature (Richards, 1996). There is a close correlation between the frequency of buttressed species, the size of the buttress and the nature of the soil (Davis and Richards, 1933 & 1934). In montane rainforests buttressing is absent or uncommon as was observed in mossy forest on Mt. Dulit, Sarawak, Malaysia (Richards, 1996).

Some families show greater tendency to produce buttress than others. Leguminosae and Sterculiaceae include many species having large buttresses, while in Annonaceae, Fagaceae and Lauraceae, well developed buttresses are usually absent and among the Malaysian species they are generally absent in Myristicaceae (Richards, 1996).

Paijmans (1970) found that the ratio of height/width of the buttress increased in lower girth classes up to 229 and the trend was reversed with the width of buttress tend to increase more than the height. Percentage of buttressed trees increased with girth. The number of buttresses per tree was most often 3 or 4.

In the montane and lowland rainforests of Ecuador, Grubb et al. (1963) found that the three tropical lowland rainforests had usually large buttressed trees or usually frequent-buttressed trees, while in the montane rainforests buttressed trees were totally nil.
Rarity

If the relative abundances of species in a particular plant or animal group in a given community are somehow measured, there will be some common species, and some rare species and many species of varying degrees of rareness (May, 1975). MacArthur (1965) revealed that in tropics the species could be rare without running greater danger of extinction, due to fewer climatically extreme conditions.

Geldenhuys and Murray (1993) studied the floristic and structural composition of Hanglip forest. All trees $\geq$ 10 cm dbh were measured on 62 circular plots of 0.04 ha and six stands were identified on the basis of the TWINSPLAN. It was found that the species Octotea kenyensis considered as rare species and was only recorded in 5th and 6th sites.

Paijmans (1970) in his 3.2 ha study found that 146 species out of the total 392 species had single representatives. Rare trees normally would tend to be clumped to ensure reproductive success. This might be due to the rarity of the microsite needed by the species (Hubbell, 1979). Poore (1968) found that at least some of these were determined by soil differences. Thorington et al. (1982) in their 5 ha study in Barro Colorado Island, Panama found 45 species of the total 112 species had a maximum of two representatives. Ho et al. (1987) in their 11.7 ha study in Malaysia found 154 species of the total 261 species were represented by a maximum of 2 individuals.

Many species are represented by single individual in the moist tropical unflooded and flooded rainforests in Yasuni National Park, Ecuador. In a sample size of 420 trees in 2 transects, 97 species (55%) in the unflooded forest and 92 species (62%) in the flood plain forest were represented by single individual (Balselv et al. 1987).
Newbery et al. (1992) found that at least 36% and 67% of the species having single tree respectively in the two 4-ha plots of Danum valley, Sabah, Malaysia. Manokaran and Kochummen (1987) studied the structure and composition a lowland dipterocarp forest for a 34 years (1947-1981) in Sungei, Menyala forest reserve, Malaysia and reported that the uncommon species represented by single tree, was 87(35.8% of species) in 1947 and 98(40.2%) in 1981.

Population structure

Poore (1968) studied the population structure of trees ≥ 30 cm gbh for individual species as well as in two families, Dipterocarpaceae and Burseraceae. He found expanding ‘L’ shaped tree distribution. Dipterocarpaceae were represented by many more large trees than the remaining families and the number of individuals of dipterocarps decreased more slowly with increasing girth.

Paijmans (1970) obtained leptokurtic distribution of trees ≥ 30 cm gbh using a class interval of 3m in his four study plots in New Guinea. Both, the number of species and the number of individuals decreased with increasing girth, but the number of species not so sharply. The presence of largest trees in plot 1 (girth between 159 - 279) such as Euodia sp., Sloanea sp., Endospermum sp. and some Lauraceae indicates the role of disturbance probably a 100-year or more ago.

To determine population structure of trees in tropical moist forest of BCI, Knight (1975) established seven dbh size classes. The total number of individuals in all the quadrats belonging to each size class was determined for each species and was divided by the total number of tree individuals in all size classes of all species in the stand, thus giving relative density data for each species size class, to have an estimate of population structure and was used for evaluating the succession status of each
species. According to Hubbell (1979) if a population is composed largely or exclusively of adults then either the population is declining or else reproduction in the species is episodic in nature.

Singh et al. (1984b) classified the girth distribution of species in Silent Valley tropical rainforest, Kerala, as expanding, decreasing, youthful, interrupted and accidental. According to him species-rich-forests have a low percentage of expanding population structures. They stated that the high species richness in Silent Valley is made possible in part by the species combination varying from one girth class to another. Thus species are in constant flux in space and time and this is in turn is possible when suitable habitats of sufficient size are available to encompass all the stages of growth of all species. Thus, there occurs as a mosaic pattern or cyclic change in regeneration giving rise to variation in the combination of dominants.

Rai and Proctor (1986) studied the allometric distribution, girth and basal area at 5 cm intervals in lowland rainforest, Karnataka. There was a decrease in number of trees from lower size class to higher size class. In the case of basal area, 2 sites showed bimodal distribution and another two showed opposite trend to the girth distribution.

Campbell et al. (1986) studied the distribution of trees by class intervals of dbh in terra firme and Várzea forests, which showed a typical, reverse ‘J’-shaped curve.

Swaine et al. (1987) in their study at Kade, Ghana showed that the distribution of size classes is typical of natural forest regenerating from seed with high numbers in the smaller size classes and a more or less logarithmic decline in number with increasing size suggesting a stable size and age distribution.
Manokaran and Kochummen (1987) in their study on a lowland dipterocarp forest in Malaysia from 1947 - 1981 found that the girth distribution for trees ≥ 10.1 cm dbh in a 2.02 ha plot to be 'L' shaped with no real change or the diameter class distribution for 1947 and 1982 was very similar.

Lieberman and Lieberman (1987) reported scanty appearance of large trees ≥ 10 cm dbh in La Selva, Costa Rica. Mortality was independent of size class in trees ≥ 10 cm dbh. Understorey trees grow slowly and consistently, and have short life spans. Sub-canopy trees also grow slowly, but live longer; shade-tolerant canopy trees show various and often rapid growth and are long-lived; and shade-intolerant canopy trees grow rapidly, show little variation and have short life spans.

Ho et al. (1987) found that the frequency of all girth class showed a negative exponential form whilst the number of trees of girth classes 0.910 m and 1.2 m were similar. Species with ≥ 19 individuals were divided into light-demanding and shade-tolerant species, based on the number of trees in different girth classes. The former have a preponderance of big trees and the latter a greater number of small ones. None of the common species was restricted in its natural girth range to suggest these species were invading or dying out in the plot. Maps of the ten most frequent species showed no clear evidence of an uneven distribution of girth classes along the length of the plot.

Pascal et al. (1988) in four forest plots in ‘Kan’ forests Karnataka, found that all tree species as a whole, as well as for some important species are in leptokurtic girth distribution.

Williams-Lineara (1990) showed density and basal area of woody plants (<5 cm dbh) decreased with distance from the pasture forest border into the forest.
interior except along the forest edge, of the most recently cleared area at tropical premontane wet forest of Panama.

Balslev et al. (1987) studied the distribution of trees in different size classes from both moist flooded and unflooded forests. Trees exhibited a reverse J shaped curve for undisturbed forest. The unflooded forest had higher proportion of individuals with ≤ 25 cm dbh when compared to flood plain forest. Tree distribution by height class formed a skewed bell-shaped type.

Russell-Smith and Lee (1992) made a comprehensive survey of rainforest vegetation in Northern Territory, Australia. Population size classes of presumed adults of each species were censused in 394 (<2.5 ha) rainforest patches. It was found that the populations of adult plants were typically very small. Most species were represented by less than 50 adults in 81% of patches. Adult population size is dependent on patch size. Of the 137 common rainforest species occurring in 10% or more sample sites only 20% of species typically occurred in population of more than 50 adults per patch. Newbery et al. (1992) obtained frequency distribution of species on two 4-ha plots that showed more or less similar distribution with respect to logarithmic \( \log_{10} \) abundance classes for trees ≥ 10 cm and 100 cm gbh.

Sukumar et al. (1992) found that the deciduous forest of Mudumalai had a more or less expanding population structure with a deficiency of individuals in the smallest size class. Some species had few or no individuals in the smallest size class (< 5 cm dbh) as compared to the largest one.

Geldenhuys and Murray (1993) analysed the population structure of selected dominant species. Size-class distribution (both height and diameter) has been used to infer development trends of species and forests. The population structure of most
canopy species exhibits a negative exponential or inverse ‘J’ shaped curve for a mixed evergreen forest in South Africa.

Johnston and Gillman (1995) worked out the population structure of trees in four 1-ha plots in the lowland forest, Kurupukari, Guyana. The number of species and individuals versus girth class exhibits a reverse J-shaped histogram. In the tropical evergreen forest of Uppangala (Pascal and Pelissier, 1996) and Mylodai area of Courtallum R.F. (Parthasarathy and Karthikeyan, 1997 b) the forest stand exhibited an expanding population structure.

Spatial patterns

Plant populations exhibit three patterns of spatial distribution - regular or uniform, clumped or aggregated, and random. The individuals of a species is said to be random if the position of each individual plant is independent of that of all the others; aggregated populations, are those where there is a tendency for individuals of the species to occur in clumps, and in regular populations the plants are more evenly spaced than they were distributed according to chance (Pielou, 1960).

Clumped distribution is very common in nature and two causes have been suggested for this (Feller, 1943). Firstly the seeds may fall at random over an area but if the habitat is not homogeneous, the proportion of germinating and thriving will vary from site to site so that density is high in some sites and low in others. Secondly the habitat may be homogeneous, but the individual plants may occur in family groups, owing to the fact that they reproduce vegetatively or by seeds with a small radius of dispersal.

Uniform distribution is extremely rare. According to Grieg-Smith (1983) a regular pattern would be expected, if the members of a population were so abundant
that they compete with each other for available space. Most available published works indicate a prevalence of clumped and a paucity of regular distribution patterns of tree species in tropical forests.

Poore (1968) mapped trees ≥ 91 cm dbh in a 26 ha (620 x 420 m) plot of Malaysian lowland rainforest and found that 6 of the 13 most abundant species were randomly distributed while 7 were clumped. Uniform distribution was found in only one species. Forman and Hahn (1980) demonstrated that the spatial patterns of 16 most common species (≥ 10 cm dbh) in a 4-ha forest plot at St. John, United States of Virgin Islands. They found three quarters (12 species) showed clumped pattern of distribution, but only one species had uniform distribution, while the remaining 3 species showed random pattern.

Lang (1969) and Lang *et al.* (1971) studied spatial patterns of trees (≥ 2.5 cm dbh) in a 1.5 ha plot in BCI, Panama. Of the five uncommon species, 2 were clumped and 3 random. Among the seven species 6 were clumped and one random and all the 3 abundant species were clumped.

Thorington *et al.* (1982) found that over dispersed of seeds and seedlings of trees are commonly eaten by animals in large quantity, but it is not a dominant factor for conspecific species. The highest percentage (41 %) of trees were clumped among five 1-ha plots in the tropical moist forest of BCI, Panama. They mapped trees > 60 cm dbh in five 1-ha plots. Of the 63 species 26 species showed significant clumping. Clumping of tree species significantly acts as seasonal use by the animals (especially howler monkeys), its survival and reproduction.

Spatial patterns may be determined by habitat, alternative population recruitment strategies and differential competitive ability of seedlings (Janzen, 1970).
In addition to high juvenile mortality caused by density or distance responsive predators, pathogens etc., as postulated by the escape hypothesis implies disproportionate success for seeds that escape the vicinity of the parent, as compared with those that fall near by parent (Howe and Smallwood, 1982). Swan (1988) reported a high degree of clumping in tree fall gaps due to the pressure of regeneration patches.

Hubbell (1979) stated that the dry forests differ from rainforests especially, in a smaller number of species; it does not represent a general pattern of tropical forest structure. The distribution of adults and juveniles on the plot showed that the trees were not uniformly distributed. Over 75% of species were clumped and the rest 24% showed a random distribution.

Sukumar et al. (1992) at Mudumalai tropical deciduous forest found that the most of the species showed clumped pattern of distribution, with the exception of *Gmelina arborea*, which was randomly dispersed.

Poortor et al. (1994) determined spatial distribution of canopy gaps in three sites (total 71-ha) in tropical moist forest of Tai National Park, Ivory Coast. The study revealed a clustered distribution of gaps for two of the three sites. The mean number of gaps per hectare and the percentage area in gap phase are remarkably similar for the three sites. Both direct and indirect evidence showed that gaps may influence the spatial distribution of trees in gaps but could also be the result of clustered character of gap distribution on its own. Visalakshi (1995) reported that most species were clumped at 1000 m² scale, in two tropical dry evergreen forests, while species such as *Canthium dicoccum* and *Pterospermum canescens* were randomly distributed.
According to Hubbell and Foster (1983) most of the tree species appear to have their own individualistic dispersion patterns in their 50-ha plot study in BCI, Panama, with only weakly developed association by habitat in some species, and most of the species being unevenly distributed and many exhibiting clumped dispersion.

According to Janzen (1970) and Connell (1971) any event that increases the efficiency of predators by eating seeds and seedlings of a given tree may lead to a reduction in population density of the adults and for their parents. Either events will lead to more space in the habitat for other species of trees and therefore a higher total number of species, provided seed sources are available over evolutionary time. The net result is that this density, or distance dependent predation will result in uniformly spaced adult trees.

Clark and Clark (1984) studied population of *Dipteryx panamensis*, a tropical rainforest tree, in order to evaluate the Janzen and Connell model. According to them the Janzen and Connell spacing hypothesis has been impeded by the inappropriate use of spatial dispersion pattern indices as test criteria. This is because a reasonable expectation from the model would be clumping of young trees around adults but at a lesser degree than if progeny survival were unaffected by density or proximity to adults.

Armesto *et al.* (1986) compared the spatial patterns of tree species (≥ 10 cm dbh) in 2 north temperate, 2 south temperate, 1 sub tropical and 3 tropical forests. An index of dispersion based on distances to nearest conspecifics was used to infer tree spatial patterns. Clumped patterns were predominant in all tropical forests (75 - 100% of species) and in one north temperate forest. In 2 temperate forests and 1 north temperate one, random patterns were common (50-88%). The difference in the
proportions of random versus clumped spatial patterns seems to be related to different histories of disturbance in the forests compared.

Webb et al. (1967), Ashton (1972) and Austin et al. (1972) indicated that in the absence of major disturbance, soil and water conditions play major roles in controlling species distribution.

Paijmans (1970) study in New Guinea showed that most species were more or less randomly distributed although instances of patterned distribution were found in each plot, particularly in plot 1 where Sloanea sp.1, Lithocarpus and Euodia sp. were clumped in the western part, while having few or no individuals in the remainder of the plot. According to him the factors governing spatial distribution such as variation in environmental history and chance may all be operating at once.

Armesto et al. (1986) postulated that herbivore impact is not greater in tropical than in temperate areas but equivalent or even lower as can be seen from the predominance of clumped patterns in tropical areas and random patterns in the temperate ones. The spatial scale of natural disturbances and the time frame involved, play significant roles in determining the frequency of clumping and randomness in forest communities. Additionally the scale of disturbance interacts with the underlying resource patchiness of the local area, which form heterogeneity of substrates, steepness of slope etc. (Armesto et al., 1986).

In forests subjected to frequent disturbance (relative to the life cycle of dominant species) the original resource patchiness is obliterated or reduced leading to random pattern in the disturbed areas. Initially colonisation may occur in clumps due to seed dispersal patterns and vegetative spread (Hibbs, 1973). Later with the thinning
of clumps (Kershaw, 1973; Grieg-Smith, 1983) most trees should be randomly scattered (Williamson, 1975).

As the forest matures clumping becomes common in association with gap dynamics (Putz and Milton, 1982; Brokaw, 1982; Runkle, 1982), but this is again interrupted by frequent disturbances. Canopy gaps formed due to tree falls result in increased light mound upheaval and nutrient release (Bormann and Likens, 1979; Putz, 1983). This leads to predominantly clumped tree patterns with selection for gap-dependant life cycle (Grubb, 1977).

Newbery et al. (1986) investigated two-dimensional pattern for the 64 most abundant species (≥ 10 cm dbh), by spectral analysis using the basal areas of trees in 20 x 20 m contiguous quadrats in Kerangas forest in Sarawak. Thirty species showed clumped pattern of distribution. The most frequent scales of clump size were between 35 and 55 m across.

Lianas

Lianas are woody climbing plants that rely on other plants for mechanical support. Their axes have reduced amounts of supporting tissue and they are highly light demanding. Putz (1983) studied liana biomass, diversity and leaf area of a terra firme forest in Rio Negro basin, Venezuela, San Carolos de Rio Negro. From twenty sample plots covering 0.2 ha, he enumerated 45 species of lianas and an average of 34.5 individuals (s.d. = 9.8) of 11.4 species (s.d. = 2.2) in each plot; 39% of these were self supporting.

Putz (1984) determined liana abundance, height, diameter and analysed climbing mode in mature tropical moist forest on Barro Colorado Island, Panama. All lianas ≥ 1 cm dbh in 10 randomly located 0.1 ha plots were measured, in which 3165
lianas, (both climbing and supporting upright lianas) ≥ 10 cm tall representing 65 species. He stated that lianas play a major role in forest dynamics. Trellis availability was found to be a major factor limiting lianas, access to the forest canopy tree and liana stems on the edges of the tree fall gaps provided a major pathway to the canopy for climbing plants. Trees carrying lianas suffered higher mortality rates and upon falling caused more other trees to fall than did liana-free trees.

Putz et al (1989) discussed the biomechanics of twining to silvicultural methods for controlling vine infestations. Woody vines differ from trees primarily in stem allometry and anatomy. Tree stems thicken substantially to support the growing crown; liana stems rely on other plants for mechanical support and consequently remain thin. The anatomical structure of liana stem is associated with tremendous wood flexibility; the xylem of some lianas continued to conduct water when stems were twisted by 180°, but no tree wood tested withstood even a single rotation (Putz et al., 1989) The role of vines in forest communities was discussed. There is sufficient evidence that vines have deleterious effects on trees and that they flourish in disturbed areas. It was also noted that the presence of large diameter vines is a good indication that a forest has not been subjected to serious disturbance for many years (Putz et al., 1984).

Medley (1992) found that palms and lianas were generally abundant in riverine forest ecosystems as observed in Tana River National primate reserve, Kenya.

Campbell and Newbery (1993) studied the ecological relationship between lianas and trees in lowland rainforest in Sabah, Malaysia, in which all lianas ≥ 2 cm gdbh were recorded on trees ≥ 30 cm gdbh in two 4-ha plots. The plots harboured a total of 7054 lianas belonging to 22 species. Fifty seven percent of trees supported lianas.
Wolf (1994) studied the epiphytic vegetation on host trees along an altitudinal transect approximately 200m from 1000m up to 4130 m.a.s.l on the west slope of the central Cordillera in Colombia. Both vascular and non-vascular epiphytes, the latter often more abundant in the montane rainforest.

Schnitzler (1995) made a study on the phenerophytic liana (*Clematis vitalba* and *Hedera helix*) population structure of the woody layer in Rhine valley, France. The total surface covered was 6-ha (4.3 ha in Rhine forest; 1.7 ha in El forest). All individuals including the lianas *Clematis vitalba* and *Hedera helix* (> 2m high in young stages, 74 m high in older stages) were counted. Individuals > 6-cm dbh were measured. The most dense forest are *Ligustro-populetum albae* - 4900 stems ha⁻¹ young; 12600 stems ha⁻¹ when old). *Fraxino-populetum albae* 6562 stems ha⁻¹ young; 10432 stems ha⁻¹ when old. River forests are species-rich in old stands with high values of species richness up to 18.9 species ha⁻¹ and the poorest obtained only 4.3 species ha⁻¹. Liana-rich communities occur exclusively in the Rhine valley forests. Liana density (related to total woody density) peaks in the young mesophilic stands 20.4%, but 18.1% in *Ligustro-population nigrae*; the highest number of giant lianas is reached in *Ligustro-populetum nigrae* with 120 individuals ha⁻¹.

A common feature of all lianas is that their axes have reduced amounts of supporting tissues so that they have to climb on trees or any other support to reach light with the help of ascending stems or adventitious roots (Schnitzler, 1995). Lianas are considered as a nuisance by foresters because they overshadow the young trees, deform or break tree stems (Putz, 1991) and accelerate death of their hosts when ill (Beekman, 1981). Competition between lianas and hosts for water, nutrients and light was often been advanced as a cause of destruction, without any consideration of their
beneficial effects on ecosystems (role in reducing soil erosion, maintenance of animal diversity, contribution to annual leaf biomass etc. (Tremolieres et al., 1988; Putz, 1991). Hegarty and Caballe (1991) stated damaging the ecosystem by floods, however floods multiplying the number of gaps and the possibilities of colonisation by lianas. Access to light favoured by disturbance is the main determining factor for liana invasion at community scale.

Forest regeneration

Tree regeneration by seed is a critical and frequently studied component of forest succession (Jones et al., 1994; Grubb, 1977). Uhl and Murphy (1981) studied the composition, structure and regeneration in the terra firme rainforest of the San Carlos, southern Venezuela. They stated that most of the common species are well represented in all size classes indicating that the forest is floristically stable and added that the gaps are common in the forest and beneficial to the regenerating seedlings and saplings.

Putz (1984) studied the liana-overloaded trees following by tree felling in randomly located areas, represented by different stages of regeneration. In fact tree fall gaps enhanced the regenerating capacity on forest floor.

Ho et al. (1987) enumerated regeneration capacity of the soils in Jengka forest reserve, Malaysia. They observed the dominant establishment of Elateriospermum tapos (Euphorbiaceae) that regenerated profusely on shades or in small gaps. Floristically the segamat soil series differed from that of the Butu Anam series due to soil properties and is dominated by the dipterocarps.

Manokaran and Kochummen (1987) studied the recruitment, growth and mortality of the tree species in a lowland dipterocarp forest. They found at least ten
species were commonest in the plot. The best represented family was Dipterocarpaceae. Mortality was greater than average for understorey species and lower for emergents. Pioneer and late seral species together showed significantly higher mortality rates.

Williams-Lineara (1990) stated that the mean seedling recruitment and mortality were higher and total survival was lower on the forest interior of Panama. Species composition of seedlings in the experimentally disturbed sites resembled that of the soil seed bank. Soil disturbance treatment increased the density of woody plants and the number of light demanding species. Light demanding species however were rare along the edge and along older edges in the area, in part because soil disturbance normally did not occur during edge creation. Change in seedling density, recruitment and mortality with time after edge creation were similar in forest edge and forest interior edge. Plants were derived from the seedlings and saplings present before the edge creation.

Fensham and Bowman (1992) studied the influence of over-wood on regeneration in tropical Eucalyptus forest on Melville Island in which saplings > 2 m height and < 10 cm dbh were considered. Five target species produced mass of short woody sprouts in ground layer and low number of saplings. The formation of woody sprouts or sapling initiation may be related to the size of the ligno-tuber and the presence of a tap root for some species. It also indicated that disturbance, like fire or termites, herbivory and also the development of a tap root seems to increase the probability of sapling success.

Hickey (1994) studied the regeneration of wet Eucalyptus mixed forest in Tasmania and stated that logging and wildlife on a 80-100 year rotation considered
insufficient for redevelopment of mature forests. Barker and Kirkpatrick (1994) determined the range of the regeneration of *Phyllocladus asplenifolius* (Podocarpaceae) in terms of different microsite occupation. The species showed a preference for elevated sites including long buttresses and old stump mounds.

Chandrashekara and Ramakrishnan (1994) studied the vegetation and gap dynamics of a tropical wet evergreen forest at Nelliampathy in the Western Ghats of Kerala. The regenerating individuals of trees (>30 cm gbh) in hundred 10 x 10m quadrats, for sapling (10.1 < 30 cm gbh) in twenty five 5 x 5m quadrats for shrubs and twenty five 1 x 1m quadrats for tree seedlings were considered in the 100 x 100m area. Out of 33 tree species recorded, 8 were represented in seedling, sapling and mature stages, while 11 species were present only as mature trees and 11 as sapling and mature stages.

Jones et al. (1994) studied in regeneration in four flood plain forests of south Carolina. They estimated seed fall and recorded age, growth and survival of 10,933 tree and vine seedlings. The forests differed in flood frequency, soils and vegetation structure.

Forest degradation, human impacts and conservation

Singh et al (1984a) described the vegetation of Silent Valley, Kerala and stressed the need for conservation by making a plea to consider as it a Biosphere reserve. Hubbell and Foster (1986) studied the implications for tropical tree conservation in Barro Colorado Island, Panama. Salwasser (1990) enumerated the major factors affecting biological diversity that include, pollution, fragmentation of habitats, overuse of resources, conversion of wild areas to agriculture and industry.
and other human uses. Denich (1991) observed that deforestation causes secondary succession of tropical forests.

Hubbell and Foster (1992) emphasised that the basic and applied ecological research have a vital and cost effective role to play in tropical forest conservation and management and stated that sustainable management of natural tropical forest is not possible without a better holistic understanding of how such forests actually work ecologically and interact with humans.

Gentry (1992) studied two major patterns of diversity and important conservation planning - the distribution of diversity and endemism on tropical forests, in which high diversity forests occur in all three continents and are concentrated in lowland but with greatest areas with high and evenly distributed rainfall, in South American forests. Tree and liana species richness is greatest in upper Amazon, non-tree species richness greatest in N. Andean foothills and S. Central America, underlining conservation need of these area.

Hobbs and Huenneke (1992) stated that disturbance is an important component of many ecosystems. They revived the effects of disturbances such as fire, grazing, soil disturbance, trampling, fragmentation, and nutrient addition, on plant species diversity and invasion could change the native species and enhance the invasion of other natural communities.

Ledig (1992) stated that humans have converted forest for urban and agricultural uses, which cause degraded environment with atmospheric and soil pollutants. Deforestation causes a vast scale of forest areas to reduce diversity by direct elimination of both locally and adapted populations, which will bring atmospheric pollution and global warming as a major threat in near future, especially
because of the fragmented forest areas. Yadav et al. (1993) observed that shifting cultivation practised in Tirpura hill slopes cause destruction of forest wealth and accelerate land degradation.

Valkenburg and Ketner (1994) studied the floristic changes that occur following human disturbance in mid-montane forest in Papua New Guinea. Of the two forests, disturbed forest Mt. Kaindi and undisturbed forest Mt. Mission, Nothofagus pulleti was locally dominant in Mt. Kaindi. Change of floristic composition was observed between 1800 - 2000 m. The survival of seedlings and saplings of Nothofagus increased by light penetration. In addition natural events such as landslides, drought, fire and storms and also human activities are overruling the natural disturbances which cause the disappearance of species such as Nothofagus and Castanopsis due to lack of seed.

In forest areas traditional agriculture did not appreciably erode diversity where the population density remained low, however overall, in many tropical forests pressures like shifting cultivation, gathering forest products, monoculture etc. diminish the forest biodiversity.