II. REVIEW OF LITERATURE
Review of literature

Biodiversity of tropical forests

The introduction of the term biological diversity or its short form biodiversity is rather new, which emerged some twenty years ago (Lovejoy, 1980a; b; Wilson and Peters, 1988; Reid and Miller, 1989; McNeely et al., 1990; Chauvet and Oliver, 1993), but the origins of the concept go far back in time. Ideas regarding the linkages and relationships between organisms and their environment, both biotic and abiotic, were developed from the eighteenth century onwards, as naturalists such as Darwin, Humboldt and Wallace observed the patterns of distribution of species and vegetation types in their natural environments, but it was not until the early part of the twentieth century that ecology developed formal tools for the measurement and modeling of these relationships and their diversity. Palmer (1995) stated that species diversity appears to be the most straightforward concept of the components of biodiversity than the other two components namely the genetic diversity and community diversity.

One of the striking features of life on the earth is the concentration of biological diversity in tropical forests. Although many authors (Fisher, 1960; MacArthur, 1965; Pianka, 1966; Rosenzweig, 1968; Stevens, 1989) have noted increasing species diversity with decreasing latitude, few have focused on diversity differences among different tropical sites. While some researchers have focused on climatic correlates of diversity (Gentry, 1982; 1988a; Clinebell et al., 1995), and on contrasting altitudes (Grubb, 1971; 1977; Proctor et al., 1983; 1988; Heany and Proctor, 1990; Lieberman et al., 1996; Kitayama, 1992; Aiba and Kitayama, 1999; Srinivas and Parthasarathy, 2000), the others have emphasized soil nutrients as a determinant of tropical forest species richness (Huston, 1980; Gentry, 1982; 1988a;
Stark, 1970; Adam, 1989; Ashton, 1977; 1992). Phillips et al. (1994) documented that there is a strong positive correlation between a mature tropical forest’s productivity and its species richness.

Various theories have been proposed to explain the diversity maintenance in a tropical forest community and most of them are not mutually exclusive. Oversimplifying a bit, one can broadly divide theories of diversity maintenance in community ecology into two classes - niche-assembly theories and dispersal-assembly theories (Hubbell, 1997; 1998). According to niche-assembly theory, species co-exist in interactive equilibrium with other species in the community and each of which has evolved to be best competitor in its own ecological niche. The second perspective, dispersal assembly theory holds that communities are largely accidental collection of species whose biogeographic ranges happen to overlap historical and individualistic reasons. In this theory, the communities are viewed as assemblages of species coming and going and locally co-existing in slowly drifting mixtures and shifting relative species abundances. Hubbell (1998) attempted to reconciliation of these two basic perspectives.

Plant biodiversity studies in tropical evergreen forests have mostly been concentrated on tree species than the other life-forms (lianas, understory plants and herbaceous vascular epiphytes etc.), ostensibly because, the diversity of trees is fundamental to total rainforest diversity, as they provide resources and habitat structure for almost all other rainforest species (Cannon et al., 1998). Moreover, being a dominant life-form, they are easy to locate precisely and to count (Condit et al., 1996), and also relatively well known taxonomically (Gentry, 1992).

Studies on tropical tree diversity have accumulated over the past decades and there is a great deal of interest to decipher the patterns and processes relating to
tropical forest diversity and in generating comprehensive review papers (Gentry, 1988a; 1990; Phillips et al., 1994; Condit et al., 2000).

Tree diversity inventories in the tropics have employed a wide range of sampling protocols that vary in tree size threshold considered for sampling, and the number, size and shape of the plots. Tree size - i.e. girth or diameter at breast height (gbh/dbh; at 1.3 m from the ground level) has been considered as a criterion for mensuration. There were studies including the enumeration of individual trees as small as 2.5 cm dbh (Knight, 1975) through 4.5 cm dbh (Bunyavejchewin, 1999), 5 cm dbh (Pelissier and Riera, 1993; Valencia et al., 1994; Johnston and Gillman, 1995), 15 cm dbh (Prance et al. 1976), 30 cm dbh (Nadkarni et al., 1995), and 91 cm gbh (Poore, 1968; Ho et al., 1987), to 152.4 cm gbh threshold (Fox, 1967). Limits of 1 cm dbh are rarely used (e.g. Bongers et al., 1988), but has been gaining a momentum in the last decades (Hubbell and Foster, 1983; 1992; Condit, 1995). The often used limits are 10 cm dbh or 30 cm gbh (see Campbell et al., 1986; 1992; Gentry, 1988a; 1988b; Lieberman et al., 1996; Phillips and Gentry, 1994; Phillips, 1996). This size class normally includes all canopy (overstory) and emergent tree species useful to forestry and they supposedly play a major role in forest structure and functioning than the lower sizes (Newbery et al., 1992). Some studies have also included lianas in the inventory (e.g. Gentry, 1988b; Lieberman et al., 1996; Makana et al., 1998; Parthasarathy, 1999). Tree height has also been considered as a criterion for tree diversity inventory. For instance, all trees of ≥1 m height were inventoried in the Lower Rio Negro, Amazonia (Rodrigues, 1961; Prance, 1979).

Floristic inventories and studies of forest dynamics usually rely on sampling plots (Dallmeier, 1992). The effects of plot size (e.g. Kilburn, 1966; Greig-Smith, 1983) and the influence of plot shape (Condit et al., 1996; Laurance et al., 1998) on
the estimates of plant diversity have been assessed; at least the former in detail, while the latter less extensively, especially in the tropics. Plot-less methods have also been employed for tree diversity inventory. For example, Balslev et al. (1987) established two ~ 1 hectare plots, one each in terra firme and varzea forests at Anangu, Ecuador, and enumerated all trees ≥10 cm dbh, employing point-centered-quarter method.

Most studies have followed the plot method, including square plots (e.g. 100 m × 100 m, Campbell et al., 1986; Gentry, 1990) to rectangular plots (e.g., 80 m × 125 m; Prance, 1990), to long belt transects (e.g. 10 m × 1000 m; Boom, 1986). Plot-based research occurs within a range of plot sizes from 0.1 ha plots (e.g., Gentry, 1988a), to 1 ha plots (e.g. Black et al., 1950; Uhl and Murphy, 1981), 50 ha plot (in Barro Colorado Island, Panama [Hubbell and Foster, 1983]) and up to 52 ha plot in the Lambir National Park, Malaysia (Condit et al., 2000). One-hectare plots have been widely used in tropical forests. In the recent years the methodological emphasis in the study of tropical forests has shifted to large-scale permanent forest plots. The rationale is to provide sufficiently precise estimates of diversity, density, dispersion pattern, mortality, recruitment, growth and net rates of change in structure and populations (Hall et al., 1998). The 50 ha plot at Barro Colorado Island, Panama, established in 1982, was the first of the ‘mega-plot’ (Hubbell and Foster, 1983). Since then, eleven large-scale (≥16 ha) permanent inventory plots have been established in each of the major tropical forest formations of the world (Condit, 1995): eight plots in evergreen forests (Pasoh Forest Reserve and Lambir Hills National Park, Malaysia; Sinharaja World Heritage site, Sri Lanka; Palanan Wilderness Area, Philippines; Ituri forest, Zaire; Korup National Park, Cameroon; Yasuni National Park, Ecuador and Huai Kha Khaeng Wildlife sanctuary, Thailand), two plots in seasonal moist forests
(Barro Colorado Island, Panama and Luquillo experimental forest, Puerto Rico) and one in deciduous forest (Mudumalai Game reserve, India).

Table 1 summarizes literature on quantitative ecological inventory of trees within the sample plots of mostly ≥1 ha in the forests of tropical America, Africa, Asia and Oceania. Tree species and family richness, stem density, and basal area varied considerably across the tropics. By and large, the American tropical forests have been extensively studied than the Asian and African forests. It all began with the pioneering study of Davis and Richards (1934). They inventoried all trees ≥10 cm dbh in five 1.5 ha plot of tropical evergreen forest at Moraballi Creek, Guyana. The floristic inventory of trees in the tropics has increased substantially after its status was reviewed by Prance (1977), especially on Amazon forest composition and structure. Gentry (1988b) opined that the highest alpha diversity of trees in the world occurred in upper Amazonia. He recorded 275-283 tree species ha⁻¹ for trees ≥10 cm dbh at Yanomono and Mishana near Iquitos, Peru. Inventories in upper Amazonian Ecuador (Balslev et al., 1987; Korning et al., 1991; Valencia et al., 1994) corroborate Gentry’s opinion. Valencia et al. (1994) encountered a strikingly high tree species richness of 473 species ha⁻¹ for trees ≥5 cm dbh, and 307 species ha⁻¹ for trees ≥10 cm dbh. This inventory formed the world’s highest record of tree species richness on a hectare basis for stems ≥10 cm dbh.

Although, tree diversity inventory was initiated long back in African tropical forests (Richards, 1939), only a handful of literature (at ≥1 ha scale inventories) are available to date from these forests (Table 1). Some of the available studies within the African forests, which sampled less than 1 ha area, include Gartlan et al. (1986), Newbery et al. (1986a), Taylor et al. (1996) and Geldenhuys (1998). Recently four
Table 1. Locations, forest types, sampling methods and summary of results of tree diversity inventory in the tropical forests. nc-non-contiguous plots; ns-plot dimension not stated; d-diameter; c-circumference (girth at breast height); OP-on plot; R-range per hectare; m-mean per hectare; †- lianas also included in the study.

<table>
<thead>
<tr>
<th>Location</th>
<th>Sample size (ha)</th>
<th>Dimension (m)</th>
<th>Minimum d/gh (cm)</th>
<th>Number of Trees</th>
<th>Number of Species</th>
<th>Number of Family</th>
<th>Basal area (m² ha⁻¹)</th>
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<td>122×122</td>
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<td>-</td>
<td>-</td>
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<td>28</td>
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<td>31</td>
<td>-</td>
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<td>Pires et al., 1953</td>
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<td>36</td>
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<td>Species</td>
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<td>Basal area (m² ha⁻¹)</td>
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<td>186&lt;sup&gt;Op&lt;/sup&gt;</td>
<td>-</td>
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<td>(4.4, 4 &amp; 4)</td>
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<td>446&lt;sup&gt;m&lt;/sup&gt;</td>
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<td>265&lt;sup&gt;Op&lt;/sup&gt;</td>
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<td>-</td>
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</tr>
<tr>
<td>Location</td>
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<td>Dimension (m)</td>
<td>Minimum d/gh (cm)</td>
<td>Number of Trees</td>
<td>Species</td>
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<td>Basal area (m² ha⁻¹)</td>
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<td>Spichiger et al., 1989; 1990</td>
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<td>Rio Jurua, Acre</td>
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<td>16-37 R</td>
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<td>1⁻¹</td>
<td>2,35,895 Op</td>
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<td>Thompson et al., 1992</td>
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<td>37.94</td>
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**African tropical forests**

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**Indian tropical forests**

**Evergreen forests, Western Ghats**

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<th>Minimum d/gh(cm)</th>
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<th>Number of Species</th>
<th>Number of Family</th>
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16
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<td>64-82 R</td>
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<td>38-50 R</td>
<td>74 m</td>
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<td>38-50 R</td>
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<td>Semi-evergreen forest, Shervarayan hills</td>
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<td>Two 100×100 &amp; one 200×100</td>
<td>30 c</td>
<td>3,260 Op</td>
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<td>44 Op</td>
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<td>640-986 R</td>
<td>33-50 R</td>
<td>26-35 R</td>
<td>21.6-44.3 R</td>
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<td>89 Op</td>
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<td>Minimum d/gbh (cm)</td>
<td>Number of Trees</td>
<td>Species</td>
<td>Family</td>
<td>Basal area (m² ha⁻¹)</td>
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<td>Four 100×200</td>
<td>30^c</td>
<td>3825^Op</td>
<td>78^Op</td>
<td>36^Op</td>
<td>43.6</td>
<td>Chittibabu &amp; Parthasarathy, 2000</td>
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<td>478^m</td>
<td>45^m</td>
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<td>Williams et al., 1969</td>
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<td>1533^Op</td>
<td>309^Op</td>
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<td>49.54^m</td>
<td>Jaffre &amp; Veillon, 1990</td>
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<td>140^m</td>
<td>39^m</td>
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</table>
plots of 10 ha each were established in the two forest types (two each in monodominant and mixed forests) of the Ituri forest in northeastern Democratic Republic of Congo (Makana et al., 1998). They reported only the results of 3 ha analysis from a 10 ha block in each forest type (Table 1). In their study, all lianas \( \geq 2 \text{ cm dbh} \) were also included in the inventory.

Quantitative ecological inventory of Asian tropical forests was initiated in the second part of the 20\(^{th}\) century (e.g., Ashton, 1964; Nicholson, 1965; Wyatt-Smith, 1966 etc.). Several studies have been conducted in the tropical lowland forests of South-east Asia (Table 1). Newbery et al. (1992) opined that forest inventories of this region often use many small plots, and invariably group species, especially the non-commercial ones, approximately to family or genus level only, so that the data are of limited ecological use. Notably, the Dipterocarpaceae have received the most detailed and accurate attention in all the enumerations. Many other families of considerable ecological, but little commercial interest (Euphorbiaceae) are poorly known. Of the total eight ‘mega-plots’ established and maintained in the tropical evergreen forests of the world, four are located in Asia (Kochummen et al., 1990; Condit et al., 2000) by the initiative of the Centre for Tropical Forest Science network, which include two plots in Malaysia (Pasoh Forest Reserve and Lambir National Park) and one plot each in Sri Lanka (Sinharaja Biosphere Reserve) and Thailand (Huai Kha Khaeng).

Similar studies on quantitative ecological inventory of trees in the Indian tropical forests are limited (Table 1). It was initiated by Rai (1981), who inventoried all trees \( \geq 10 \text{ cm dbh} \) in four plots of 2.7, 2.7, 2.63 and 1.09 hectares respectively at Devimane, Malemane, Kodkani and Katilekan areas of the Western Ghats. Most studies in the Indian evergreen forests have been conducted during the last decade of
the 20th century. Contemporaneous studies have been conducted in the semi-evergreen (Kadavul and Parthasarathy, 1999a; b) and evergreen (Chittibabu and Parthasarathy, 2000) forests of the Eastern Ghats, and in a deciduous forest (Sukumar et al., 1992) and evergreen and moist deciduous forests of the Western Ghats (Bhat et al., 2000) and in the dry evergreen forests on the Coromandel coast (Parthasarathy and Karthikeyan, 1997a; Parthasarathy and Sethi, 1997). Ghate et al. (1998) studied different vegetation types (from evergreen forest to scrub) of the Western Ghats totaling a 75 ha area, along a latitudinal belt from 8° N 21° N.

A few studies have been conducted in the Oceania region on tree diversity (Williams et al., 1969; Jaffre and Veillon, 1990; Wright et al., 1997; Oatham and Beehler, 1998; Strasberg, 1996).

Species-area and species-individual relationships

Species-area and species-individual curves have been central to community ecology for decades (Fisher et al., 1943; Preston, 1948; 1962a; b; MacArthur and Wilson, 1967; Condit et al., 1996). The observation that the species number tends to increase continuously and monotonically with area was first published in the work of Watson (1835) and later it was reiterated. The species-area curve has been cited as one of the few 'laws' of community ecology (Schoener, 1976; Gould, 1979; McGuinness, 1984a; b). In the 20th century the emphasis shifted from observing the relationship to expressing its form mathematically (Arrhenius, 1921; Preston, 1960; 1962a; b; Gleason, 1922; 1925). Bunge and Fitzpatrick (1993), Colwell and Coddington (1994) and Chazdon et al. (1998) provided a broad overview of statistical approaches for estimating species richness from samples.
Most models of community structure based on habitat partitioning suggest that there will be an asymptote in the species-accumulation curve. Species-area curves have been constructed for several forests, and the species numbers continue to rise over several hectares (Whitmore, 1990; Richards, 1996). Condit et al. (1996) analysed the species-area curves raised for three size classes of trees ≥1 cm, ≥10 cm, ≥30 cm dbh in the three 50 ha plots, established one each in Pasoh (of Malaysia), Barro Colorado Island (Panama) and Mudumalai (India). They observed that species continued to accumulate in all the three inventories up to and beyond 50 ha. This contradicts a widely held belief among the tropical ecologists that tree species richness reaches an asymptote at 1-3 ha (Boom, 1986; Gentry, 1988b; Tuomisto et al., 1995).

Species diversity and abundance

One of the key goals of ecology is to explain the distribution and abundance of species (Harte et al., 1999; Kunnin et al., 2000). Diversity of a community can be assessed by the proportional species abundance data either by using statistical sampling theory (Fisher et al., 1943; Preston, 1948; 1962a) or by a variety of nonparametric measures (Simpson, Shannon etc.). Due to the complex nature and lack of theoretical justification for statistical sampling theory, the nonparametric measures have gained a great deal of popularity in the recent past (Magurran, 1988; Krebs, 1989).

If the relative abundance of species in a particular plant or animal group in a given community is somehow measured, there will be some common species, and some rare species and many species of varying degree of rareness (May, 1975). The concept of dominance, that is, the idea that certain species so pervade the ecosystem that they exert a powerful control on the occurrence of other species, is one of the oldest concepts in ecology. McNaughton and Wolf (1970) opined that dominance is
an expression of ecological inequalities arising out of different exploitation strategies. Possible mechanisms which determine and maintain the dominance of one canopy tree species in lowland tropical forests have been reviewed, again with a focus on wet or very wet forests on well drained soils (Hart et al., 1989; Connell and Lowman, 1989; Hart, 1990) and tolerance of soil fertility (Gentry, 1988a), type of mycorrhizal association (Grime et al., 1987; Janos, 1987), escape from predators (Janzen, 1974; 1984), and succession (Connell, 1978; Hart, 1990).

A consequence of high species diversity that characterizes tropical forests is the low density of most tree species and the large expected distances between the conspecific trees. This phenomenon was noted long back (Wallace, 1878). Rarity is a general characteristic of tropical plant communities (e.g. Poore, 1968; Paijmans, 1970; Thorton et al., 1982; Hubbell and Foster, 1986; Ho et al., 1987; Bongers et al., 1988; Valencia et al., 1994; Pitman et al., 2001). 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; Hubbell and Foster, 1986). Poore (1968) found that at least some of the rare species were determined by soil differences. One of the major sources of complexity in the analysis of rarity is the “sample” or “scale problem” (Soule, 1986).

Population structure

Ecologists often use size distribution to indicate the health of a population. In the absence of direct estimates of population size through time, this seems a reasonable short-cut. Population structure data for tree species (age or size categories vs. number of individuals; vertical structure sensu Lamprecht, 1962; Finol, 1971) are necessary for a meaningful interpretation of forest succession. If there are large
number of juveniles relative to the adults, this is taken to indicate that a population is stable, perhaps growing, but few juveniles can be seen as a warning that the population is in decline. One main goal of forest ecology is to infer past changes and predict future changes in species composition, and examining the size class distributions of canopy species is often used towards this end (Lorimer, 1980; Knowles and Grant, 1983; Ogden, 1985; Hart et al., 1989; Franklin et al., 1993; Read et al., 1995). When canopy species are well represented among saplings, the forest is often considered to have reached a climax state. Species in the process of being eliminated from the forest—perhaps during succession (fail to reproduce) and thus lack sufficient advanced regeneration. The lack of congruence between overstory and understory in these stands is both an indicator of change as well as a major concern for conservation (Foster et al., 1996). In trees, the discrete stages in a life table are typically stem diameter classes (Usher, 1966; Enright and Ogden, 1979; Platt et al., 1998; Newbery and Gartlan, 1996; Condit et al., 1998). In most studies the same size-class limits are used for small sized species as for large-sized ones (e.g. Knight, 1975; Kadavul and Parthasarathy, 1999a;b). This results in a differentiation between large and small structure species rather than between species with different population structure.

Bongers et al. (1988) argued that for the description of the size structure of populations of different species, it is not very accurate to use uniform size classes for the construction of frequency diagrams. They obtained a separate diagram for each species, taking into account the population size and the maximum size of the trees within the plot and distinguished three kinds of population structure, differentiated by recruitment modes, viz. continuously high, discontinuously high, and continuously low recruitment. Condit et al. (1998) analysed the population structure of 216 tree
species and concluded that static information on the size distribution is not a good predictor of future population trends while demographic information is.

Spatial distribution

The spatial distribution of individuals in a species is central in ecology theory (Dale, 1999). Plant populations exhibit three patterns of spatial distribution - clumped, or uniform or random. Patchiness, or the degree to which individuals are aggregated or dispersed, is crucial to how species uses resources, to how it is used as a resource, and to its reproductive biology. Spatial patterns have been a particularly important theme in tropical ecology, because high diversity in the tropics begets low densities. Since Wallace (1835) noted how difficult it was to find individuals of the same species, the hyperdispersion of tropical trees has focussed much of theoretical tropical ecology. Hubbell (1979) published a large study of dispersion of trees in a dry forest in Costa Rica. His results were contrary to Wallace’s long-prevailing wisdom and the Janzen-Connell prediction (Janzen, 1970; Connell, 1971) that wide dispersion is a defence against predators. Most species are aggregated, so that near neighborhoods of a tree had higher than average density of conspecifics. Since that study, though contradictory results appeared, particularly from Lieberman and Lieberman (1994), who found that most species in a wet forest in Costa Rica were not aggregated. Various other studies have reported a similar trend (e.g. Pires et al., 1953; Ashton, 1964; 1969; Poore, 1968; Lang et al., 1971; Greig-Smith, 1979; Lieberman, 1979; Thortonget al., 1982; Newbery et al., 1986b; Condit, 1995).

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 distributions. 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 (Armesto et al., 1986).

**Forest dynamics: tree population changes over time and space**

Understanding forest dynamics is fundamental to several aspects of rainforest ecology, including successful management of tropical forests for human uses. Due to the lack of annual rings in many tropical tree species, the study of tropical forest dynamics has relied heavily on the use of permanent plot data (Bunyavejchewin, 1999). One of the primary purposes of permanent plots is to monitor forest diversity and processes over time. Potentially, permanent tree plot could be employed as part of an early-warning system to detect the possible effects of environmental change on forests (Risser et al., 1993). Since the mid-20th century, a substantial body of data has been gathered on the rate of tree mortality and recruitment ("turnover") in humid tropical forests (Phillips and Gentry, 1994).

Brokaw (1985a), Martinez-Ramos (1985) and Denslow (1987) emphasized the importance of canopy openings (gaps) in the structural and compositional dynamics of many tropical forests. Research on tropical forest dynamics has focused on characteristics of gaps, including definition of boundary, frequency of creation, size of canopy openings, species dependence, internal heterogeneity, physiologic requirements, species packing and equilibrium or non-equilibrium status, among others (e.g. Hartshorn, 1978; 1980; Denslow, 1980; Brokaw, 1982; 1985b; Orians, 1982; Augspurger, 1984; Bazzaz, 1984; Hubbell and Foster, 1983; 1986; 1987; Brandani et al., 1988).

Interest in tree mortality and forest dynamics has increased recently because forest dynamics is thought to be involved in determining tree species diversity
(Phillips et al., 1994), and also thought to be related to global climate change, in particular (Phillips and Gentry, 1994). Phillips and Gentry (l.c.) concluded that tree turnover rates have increased in tropical forests during the latter part of the 20th century. This has proved to be a controversial finding (e.g. Phillips, 1995; Sheil, 1995), but one which could have important implications for biodiversity and atmospheric change.

Variation in forest dynamics can already be found on a small spatial scale and be related to the differences in soil richness (e.g., van Schaik and Minmanto, 1985). Burslem and Whitmore (1999) reported an annual mortality rate of 1.14% to 2.32% and recruitment rate of 1.12% to 7.70% in the evergreen forest of Kolombangara, Solomon Islands. On a global scale, Phillips et al. (1994) correlated a whole range of variables (e.g., soil quality, rainfall, rainfall seasonality) to forest turnover and found that forest turnover explained most of the variation in tree species. In general, forests with higher rates of tree turnover have a higher number of tree species.

Table 2 summarizes the results of tree dynamics monitored in various tropical forests of America, Asia, Africa and Oceania. Recensus period of these plots varied between 2 years (in Venezuela; Veillon, 1985) and 38 years (Malaysia; Manokaran and Kochummen, 1987), and the plot area ranged from 0.36 ha (Mexico; Pinero et al., 1984) to 50 ha (BCI, Panama; Condit et al., 1995). The reported values of annual tree mortality varied between ~1 to 2% and rarely 3% (Holta, 1989; Hubbell and Foster, 1990; Korning and Balslev, 1994; Felfili et al., 1995). Similarly, the annual recruitment varied between 0.23% (Pascal et al., 1998) and 3.36% (Holta, 1989).
Table 2. Summarized results of tree dynamics data in the permanent plot of various tropical forests in America, Asia, Africa and Oceania. NA- denotes Not Applicable.

<table>
<thead>
<tr>
<th>Location</th>
<th>Area (ha)</th>
<th>gdbh (cm)</th>
<th>Inventory period (interval years)</th>
<th>Trees ha(^{-1})</th>
<th>Mortality (%/year)</th>
<th>Recruitment (%/year)</th>
<th>Source</th>
</tr>
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<td><strong>America</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
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<tr>
<td>Belem, Brazil</td>
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<td>1956-71 (15)</td>
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<td>1.84</td>
<td>0.81</td>
<td>Pires &amp; Prance, 1977</td>
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<tr>
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<td>2</td>
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<td>1975-80 (5)</td>
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<td>1.06</td>
<td>NA</td>
<td>Putz &amp; Milton, 1982</td>
</tr>
<tr>
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<td>10(^d)</td>
<td>1968-78 (10)</td>
<td>512</td>
<td>1.04</td>
<td>NA</td>
<td>Lang &amp; Knight, 1983</td>
</tr>
<tr>
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<td>10(^d)</td>
<td>1975-83 (7)</td>
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<td>1.06</td>
<td>NA</td>
<td>Pinero et al., 1984</td>
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<td>0.25</td>
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<td>1968-80 (11.9)</td>
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<td>0.97</td>
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27
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<tr>
<th>Location</th>
<th>Area (ha)</th>
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<th>Trees ha(^{-1})</th>
<th>Mortality (%/year)</th>
<th>Recruitment (%/year)</th>
<th>Source</th>
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<td>Lieberman et al., 1985; 1990</td>
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<td>Gentry &amp; Terborgh, 1990</td>
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<td>1.50</td>
<td>Jardim, 1990</td>
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<td>Mortality (%/year)</td>
<td>Recruitment (%/year)</td>
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</table>

**Asia**

<p>| Separilok, Sabah           | 1.81      | 10$^d$     | 1956-68 (12)                   | 667             | 1.11                | 1.42                 | Nicholson, 1965     |
| Bukit Lagong, Malaysia     | 2         | 10$^d$     | 1947-85 (36)                   | 515             | 1.40                | 1.15                 | Wyatt-Smith, 1966   |
| Separilok, Sabah           | 1         | 10$^d$     | 1957-66 (9.2)                  | 435             | 1.92                | 1.53                 | Fox, 1973           |</p>
<table>
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<tr>
<th>Location</th>
<th>Area (ha)</th>
<th>g/dbh (cm)</th>
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<th>Mortality (%/year)</th>
<th>Recruitment (%/year)</th>
<th>Source</th>
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<td>Location</td>
<td>Area (ha)</td>
<td>g/dbh (cm)</td>
<td>Inventory period (interval years)</td>
<td>Trees ha⁻¹</td>
<td>Mortality (%/year)</td>
<td>Recruitment (%/year)</td>
<td>Source</td>
</tr>
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