CARBON SEQUESTRATION IN THE HUMID TROPICAL AND SUBTROPICAL FORESTS OF MEGHALAYA

ABSTRACT

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ABSTRACT

Data on carbon sequestration potential for different forest types in tropical and montane sub-tropical regions of north-eastern India are non-existent. Carbon data for different compartments are essential to understand the carbon sequestration potential of different forest types. In order to bridge this data gap, the present study was undertaken to assess the carbon sequestration potential in soil and vegetation in nine different forest types of Meghalaya, of which five were in tropical landscape and four were in montane sub-tropical landscape. The five tropical forest types were: (i) old-growth broad-leaved forest (TOBF), (ii) regenerating broad-leaved forest (TRBF), (iii) teak (*Tectona grandis*) plantation forest (TTPF), (iv) sal plantation forest (TSPF), and (v) mixed bamboo forest (TMBF). All these forests were located in Nongkhyllem wildlife sanctuary and Nongkhyllem reserve forest. The four montane sub-tropical forest types included were: (i) old-growth broad-leaved forest (SOBF), (ii) old-growth pine (*Pinus kesiya*) forest (SOPF), (iii) regenerating pine (*Pinus kesiya*) forest (SRPF), and (iv) regenerating broad-leaved forest (SRBF). All these forest types were located in Upper Shillong area. The elevation range of the tropical and montane sub-tropical forest types ranged from 205 - 297 and 1672 – 1920 m. a.s.l., respectively. All the forest types receive very high rainfall and rainy season commences with the onset of southwest monsoon in May and continues up to September. Three fourth of the total annual rainfall is received during this season.

Carbon was estimated in the above mentioned forest types in tropical and subtropical landscapes in three important pools viz., vegetation biomass carbon pool, soil organic carbon pool, and microbial biomass carbon pool. Four important fluxes of carbon viz., Net primary production (NPP), litter fall, litter decomposition, and soil respiration were also estimated.

Community characteristics, tree population structure and soil physic-chemical properties of each forest type were analyzed to relate and explain the observed patterns of the carbon pools and fluxes.

Soil organic carbon (SOC) was estimated colorimetrically. Carbon percentage value thus obtained was converted into Mg C ha^{-1} value by using the formula: SOC (Mg C ha^{-1}) = %C x bulk density x soil depth. Microbial biomass carbon values were obtained following chloroform fumigation method. The plant carbon content was estimated by igniting the samples at 550°C for 6 hours in muffle furnace and carbon content was calculated as 50% of ash free mass. The biomass carbon for different forest types was obtained by multiplying biomass values with the carbon percentage obtained. Tree above ground and below ground biomass was estimated by developing/adopting appropriate allometric models. Litter flux was estimated at seasonal interval over a period of three years by using litter traps of 1m x 1 m x 0.15 m size, randomly placed in each permanent plot. Leaf litter decomposition in each forest type was studied using litterbag technique. Soil respiration (CO_2 efflux) rate was determined following alkali absorption method. Net primary productivity was calculated following 'net positive increment in biomass' approach and was derived from the data collected over a period of four years.

Vegetation characteristics of the tropical and montane sub-tropical landscape

The dominant tree species in the tropical landscape were *Schima wallichii*, *Actinodaphne obovata*, *Artocarpus chaplasha*, *Tectona grandis* and *Shorea robusta* depending on the forest type. *Dendrocalamus hamiltonii* and *Teinostachyum dullooa* were the two dominant bamboo species in the tropical mixed bamboo forest. The dominant tree species in the montane sub-tropical landscape were *Lindera latifolia*, *Pinus kesiya* and *Rhododendron arboretum*.

Tree density in the tropical landscape was maximum in the regenerating broad-leaved forest (1728 individuals ha^{-1}) and minimum in the mixed bamboo forest (794 individuals ha^{-1}). *Dendrocalamus hamiltonii* had a density of 3420 culms ha^{-1} whereas *Teinostachyum dullooa* had 1524 culm ha^{-1}. In the montane sub-tropical landscape, tree density was maximum in the regenerating broad-leaved forest (2106 individuals ha^{-1}) and minimum in the old-growth pine forest (628 individuals ha^{-1}). About 91.2 – 94.5% tree species in the tropical landscape exhibited clumped distribution pattern, and the rest 5.5 – 8.8% exhibited random distribution pattern. No tree species exhibited regular dispersion pattern in tropical landscape. The figures for clumped, random and regular distribution patterns in montane sub-tropical landscape were 48.6 – 83.3%, 5.7 – 33.3% and 11.1 – 45.7%, respectively.

Species richness in the tropical landscape was highest in the old-growth broad-leaved forest with 94 species and lowest in teak plantation forest with 33 species. In the montane sub-tropical landscape, the regenerating broad-leaved forest had higher species richness with 35 species while the regenerating pine forest had lowest species richness with 6 species.

Physico-chemical properties of soil and soil organic carbon

The soils of both the landscapes were sandy in texture. The clay content in the surface and sub-surface soil layers were higher in the tropical landscape than the montane sub-tropical landscape. The clay percentage ranged between 9.2 – 13.3% in the surface layer and 11.2 – 17.3% in the sub-surface layer in the tropical landscape. In the montane sub-tropical landscape, the clay percentage ranged from 7.2 – 9.4% in the surface and 9.2 – 11.4% in the sub-surface layers. Water holding capacity (WHC) was higher in the surface soil layer than the sub-surface layer in both the landscapes. Since the surface layer had greater accumulation of organic matter, it is argued that SOC influenced WHC more than the clay particles. A strong negative correlation between bulk density and SOC was observed in the tropical ($R^2 = 0.84$; $p = 0.002$) and montane sub-tropical forest types ($R^2 = 0.96$; $p = 0.003$). The soil moisture content (SMC) ranged from 15.3 – 21.4% in the tropical landscape, and 24.6 – 36.4% in the montane sub-tropical landscape. SMC was lower in the surface layer of all the tropical forest types than in the surface layer of all the montane sub-tropical landscape. The soils of both the tropical and montane sub-tropical forest types were acidic in reaction (pH = 4.2 - 6.3 and 4.2 – 5.9, respectively).

Total kjeldahl nitrogen (TKN) was high in the montane sub-tropical landscape (0.3 - 0.5%), and it was low in the tropical landscape (0.18 – 0.22%). SOC was positively correlated with TKN only in the montane sub-tropical landscape ($R^2 = 0.834$; $p < 0.01$). Available phosphorus (45.5 $\mu g\ g^{-1}$) was relatively high in the regenerating broad-leaved forest of montane sub-tropical landscape and sal plantation forest (19.9 $\mu g\ g^{-1}$) of tropical landscape, and low in the teak plantation forest (2.3 $\mu g\ g^{-1}$) of tropical landscape and old-growth broad-leaved forest (2.6 $\mu g\ g^{-1}$) of montane sub-tropical landscape. Available phosphorus did not show any significant correlation with soil organic carbon in the tropical landscape, but showed positive correlation in the montane sub-tropical landscape (R^2=0.91; $p<0.0001$). The exchangeable K was highest in the sal plantation forest (454.0 $\mu g\ g^{-1}$) and lowest in the teak plantation forest (117.5 $\mu g\ g^{-1}$) of tropical landscape. In the montane sub-tropical landscape, it was higher in the old-growth broad-leaved forest (239.5 $\mu g\ g^{-1}$) and lower in the old-growth pine forest (74 $\mu g\ g^{-1}$). Exchangeable Potassium showed a positive correlation with SOC in the tropical ($R^2 = 0.751$; $p < 0.01$) and montane sub-tropical ($R^2 = 0.415$; $p < 0.05$) landscapes.

Soil carbon pool in tropical and montane sub-tropical forest ecosystems

The soil organic carbon (SOC) pool in the tropical landscape upto 1 m depth was largest in the old-growth broad-leaved forest (83.2 Mg C ha^{-1}), followed by mixed bamboo forest (81.2 Mg C ha^{-1}), regenerating broad-leaved forest (55.9 Mg C ha^{-1}), sal plantation forest (48.2 Mg C ha^{-1}) and teak plantation forest (41.7 Mg C ha^{-1}). In the montane sub-tropical landscape, SOC pool was largest in the regenerating broad-leaved forest (102.4 Mg C ha^{-1}), followed by old-growth broad-leaved forest (81.8 Mg C ha^{-1}), old-growth pine forest (58.7 Mg C ha^{-1}) and it was lowest in the regenerating pine forest (35.2 Mg C ha^{-1}). The SOC pool was highest during summer and lowest during winter season in all the forest types of both the landscapes. SOC declined with depth ($p<0.01$). The SOC showed a negative correlation with bulk density in all the forest types of both the landscapes.

The soil C/N ratio increased with depth in all the tropical and montane sub-tropical forest types, except for the sal plantation forest where it showed a reverse trend due to less SOC in the sub-surface layer. The C/N ratios in the tropical and montane sub-tropical landscapes ranged from 7.1 – 10.3 and 4.8 – 10.6, respectively.

Microbial biomass and soil organic carbon

The microbial biomass carbon (MBC) pool in the surface and sub-surface soil layers of tropical landscape was highest in the sal plantation forest (420.4 & 293.9 μg g^{-1}, respectively) and lowest in the regenerating broad-leaved forest (278.4 & 169.4 μg g^{-1}, respectively). The corresponding values in the montane sub-tropical landscape were highest in the regenerating broad-leaved forest (359.7 & 240.7 μg g^{-1}, respectively), and lowest in the regenerating pine forest (189.9 & 123.5 μg g^{-1}, respectively). The contribution of MBC to the soil organic carbon in the surface soil layer of tropical landscape was 1.2% - 4.9%, and the proportion in the montane sub-tropical landscape was 0.9 - 2.0%.

The MBC showed a strong positive correlation with SMC in the tropical (R^2=0.9, p<0.001) and montane sub-tropical (R^2=0.68, p<0.005) landscape. A strong positive correlation was also observed between SOC and MBC in the tropical (R^2=0.9, p<0.001) and montane sub-tropical (R^2=0.91, p<0.001) landscape.

Carbon flux through litterfall in tropical and montane sub-tropical landscapes

The litter production in the tropical and montane sub-tropical forest types ranged from 6.5 – 10.8 Mg ha^{-1} yr^{-1} and 4.2 – 10.1 Mg $ha^{-1}yr^{-1}$, respectively. High litter production was observed in sal plantation forest (10.8 Mg ha^{-1} yr^{-1}) of tropical landscape and old-growth broad-leaved forest (10.1 Mg $ha^{-1}yr^{-1}$) of montane sub-tropical landscape. Similarly, the litter carbon pool in the tropical and montane sub-tropical forest types ranged from 3.3 – 5.1 Mg C ha^{-1} yr^{-1} and 2.1 -4.6 Mg C ha^{-1} yr^{-1}, respectively. The litter accumulation in the tropical and montane sub-tropical landscape ranged from 2,172 – 5,865 kg ha^{-1} and 3,867 – 6,015 kg ha^{-1}, respectively. The decomposition rate was fastest in the mixed bamboo forest (k=0.06) of tropical landscape and the old-growth broad-leaved forest of montane sub-tropical landscape (k=0.3). The decomposition rate in the tropical landscape followed the trend: TMBF>TOBF>TSPF>TTPF>TRBF and the montane sub-tropical landscape followed the trend: SOBF>SRBF>SRPF>SOPF.

CO_2 efflux in the tropical and montane sub-tropical forest ecosystems

CO_2 efflux rate in the tropical landscape was higher than the montane sub-tropical landscape. CO_2 efflux rate in the tropical landscape was highest in the mixed bamboo forest (1.7 mg CO_2 kg^{-1} day^{-1}) and followed the order: TMBF>TSPF>TOBF>TTPF>TRBF. In the montane sub-tropical landscape, the value was highest in the regenerating broad-leaved forest (1.3 mg CO_2 kg^{-1} day^{-1}) and followed the order: SRBF>SOBF>SOPF>SRPF. Soil CO_2 efflux showed positive correlation with microbial biomass carbon in the tropical ($R^2 = 0.65$; $P<0.01$) and montane sub-tropical landscape ($R^2 = 0.81$; $p<0.01$) indicating a strong influence of microbial biomass on CO_2 efflux.

Ecosystem level biomass and carbon sequestration in the tropical and montane sub-tropical forest ecosystems

Total aboveground biomass (TAGB) in the tropical forest types was highest in the sal plantation forest (472 Mg ha^{-1}) and followed the trend: TSPF>TMBF>TOBF>TTPF>TRBF. The TAGB in the montane sub-tropical forest types was highest in the old-growth pine forest (425.9 Mg ha^{-1}) and followed the trend: SOPF>SOBF>SRPF>SRBF. The contribution of belowground biomass (BGB) to the total ecosystem biomass in the tropical landscape ranged from 11.0 – 16.0%, with highest being in the regenerating broad-leaved forest (16.0%). The corresponding figures in the montane sub-tropical landscape ranged from 8.7 – 43.4% with highest contribution in the regenerating broad-leaved forest (43.4%). The herb and shrub biomass contributed a negligible amount to the total ecosystem biomass in both the tropical and montane sub-tropical landscapes. The proportion of AGB in the higher diameter classes i.e. >60 cm in the tropical landscape was high in the mixed bamboo forest (52%) followed by old-growth broad-leaved (43%) and sal plantation forest (32%). The proportion of AGB in the higher diameter classes i.e. >60 cm in the sub-tropical landscape was only 16% and it was only in the old-growth pine forests.

The ecosystem level biomass in the tropical landscape was highest in the sal plantation forest (545.2 Mg ha^{-1}) and followed the trend: TSPF>TMBF>TOBF>TTPF>TRBF. The highest figure in the montane sub-tropical landscape was in the old-growth pine forest (466.6 Mg ha^{-1}) and followed the trend: SOPF>SOBF>SRPF>SRBF.

The ecosystem level carbon sequestration value in the tropical landscape was higher than the montane sub-tropical landscape. In tropical landscape, it was highest in the sal plantation forest (314.4 Mg C ha^{-1}) and followed the trend: TSPF>TMBF>TOBF>TTPF>TRBF. The highest figure in the montane sub-tropical landscape was for the old-growth pine forest (286.1 Mg C ha^{-1}) and followed the trend: SOPF>SOBF>SRBF>SRPF.

Net primary productivity in the tropical and montane sub-tropical forest ecosystems

The ecosystem level NPP in the tropical landscape was highest in regenerating broad-leaved forest (18.4 Mg ha^{-1} yr^{-1}) and followed the trend: TRBF>TSPF>TOBF>TMBF> TTPF. In the montane sub-tropical landscape, the figure was highest in the old-growth pine forest (22 Mg ha^{-1} yr^{-1}) and followed the trend: SOPF>SOBF>SRBF>SRPF. The contribution of belowground NPP to the total ecosystem NPP in all the tropical and montane sub-tropical forest types were low (6.4 – 12.5%), except in the regenerating pine and regenerating broad-leaved forests of montane sub-tropical landscape where it was relatively high (28.5 and 33.9%).

Very few studies quantifying ecosystem level C are available. In the present study, the ecosystem level biomass carbon in the sal plantation forest was greater than the old-growth broad-leaved forest and mixed bamboo forest in tropical landscape. Similarly, old-growth pine forests had greater biomass C than the old-growth broad-leaved forest in montane sub-tropical landscape. The study revealed high soil carbon sequestration potential of the regenerating forests than the old-growth forests. NPP was greater in the old-growth pine forest and the regenerating broad-leaved forests than the other forest types. The future carbon sequestration potential of the forest with lower diameter classes was argued to be high, while the sal and the old-growth forests had lower carbon sequestration potential. Based on the NPP values, ecosystem level biomass C, tree diameter distribution, species composition and the age of the forest, it was concluded that the sal plantation forest and the old-growth pine forest had greater carbon stock than the other forest types, and also have sequestered more C than the others. The study concludes that large variation in carbon content exists among different forest types. Overall, the carbon sequestration potential of natural forests of northeastern India is high, even with a past disturbance history. The forests of both the landscapes are yet to be fully matured and therefore, have the potential to store additional carbon in the future and substantially contribute to the reduction of atmospheric CO_2.

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I, Ratul Baishya, hereby declare that the subject matter of this thesis entitled "***Carbon sequestration in the humid tropical and sub-tropical forests of Meghalaya***" is the record of the work done by me, that the contents of the thesis did not form basis of the award of any previous degree to me or to the best of my knowledge to anyone else, and that the thesis has not been submitted by me for research degree in any other University / Institute.

This work is being submitted to the North-Eastern Hill University for the award of the degree of Doctor of Philosophy in Botany.

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Abbreviations

AGB = aboveground biomass
AGBC = aboveground biomass carbon
ANPP = aboveground net primary productivity
BD = bulk density
BGB = belowground biomass
BGBC = belowground biomass carbon
BNPP = belowground net primary productivity
C = carbon
CO_2 = carbon dioxide
DBH or D = diameter at breast height
ha = hectare
IVI = Important value index
K = exchangeable potassium
m = meter
MBC = microbial biomass carbon
MBF = matured broadleaved forest
MBN = microbial biomass nitrogen
Mg = mega gram (106 g)
MSE = mean square error
NPP = net primary production
P = available phosphorus
R^2 = coefficient of determination
RMSE = root mean square error
SE = standard error
SMC = soil moisture content
SOBF = sub-tropical old-growth broad-leaved forest
SOC = soil organic carbon
SOM = soil organic matter
SOPF = sub-tropical old-growth pine forest
SRBF = sub-tropical regenerating broad-leaved forest
SRPF = sub-tropical regenerating pine forest
SSE = sum of square error
TAGB = total aboveground biomass
TAGBC = total aboveground biomass carbon
TBGB = total belowground biomass
TBGBC = total belowground biomass carbon
TKN = total Kjeldahl nitrogen
TMBF = tropical mixed bamboo forest
TOBF = tropical old-growth broad-leaved forest
TRBF = tropical regenerating broad-leaved forest
TSPF = tropical sal plantation forest
TTPF = tropical teak plantation forest
WHC = water holding capacity

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CHAPTER I
INTRODUCTION

Carbon sequestration is defined as the net removal of CO_2 from the atmosphere into long-lived pools of carbon. The pools can be living aboveground biomass such as trees or such products with a long life created from biomass as timber, and living biomass in soils, for example, roots and microorganisms or recalcitrant organic and inorganic carbon in soils and deeper subsurface environments (CSiTE 2002).

Carbon is one of the key elements in the earth's biogeochemical cycle as it enables living organisms to exist and flourish. Besides blanketing the earth to keep it warm, it is also an elemental building block of molecules that make up all organisms on earth. Quantifying the carbon balance in forests is one of the main challenges if carbon fixation is to be considered amongst the objectives of forest management (Montero *et al.* 2005). With the introduction of the Kyoto Protocol, the subject of carbon sequestration has become stronger and important. The Protocol represents an international effort in limiting the continued release of greenhouse gases (GHGs) into the atmosphere and has an aim to reduce by 5% lower than 1990 levels by year 2012 and mitigate global warming. To expedite and achieve this, the Kyoto Protocol has attempted to motivate the world by adding an economic value to the management of carbon. However, the implementation of this protocol has met a number of challenges. One of these is the lack of an agreed upon method to be used in the quantification and monitoring of carbon sequestration. When carbon becomes an internationally tradable commodity, accurate, precise, environmentally benign and cost effective methods will inevitably be required. The world bodies met at the United Nations Copenhagen summit in Denmark from the 8th -18th Dec 2009 to find an amicable solution to the global menace of rising CO2 concentration, temperature and the consequences that comes with it.

Carbon stored in vegetation is of great interest. Carbon storage is easily modified through silvicultural practices such as rotation length, and thinning regime. The amount of carbon in the wood on the other hand affects the lifespan of wood products. Aboveground biomass is usually estimated from forest inventory data using biomass equations and expansion factors at different spatial scales (Barrio-Anta *et al.* 2006; Fang and Wang, 2001; Isaev *et al.* 1995; Schroeder *et al.* 1997). Belowground biomass is indirectly estimated from the aboveground biomass. Information for other biomass components such as litter, dead organic matter or soil carbon is less available, because these elements are more difficult to measure and in many cases are more spatially variable than other components (Isaev *et al.* 1995; Schelesinger and Andrews, 2000).

More research is required into the effects of forest management on the carbon (C) cycle so that C storage can be integrated into management strategies. In this respect, historical records are useful for analysing the effects of past management activities on C stocks and models can be developed to estimate future C stocks under different management alternatives (Balboa-Murias *et al.* 2006; Kolari *et al.* 2004). Furthermore, it is important to consider global change, since a modification in growth rates is expected for many species under forecasted changes in temperature and rainfall (Cao and Woodward, 1998; Schroeter *et al.* 2005). Most empirical growth and yield models are based on historical data under different climatic conditions than those forecasted and are not able to account for these possible climatic changes (Pretzsch *et al.* 2002).

The mitigation of global warming is faced with controversies mainly due to the fact that the atmosphere is a complex entity to deal with and that it is an example of the adage 'tragedy of the commons'. However, terrestrial carbon sequestration in aboveground woody biomass has received attention as a promising course in an immediate attempt to mitigate global warming. Although many types of carbon sequestration have been identified, this study focuses on terrestrial sequestration in aboveground woody biomass. This is so because photosynthesis is the major natural direct way by which CO_2 in the atmosphere is fixed back to earth.

In a forest ecosystem, trees recycle carbon mainly through photosynthesis and respiration and 50% of the standing biomass is carbon itself (Brown 1997; Ravindranath *et al.* 1997). Tropical forests hold large stores of carbon that plays a major role in the global carbon cycle (Houghton *et al.* 2001). The estimation of carbon in tropical forests has attracted a great deal of experimental and theoretical attention in recent years (Malhi *et al.* 1999), and several recent advances have led to a variety of estimates of carbon stocks and fluxes (Malhi *et al.* 1998). The potential of trees and forests to sequester carbon is of major concern today in relation to the continuous increase of CO_2 in the atmosphere which contributes to the general rise of the world temperature (Cannell 1999). Several research projects are being conducted to study CO_2 fluxes for different forest types around the world (Baldocchi *et al.* 1996).

The main carbon pools of a forest ecosystem are the biomass of living trees, including their dead parts, understory plants, litter and woody debris, and soil organic matter. Other pools such as soluble soil carbon or higher-order producers and decomposers represent only a negligible part. Soil carbonates, such as calcium carbonate may represent a large stock of carbon but its residence time is of several orders of magnitude longer than organic pools. The resulting carbon loss occurs mainly through leaching of soluble carbonates and does not generally result in CO_2 emission (Pignard *et al.* 2000). Recent concerns about rising levels of atmospheric CO_2 have directed attention to carbon stocks in soils of the world (Batjes 1996), and to their role as both source and sink of carbon.

Forests contain 80% of live aboveground biomass in the world with over 59% of total live biomass residing in tropical forests (Dixon *et al.* 1994). Tropical forests not only store large amounts of carbon, but over a period of 25 years they cycle a volume of carbon dioxide equal to the total amount in the atmosphere (Malhi and Grace, 2000) and account for as much as 35% of global net primary production (Melillo *et al.* 1993). Due to their tremendous capacity for storage and cycling of carbon, small changes in net carbon balance or land use change in tropical forests can result in significant storage or release of carbon dioxide to the atmosphere. The concentration of carbon dioxide has increased in the atmosphere since the Industrial Revolution (Keeling *et al.* 1996). This increase is largely the result of the burning of fossil fuels with a significant contribution (23%) from deforestation (Dixon *et al.* 1994). In the 1990s, the atmosphere received an annual average of 7.9 ± 1.2 Gt C (Prentice *et al.* 2001) from fossil fuels; however, only 3.2 ± 0.1 Gt C remained in the atmosphere (Prentice *et al.* 2001). This leaves more than half of the annual carbon dioxide emissions removed by "sinks" widely thought to be in the world's forests and oceans. There is a great deal of debate over the magnitude and location of the terrestrial sinks (Malhi *et al.* 1999). Global-atmospheric measurements indicate that temperate and boreal forests in the Northern Hemisphere absorb large amounts of carbon (Fan *et al.* 1998). However, there is a considerable uncertainty about the role of tropical ecosystems in the global carbon cycle

(Brown *et al.* 1993; Melillo *et al.* 1996) because the relationships between deforestation, undisturbed forests and the atmospheric exchange are unclear (Tian *et al.* 2000).

The role of old-growth forests in the tropics remain particularly unclear (Houghton 1991, Melillo *et al.* 1993). Are old-growth tropical forests storing carbon and, if so, what are the pools and fluxes of carbon dioxide within the forest? In the past, primary tropical forests were virtually ignored in terrestrial carbon sink estimates because these forests are predominantly old-growth forests that were assumed to be in a state of dynamic equilibrium (Salati and Vose, 1984). However in recent years, it has been suggested that old-growth neotropical forests are absorbing carbon as a result of carbon dioxide fertilization or nitrogen deposition and that the carbon sink is large enough to offset the carbon released from neotropical deforestation (Chambers *et al.* 2001b; Grace *et al.* 1995; Malhi *et al.* 1998; Phillips *et al.* 1998; Tian *et al.* 2000). The carbon sink (per unit area) in old-growth tropical forests is estimated to be somewhere between 0.4 Mg C ha^{-1} yr^{-1} (Chambers *et al.* 2001b; Tian *et al.* 2000) and 5.9 Mg C ha^{-1} yr^{-1} (Malhi *et al.* 1998) or between 0.2 Gt C yr^{-1} and 2.95 Gt C yr^{-1} if the sink is assumed to extend to all of Amazonia.

Tropical forests account for 52% of the world's forest coverage (FAO 2009) and have experienced frequent landuse change globally (Silver *et al.* 2000). Tropical forests are also critical to the global carbon cycle because about half of the world's biomass carbon is stored in these forests and 14% of the world soil carbon is located in the soils of tropical forests (IPCC 2000). Therefore, a slight change of these carbon pools could have a significant impact on global carbon cycle. Although tropical forests are important source and sinks of carbon, there is no consistent conclusion on the net carbon effect of tropical forests on carbon sequestration partially because of the complexity of the forest structures and functions, such as species composition, productivity and decomposition, and intensive management practices (Clark 2002; Harmon 2001; Schulze *et al.* 2000).

Short-rotation plantation, referred to as a Kyoto forest, has been recommended in the Kyoto Protocol as an effective measure to reduce atmospheric CO_2 concentration. Reforestation through plantation on abandoned and degraded agricultural lands in the tropics has been proposed as an effective carbon management approach (Montagnini and Porras, 1998). However, some studies have challenged the effectiveness of the Kyoto forest. For example, it was reported that the decomposition of the residuals from previous harvesting would keep the young plantation as a net carbon source for two to three decades (Schulze *et al.* 2000). Old-growth forests might transport more carbon to soils as long term carbon than a Kyoto forest (Law *et al.* 2003). Schlesinger and Lichter (2001) found that a young plantation contributed little to long-term soil carbon storage in a 25-year-old loblolly pine plantation in Duke Forest in the US. Secondary forests are extensive in the tropics and account for more than 40% of the tropical forest land and the area covered by secondary forests is continuously increasing throughout the tropical regions because of fast land-use change (Brown and Lugo, 1990; Hughes *et al.* 1999). Secondary forests provide important ecosystem services, such as erosion prevention, wildlife habitat improvement, biodiversity maintenance, water conservation and watershed protection (Feldpausch *et al.* 2004). Sustainable use of secondary forests could reduce the human pressure on primary forests and slow down the conversion of primary forests for agricultural use. A few studies (Weaver *et al.* 1987; Brown, 1998; Guo and Gifford, 2002; Hughes *et al.* 1999) on global carbon budget and land-use change showed a great potential for carbon sequestration through reforestation and afforestation of tropical agricultural and pasture lands. The success of the management of tropical forests in the future might well depend upon the adequacy of our ecological understanding of secondary forests. Additionally, accurate calculation of carbon budgets at both national and global scales for the tropical forests depends on our capacity to quantify the accumulation of carbon by secondary forests that are dominant in the tropical forests and have experienced intensive deforestation.

Although many efforts have been made to understand the ecological processes of tropical tree plantations and naturally generated secondary forests (Binkley and Resh, 1999; Dixon *et al.* 1994; Houghton *et al.* 2000; Paul *et al.* 2002), soil and forest floor carbon dynamics, such as carbon pool size, turnover rate, litterfall, decomposition, and soil carbon quality, in these forests mostly remain uncertain. This insufficient knowledge leads to current debating about whether short-rotation plantations, secondary forests or old-growth forests are more effective in sequestering atmospheric CO2 (Cox *et al.* 2000; Law *et al.* 2003; Richter *et al.* 1999; Schulze *et al.* 2000; Silver *et al.* 2000; Wirth *et al.* 2002).

The tropical forests spread over 13.76 million sq. km area worldwide accounts for 60% of the global forests (FAO 1988, 2005) and play a key role in global C cycle both in terms of C flux and the volume of C stored. The significant influence of tropical forests on carbon cycle is attributed to the high rate of primary production besides the large pool and flux sizes (Brown and Lugo, 1982, 1984). Because of higher net productivity, the tropical forests are more effective in carbon sequestration than any other forests (Brown *et al.* 1989; Soni 2003). The tropical forests act as carbon sink and store large quantities of carbon in vegetation and soil, exchange carbon with the atmosphere through photosynthesis and respiration. These forests store 37% of the total 90% of the world's terrestrial C that is stored in forests (Houghton 1996). Very few tropical forests are at their maximum potential level of biomass density because of prevailing or past cultural disturbances. As these forests are not at their maximum carbon density, the tropical forests have a larger additional carbon sequestration capacity and have the potential to increase the global carbon store beyond the present value (Iverson *et al.* 1993). Therefore, the tropical forests have attracted a great deal of experimental and theoretical attention in recent years (Malhi *et al.* 1999; Malhi and Grace, 2000).

The Amazonian tropical forests have the highest carbon sequestration as well as release among all the ecosystems, because of large area and high productivity associated with large scale deforestation, burning, and fast rate of decomposition and land use changes (Fearnside 1997). A large biogeochemical C flux (5.5±0.5Gt) due to fossil fuel combustion has been created by human activities, where none existed before. Changes in land use have resulted in a net flux of C to the atmosphere over the last 300 years, and it is estimated that in recent years, land conversion contributes 1.6±1.0 Gt of C to the atmosphere annually IPCC 1996. Whereas the flux of atmospheric C resulting from fossil fuel combustion is exclusively positive, land conversion can either release C or sequester it from the atmosphere depending on the land conversion activity. Conversion of forest to crops creates a positive flux of C to the atmosphere because forests have a considerably higher amount of C than croplands (Olson *et al.* 1983). Furthermore, forests are usually burned, causing a nearly instantaneous release of C stored in forests to the atmosphere. Forests that are harvested for timber when cleared for cropland also result in a positive atmospheric C flux, because the vast majority C stored in timber products is released to the atmosphere relatively quickly (Maclaren 1996). The tropical forests act as sources of atmospheric carbon if disturbed by anthropogenic activities or natural calamities. However, they become atmospheric carbon sinks during land abandonment, forest regrowth after disturbance and due to afforestation, reforestation and forest conservation. Therefore, the role of tropical forests as overall CO_2 sink or source is scale-dependent and site-specific. Therefore, the need for generating data at high spatial resolution for both above and below ground phytomass, soil and other pools and fluxes of C has been emphasized by several workers for improving quantification of global C pools and fluxes (Chhabra and Dadhwal, 2004).

The potential of tropical forests for increased carbon sequestration capability can be assessed either through the amount of carbon stored or estimating the annual carbon sequestration rate (Iverson *et al.* 1993). The studies on carbon sequestration have been focusing on and expressing the sequestration in terms of biomass and carbon stock. The design and evaluation of global scale carbon models require field estimates of forest biomass. Among the phytomass components i.e. aboveground, belowground and dead wood, the live aboveground wood biomass is the most important because it is involved in the regulation of atmospheric carbon concentration and constitutes about 60% of total phytomass. Therefore, estimation of aboveground biomass (AGB) is the most important aspect of studies of carbon sequestration (Ketterings *et al.* 2001). Estimation of AGB is also a useful measure for comparing structural and functional attributes of forest ecosystems across a wide range of environmental conditions (Brown *et al.* 1999).

Two key processes of forest carbon budget are net primary productivity and tree mortality, both of which change over time as stands develop. Net primary production (NPP), the difference between photosynthesis and plant respiration, is largely a function of growing conditions and age or stage of forest development. Through photosynthesis the organic matter is accumulated in standing live trees, in the production of short-lived tissues such as leaves, fruits, flowers and the below ground accumulation and production of coarse and fine roots. It is important to emphasize that increasing photosynthetic carbon fixation alone is not enough. This carbon must be fixed into long-lived pools. Otherwise, one may be simply altering the size of fluxes in the carbon cycle, not increasing carbon sequestration.

Direct harvesting techniques for estimating biomass are labour intensive and time consuming. Various methods used for estimating carbon exchange in terrestrial environment include, (i) CO_2 fix models (Mohren and Goldewijk, 1990) (ii) eddy covariance method (Grace *et al.* 1995) and (iii) biomass estimation methods (Brown 1997; Brown *et al.* 1989; Chambers *et al.* 2001a). Of these, biomass estimation method is widely used because of its relative simplicity and ease in deriving biomass values using regression models. Although it is difficult and tedious at initial stage to develop the best fit models, tree dimension values as the input data requirement for subsequent estimations has made the regression based biomass estimation method extremely popular (Brown 1997). Several regression models have been developed to estimate biomass or biomass-related parameters (Brown *et al.* 1989), which have been useful for preparing volume tables for several forestry species and estimating carbon in tropical, temperate, boreal, and semi-arid forest ecosystems (Schroeder *et al.* 1997). The total biomass data obtained from such models is then converted into carbon content for estimating carbon pools in different compartments by multiplying with a conversion factor of 0.5 assuming that the tree biomass contains 50% carbon (Ravindranath *et al.* 1997).

Although several workers have used tree height, trunk diameter i.e. diameter at breast height (dbh) and wood density as independent variables for estimating tree aboveground biomass (AGB), the allometric relationship between AGB and dbh has been proved to be the best fit for tree biomass estimation in several forests (Brown 1997; Brown *et al.* 1989). The biomass present in other compartments of the ecosystem such as shrub, herb, litter, woody debris, root biomass is added to the tree AGB values to obtain the size of the total carbon pool in a forest ecosystem. The application of allometric equations is a commonly used, non-destructive alternative in which biomass is estimated on the basis of easily measured attributes of trees (Dudley and Fownes, 1992; Ter-Mikaelian and Korzukhin, 1997; Tritton and Hombeck, 1982). The most reliable technique for estimating forest carbon stock is through forest inventories followed by developing allometric relationships between the aboveground biomass (AGB) of a tree and its trunk diameter (Brown 1997; Brown *et al.* 1989; Clark *et al.* 2001). Because AGB represents a large fraction of total forest carbon stock, its estimation offers a practical and reliable way of evaluating the carbon balance of tropical forest. Several studies have concluded that matured tropical forests with high AGB, contain a large proportion of their aboveground biomass in large trees (Brown *et al.* 1995; Brown and Lugo, 1992; Clark and Clark, 1996). In contrast, several other workers have argued that old growth forests have less potential for carbon sequestration as the constituent older trees cease to grow (Terakunpisut *et al.* 2007). The growth rate and the carbon sequestration potential of forest decline towards plant maturity. Beyond maturity, trees generally have marginal carbon sequestration capability (Lal and Singh, 2000). In a matured forest, growth rate is largely offset by wood decay (Chaturvedi 1994). Matured and climax forests are neither source nor sink of atmospheric carbon (Dabas and Bhatia, 1996). It is said that forests that experience a net loss of biomass volume through mortality due to disease or fire become net carbon emitters (Kyrklund 1990).

Ecosystem modelling studies (Chambers *et al.* 2001a; Tian *et al.* 2000) parameterize ecosystem characteristics such as species number, wood density, maximum tree size and tree standing stocks, and ecosystem processes such as growth, mortality and decomposition. Theoretically, the approach can avoid the spatial constraints of the eddy covariance studies and can predict basin-wide ecosystem response if the model is parameterized with data from many different locations. However, if the model is parameterized by measurements from a limited spatial area (Chambers *et al.* 2001a), then regional extrapolation may cause significant errors. In addition, ecosystem modelling is limited by uncertainties when estimating model parameters and by an incomplete knowledge of ecosystem processes. Often, the key results for modelling studies (e.g. annual carbon change) cannot be tested against empirical data. Biomass inventories such as that by Phillips *et al.* (1998) attempt to quantify carbon accumulation or release by monitoring tree growth and mortality in long-term forest plots. Biomass inventories can address the specific ecological changes of different forest components over time, but also suffer spatial limitations when scaling up to regional budgets. All of these studies suffer from experimental limitations and their estimates of carbon storage in neotropical old-growth forests are far from certain. Further work is imperative to document the role of old-growth neotropical forests in the carbon cycle and to understand the live and dead pools and fluxes of carbon dioxide within an undisturbed neotropical forest. All three previous approaches ignore forest dynamics, an important component of local, regional and global carbon budgets. Changes in forest demography or successional state can affect the carbon balance in live and dead biomass pools. Demographics and carbon balances can change for extended periods of time with disturbance events such as drought, fire or wind throw. Furthermore, live and dead pools may respond differently to climatic variation and disturbance events. For example, drought may cause a decrease in live biomass and increase in dead biomass because of high mortality and decreased decomposition of dead wood. For an accurate understanding of carbon flux, both pools must be quantified and the dynamics of each analyzed as potential ecological factors driving the carbon budget. Each of these analyses can help reveal the potential disturbance events and climatic factors that affected the site prior to measurement.

Globally, boreal, temperate and tropical forest systems contain approximately 1200 Pg C and cycle 70 to 90 Pg C annually (Kauppi *et al.* 1992; Sedjo 1992). Forest ecosystems can be net sources or sinks of CO2, depending on dominant biological or physical factors, including: (1) state of the soil and vegetation (i.e. is the system undisturbed, disturbed or recovering?); (2) management practices at the site level; (3) environmental conditions (e.g. climatic, edaphic, fire, pests); and (4) atmospheric deposition of pollutants and other compounds, some of which (e.g. CO2 and nitrogen) can serve as nutrients (Cropper and Gholz, 1993; Kauppi *et al.* 1992). Forests within tropical latitudes are being harvested for timber export, fuelwood, shifting cultivation, permanent agriculture, pasture, and urbanization (Detwiler and Hall, 1988; Iverson *et al.* 1993).

The most recent estimate of annual tropical deforestation is approximately 17 million ha (IPCC 1990). Land-use change in forest ecosystems of tropical latitudes is a major CO_2 source, but the range of emission estimates is around 1.6 ± 1.0 Pg C annually (Sundquist 1993). Although land-use change influences forest C emissions, many forest regions which previously were believed to be in C equilibrium actually may be net C sinks (Tans *et al.* 1990, Kauppi *et al.* 1992). For example, mature tropical forests in south-east Asia have been shown to be accumulating C (Iverson *et al.* 1993). In addition, high C storage rates have been observed in tropical forests recovering from logging (Brown *et al.* 1992). Temperate and boreal forest biomass has been expanding substantially in Europe, North America and possibly the former Soviet Union (Kolchugina and Vinson, 1993; Sedjo 1992). Inventories of western Europe reveal forest area and growing stock increased 30 and 25%, respectively, over the period 1971 to 1990 (Kauppi *et al.* 1992). At the individual plant level, photosynthesis is dependent on ambient CO_2 concentration, light, temperature and other factors (Cropper and Gholz, 1993; Mooney *et al.* 1991). Strain and Thomas (1992) reviewed the literature on plant response to elevated CO2 and concluded: (1) if other resources are present at required levels, CO_2 enrichment will increase photosynthesis and plant growth; (2) plants limited by resource deficiencies (e.g. nitrogen or phosphorus) will respond slightly or not at all to CO_2 enrichment; (3) CO_2 reduces transpiration and improves plant water status, due to increasing photosynthesis and decreased water loss; and (4) CO_2 and global warming may affect species differentially and will result in ecosystem flora and fauna change. The potential of individual trees to act as a C sink may be highly dependent on response to soil nutrition and environmental stress rather than to atmospheric CO_2 concentration (Norby *et al.* 1992). Forest ecosystem C balances appear to be sensitive to annual differences in climate and possibly CO_2 enrichment (Cropper and Gholz, 1993; Mooney *et al.* 1991). In addition, climate change may result in an increased rate of decomposition and plant respiration, thus releasing additional CO_2 to the atmosphere.

Three broad classes of forest management actions could influence C conservation and sequestration in forest ecosystems: (1) decreasing deforestation and forest degradation; (2) establishing additional areas of forest; and (3) implementation of practices which stimulate CO_2 fixation by existing forest or agroforest systems (Winjum *et al.* 1992). Trade-offs may exist between high rates of C assimilation and large amounts of C storage. Young trees have high growth rates but contain relatively little C, while the opposite is true of some mature trees (Cooper 1983). Consequently, if biomass is harvested to maintain high growth rates, the accumulated C must be stored to prevent its return to the atmosphere as CO_2. Harvested wood could be used as durable wood products or could be placed in long-term storage (Kolchugina and Vinson, 1993).

It is generally accepted that soil carbon (C) dynamics strongly interacts with atmosphere C dynamic and that, therefore, terrestrial ecosystems could partly buffer the rising atmospheric CO_2. Indeed, the pool of C in soil and vegetation is approximately three times higher than in the atmosphere (Schlesinger 1995) indicating that any increase of C sequestration by soils should significantly offset the rising of atmospheric CO_2 and the resulting global warming. Moreover, elevated CO_2 often stimulates primary production (Gill *et al.* 2002; Korner and Arnone, 1992) and greater C input is expected to increase C sequestration in soil (Karlen and Cambardella, 1996). However, many long-term field observations show that although plant material is incorporated to soil in large quantities, soil C content does not necessarily increase (Campbell *et al.* 1991; Gill *et al.* 2002; Korner and Arnone, 1992). These results suggest a negative relationship between C input and soil C conservation (Gill *et al.* 2002).

The major C pools such as phytomass, soil, litter, and fluxes of C due to litterfall and land use changes have been estimated for India based on very coarse resolution data and extrapolation, as the primary data for many regions of the country are either non-existent or over-estimated (Dadhwal and Nayak, 1993). The data available on carbon sequestration i.e. net woody biomass accumulation in trees for long term storage in tropical forests are extremely limited and incomplete. Because of the lack of reliable data on standing biomass and rates of forests degradation, the net annual carbon emission estimates for India have also been highly variable (Ravindranath *et al.* 1997). Thus, the improved quantification of C pools and fluxes in tropical forest ecosystems is important for understanding the contribution of these forests to net C emissions and their potential for Carbon sequestration (Chhabra and Dadhwal, 2004).

The soil organic carbon sequestration is caused by those management systems that add high amounts of biomass to the soil, cause minimal soil disturbance, conserve soil and water, improve soil structure, enhance activity and species diversity of soil fauna, and strengthen mechanism of elemental cycling whereas the reverse happens when severe depletion of SOC pool occurs (Lal 2004). Soil organic carbon have a very long residence time, hundreds and even thousands of years, compared with carbon stored in aboveground vegetation. The highest potential for increasing soil organic carbon content can most likely be found in severely degraded ecosystems or areas around the world (Olsson and Ardo, 2002).

Soil carbon pool, as the major part of the terrestrial carbon reservoir, plays an important role in the global carbon cycle. Therefore, the study of soil carbon dynamics is critically important to our ability to understand the carbon balance in these forests and their response to future global change (Davidson *et al.* 2000). More carbon can be stored below ground by increasing the input rate of organic matter, increasing the depth of carbon stock, boosting the carbon density in the soils, and decreasing the carbon turnover rate in soils (Post and Kwon, 2000). Carbon turnover rate varies considerably among forest soils. Differentiating the total soil organic carbon (SOC) into labile carbon (defined as its resident time from months to several years) and recalcitrant carbon (defined as its resident time from decades to thousands of years) could provide more information in understanding the mechanisms controlling the overall turnover rate of SOC pools in forest ecosystems (Sun *et* *al.* 2010). However, in our study no differentiation was made between labile and recalcitrant carbon and soil organic carbon is directly measured as total soil organic carbon.

Microbial biomass represents a relatively small standing stock of nutrients compared to soil organic carbon and aboveground biomass of trees, but acts as a labile source of nutrients for plants and a temporary sink of nutrients for plants. It is also understood that the microbial biomass is the driving force for organic matter transformation and nutrient cycling in soil systems. Measurement of microbial biomass has been used in studies on carbon flow, nutrients cycling and plant productivity (Voroney *et al.* 1989). The effects of microbial biomass in improving the soil fertility and primary production have been studied in organic matter decomposition (Parkinson and Coleman, 1991). He *et al.* (1997) pointed out that in weathered red soil, plant nutrients from mineral weathering were very limited and biological processes played a key role in the fertility sustainability of the soil. Microbial biomass may make contributions to nutrient availability to plant by being an important nutrient pool which is potentially available to plant, whereby microbial turnover acts as a dynamic source of soil available nutrients (Ladd and Foster, 1988).

The amounts of nutrients, particularly nitrogen (N), that are available in the soil, affect the biomass allocation and total primary production of plants, and forest growth is generally limited under nutrient-poor conditions (Gower *et al.* 1992). When N supply is limited, the annual carbon investment in fine roots increases, and consequently the growth of above-ground parts decreases (Keyes and Grier, 1981). In contrast, N fertilization has been shown to increase the total foliage mass and leaf area at the stand level (Gower *et al.* 1992).

Aboveground net primary production (ANPP) commonly reaches a maximum in young forest stands and decreases as stands mature. However, the mechanism(s) responsible for the decline are not well understood. Current hypotheses for declining ANPP with stand age include: (1) an altered balance between photosynthetic and respiring tissues, (2) decreasing soil nutrient availability, and (3) increasing stomatal limitation leading to reduced photosynthetic rates. Recent empirical and modelling studies reveal that mechanisms (2) and (3) are largely responsible for age-related decline in ANPP for forests in cold environments. Increasing respiratory costs appear to be relatively unimportant in explaining declining productivity in ageing stands (Gower *et al.* 1996).

Forest soils store an immense quantity of labile carbon (C) and may be a large potential sink for atmospheric C. In terms of C sequestration, the belowground C storage may account for over 70% of the total C stored in forest ecosystems (Schlesinger and Andrews, 2000). Common forest management practices, such as fertilization, may shift the C balance and enhance the total amount of C stored both above- and belowground (Valentini *et al.* 2000). Although aboveground C-biomass management has been of prime importance to forest managers for decades, the potential implementation of C credits may present opportunities for managing C storage belowground. Thus, there is increasing interest in maximizing ecosystem C sink and storage strength (Banfield *et al.* 2002; Field and Fung, 1999; Liski *et al.* 2002; Turner *et al.* 1995; Woodwell *et al.* 1983).

Increasing concentrations of atmospheric carbon dioxide could have significant implications for long-term storage of carbon in forest soils. Since forests account for more than 75 per cent of carbon stored in terrestrial ecosystems (Schlesinger 1997) and most of this carbon is stored below ground (Dixon *et al.* 1994), the effect of future levels of atmospheric CO_2 on soil C pools is of global significance.

Deforestation may weaken the carbon sink provided by forests, and in long run the world's forests may eventually become a source of carbon to the atmosphere. As a mitigation measure to global climate change due to greenhouse gas emission, it is required to cut the rate of emission either through reducing tropical deforestation or to enhance the natural carbon sequestration potential of degraded forests through forest regeneration and afforestation. The degraded areas have a large potential to sequester carbon in the soil, which may be preferable for storage in vegetation due to their longer residence time and less risk of rapid release to the atmosphere (Lal 2001). This can only be achieved through afforestation or reforestation of such areas. The protection of existing forests, regeneration of degraded forests and raising of forest plantations in India have been contributing to enhance carbon stock (Ravindranath *et al.* 2008).

With the advent of the Kyoto Protocol and its recognition of the use of forestry activities and carbon sinks as acceptable tools for addressing the issue of the build-up of atmospheric carbon, the potential role of planted forests as a vehicle for carbon sequestration has taken on a new significance. Additionally, the emergence of tradable emission permits and now tradable carbon offsets provides a vehicle for financially capturing the benefits of carbon emission reductions and carbon offsetting activities. In a world where carbon sequestration has monetary value, investments in planted forests can be made with an eye to revenues to joint outputs: timber and the carbon sequestration services (Sedjo 1999). However, producing both timber and carbon, forest plantation appears to offer potential to promote the economic development of a lagging region. The recent Kyoto Protocol has reinforced this interest by indicating that certain types of carbon sinks, and particularly forestry, may have the potential to capture carbon and that this sequestered carbon can be counted by Annex 1 countries in meeting their carbon obligations. Furthermore, forestry is particularly attractive due to the development of a Joint Implementation mechanism, which allows carbon projects be undertaken jointly between countries with the credits directed to the country needing the credits and revenues directed to the country in which the project is undertaken. This type of arrangement will probably also apply for the Clean Development Mechanism (CDM).

The north-eastern region of India with 99,260 km^2 of tropical forests (Roy and Joshi, 2002) that spread up to an elevation of 900 m a.s.l. in the Eastern Himalayas and sub-Himalayan areas offers appropriate situation for examining the above questions. The forests of the region include undisturbed evergreen and semi evergreen forests, secondary forests developed following shifting cultivation and forest degradation, and plantation forests raised by various agencies. Baring a few pockets of undisturbed forests, most tropical forests of the region were affected by one or the other form of cultural disturbances.

The objective of the present study is to investigate the following parameters:

1. To determine carbon sequestration potential of selected elements of tropical and montane sub-tropical landscapes.
2. To study the pool sizes and annual fluxes of carbon in selected landscape elements.

CHAPTER II

REVIEW OF LITERATURE

Carbon dioxide (CO_2) is a dominant greenhouse gas and its increase in the atmosphere is mainly attributed to fossil fuel combustion and deforestation (Hamburg *et al.* 1997). Trees act as a sink for CO_2 by fixing carbon during photosynthesis and storing excess carbon as biomass. The net long-term CO_2 source/sink dynamics of forests change through time as trees grow, die and decay. In addition, human influences on forests can further affect CO_2 source/sink dynamics of forests through such factors as fossil fuel emission and harvesting/utilization of biomass (Nowak and Crane, 2002). As the tree experiences growth, the carbon stock in the plant also increases. The rate of carbon storage is greater in young forest stands and it declines with stand age in pine species planted on cropland in the southeastern U.S. The rate of carbon storage began to decline from the age of 20 and was close to zero by the age of 100 (Veld and Plantinga, 2005). Increase in the atmospheric CO_2 concentration stimulates the photosynthetic rate of trees and can result in increased growth rates and biomass production. Results from free air CO_2 enrichment (FACE) experiments show at least 25% increase in growth rate in twice normal concentrations of CO_2. Growth is therefore higher in air with an elevated concentration of CO_2 (Burley *et al.* 2004). Many other workers have concluded that increased atmospheric CO_2 have positive effect on plants such as improved productivity (Idso and Kimball, 2001; Keutgen and Chen, 2001; Pan *et al.* 1998; Schaffer *et al.* 1997).

Carbon sequestration is currently being considered as a way to mitigate the greenhouse effects and simultaneously to combat land degradation (Olsson and Ardo, 2002). To this end, much current research is oriented towards a better understanding and quantification of carbon fluxes and stocks (Post and Kwon, 200). The environmental perspectives of carbon sequestration involve improvement in soil quality, and increase in biodiversity (Batjes 1999).

Heath and Smith (2000) argued that sequestering carbon will improve soil quality, as organic carbon influences many soil chemical and physical attributes including water holding capacity, nutrient retention, pH, structure and stability, and bulk density and penetration. Ganuza and Almendros (2003) studied the effects of climate, vegetation, and different edaphic variables on the organic carbon storage in the soils of the Basque country (Spain). They found that C accumulation in soil was regulated mainly by climatic factors, where edaphic variables such as the state of the exchange complex and texture played an important role in carbon sequestration.

Grayston *et al.* (2001) studied the role of MBC in increasing SOC and reported that increase in microbial biomass leads to increase in soil fertility. Fraser *et al.* (1988) observed a direct relationship between microbial activities and the amount and quality of carbon and other nutrients available from plant residues and root exudates. Witter and Kanal (1998) found close relationship between the amounts of MBC and C concentration in different soils with variation in the ratio of MBC to organic carbon. They concluded that a difference in MBC to organic carbon ratio between the soils is mainly due to difference in the quality of soil organic matter rather than intrinsic difference in microbial efficacy of substrate utilization.

Lavahun *et al.* (1996) have also reported variation in the distribution of MBC at different depths of grasslands and two arable soils, and attributed it to the decline in organic carbon content. According to Anderson and Domsch (1989), the ratio of microbial biomass C to soil organic C provides an insight into the C status of soil. A decline in the MBC/SOC ratio indicates decline in soil organic matter (Brookes 1995). In a field study conducted by Holmes and Zak (1994), N availability was not controlled by microbial activities, but by changes in the turnover rate of the microbial biomass.

Diaz-Ravina *et al.* (1995) found significant variation in the microbial biomass in forest soils due to soil type and seasonal changes. Their study confirmed that variation in the microbial biomass in the soils is mainly due to the type of soils which explained 71% of the total variation rather than the season that accounted only 18% variation. The interaction between the soil type and season explained only 8% variation. They concluded that soil type was the most important factor for observed variation in microbial biomass in forest soils. The findings of Zeller *et al.* (2001) suggest that influence of the site and sampling time is stronger than the management regime or land abandonment. The differences in the microbial biomass C may be due to the climatic conditions, differences in ground cover vegetation, the number of roots, soil types and properties, types of land use and management, as well as variations in sampling time (Anderson and Domsch, 1989; Murrieta *et al.* 2007). Holmes and Zak (1994) concluded that N availability was not controlled by microbial activities, but by changes in the turnover rate of the microbial biomass.

Arunachalam *et al.* (1998b) studied the litterfall pattern and decomposition in the sub-tropical forests of Meghalaya and concluded that bioelements released through litter decay play a fundamental role in maintaining the sustainability of natural forests and thus influence primary productivity. Facelli and Pickett (1991) reported that species composition is an important factor that determines litter production within the same climate range. Kamei (2007) studied the seasonal pattern of litterfall in sub-tropical forests of Meghalaya and found that the climatic factors control the litterfall pattern rather than the species composition in the forest. Vogt *et al.* (1986), assigned rapid decomposition of litter as an important cause of lower litter accumulation on the forest floor of wet tropical forest. Gallardo and Merino (1993) found that litter decomposition rates are regulated by soil organisms, environmental conditions and chemical nature of the litter. Pargasan and Parthasarathy (2005) also reported that the significant differences in total litter accumulation between two tropical dry evergreen forest sites on the Coromandel coast of south India, was due to differences in temperature, soil moisture and decomposition rates besides vegetation variables. Several workers (Haase 1999; Norgrove and Hauser, 2000; Sundarapandian and Swamy, 1999; Yang *et al.* 2005) reported that the quantity of litterfall varies greatly over a range of spatial and temporal scales and is determined mainly by climate, seasonality, topography, site quality, and species composition. It has been reported that under elevated atmospheric CO_2 levels, there was an increase in productivity and litterfall (Finzi *et al.* 2001; Schlesinger and Lichter, 2001; Zak *et al.* 2003).

William *et al.* (2000) studied the doubling of atmospheric CO_2concentration in a FACE experiment and found that the average soil water content in the first 15 cm of the soil profile was approximately 15% greater beneath the chambers receiving the extra supply of CO_2, presumably due to CO_2-induced reductions in plant stomatal conductance that blunted transpirational water loss. This resulted in enhanced productivity of both above and belowground compartments including soil microbial activity and there was 8% increase in total soil carbon content. Hu *et al.* (2001) studied the negative feedback and found that doubling of air's CO_2 content, increased both microbial biomass and plant nitrogen uptake. Certini *et al.* (2002) studied the carbon dioxide efflux and concentration in two soils under temperate forests and found that the CO_2 concentration increased considerably with depth but microbial metabolism decreased from the top to the bottom of the soil profile.

The total area of the world's forest stands at 3.952 billion hectare (FAO 2005), which was about 30% of the total land area of the world. It is estimated that the world's forests store 283 Gt of Carbon in their biomass alone, and 638 Gt of carbon in the ecosystem as a whole including dead wood, litter and soil up to 30 cm depth. Vast forest areas in India accumulates a large amount of carbon as CO_2 from the atmosphere and play an important role for sequestering carbon in regional, national and world scenarios. Terrestrial (plant and soil) carbon was estimated at 2000 ± 500 Pg, which represents 25% of global carbon stocks. The analysis of C stocks from various parts of the world showed that significant quantities of C (1.1 - 2.2 Pg) could be removed from the atmosphere over the next 50 years if agroforestry systems are implemented on a global scale (FSI 2005). Studies carried out by different scientists for different countries showed that the forests of United States have 12.1 Pg of carbon (Turner *et al.* 1995), those of Europe accumulate 7.5 Pg of carbon (Kaupii *et al.* 1992), Chinese forests stock 4.63 Pg (Fang *et al.* 2001) and Japanese forests accumulated 1.39 Pg carbon (Alexandrov *et al.* 1999).

The Kyoto Protocol negotiated a framework for reducing the emission of greenhouse gases in December 1997. The protocol also recognized that some terrestrial ecosystems have the potential to sequester large amounts of carbon and thus further slow down the increase of atmospheric CO_2 concentrations (UNFCC 1997). The global soil carbon (C) pool of 2500 gigatons (Gt) includes about 1550 Gt of soil organic carbon (SOC) and 950 Gt of soil inorganic carbon (SIC). The soil C pool is 3.3 times the size of the atmospheric pool (760Gt) and 4.5 times the size of the biotic pool (560 Gt). The total carbon pool in forest ecosystem was recently estimated to be about 1150 Gt (Dixon *et al.* 1994), of which 49% is in the boreal forests, 14% in temperate forests and 37% in tropical forests. A further 1000 Gt are estimated to reside in non-forest ecosystems, such as savannas, grasslands, tundra, peat lands and wetlands (Adam *et al.* 1990). According to the figures of Dixon *et al.* (1994), 65% of the carbon is stored as soil organic carbon, and 31% as living biomass. However, there is a marked contrast between high latitude and the tropical forests in the size of these pools. In the boreal zone, 84% of the carbon is in the soil organic matter and only 16% in the active living biomass, whereas in the tropics the carbon is partitioned more or less equally between vegetation and soil. The primary cause for this difference is temperature. At higher latitudes it limits the growing season, and restricts decomposition and nutrient recycling, but at low latitudes it supports rapid decomposition of soil organic matter and rapid recycling of nutrients into vegetation growth.

Lal *et al.* (1999) studied the carbon sequestration potential of tropical forests and found that degraded areas have a large potential to sequester carbon in the soil, which may be preferable for storage in vegetation due to their longer residence time and less risk of rapid release to the atmosphere. Kimble *et al.* (2001) reported that the top 30 cm is where the greatest change in soil carbon storage for most crops occurs. Baishya and Barik (2011) reported the carbon sequestration in soils of the old-growth pine forest to be 58.7 Mg ha^{-1} upto a depth of 100 cm. Recent technological innovation aims at capturing CO_2 from the atmosphere and storing below the surface for longer periods of time. To slow the atmospheric buildup of CO_2, a report from the U.S. National Research Council recently called for building a suite of 15 to 20 power plants with carbon capture and storage (CCS) before 2020 (*Haszeldine* 2009).

The tropical rainforests account for 36% of the global net primary production and contain 59% of the global forests carbon pool (Dixon *et al.* 1994). Tropical soils represent at least 32% of the total mass of organic carbon stored in the soils of the world (Eswaran *et al.* 1993). Among tropical ecosystems, the Amazonian forest is known to play a major role in carbon sequestration and release (Cerri *et al.* 2004). However, global estimates of carbon storage in this ecosystem are few (Moraes *et al.* 1995). These carbon pools are difficult to estimate because of the limited availability of reliable, complete and uniform data for soil (carbon concentration and bulk density down to a sufficient depth) as pointed out by Batjes (1996).

The carbon sink in the old-growth tropical forests was estimated to be between 0.4 Mg C ha^{-1} yr^{-1} (Chambers *et al.* 2001b; Tian *et al.* 2000) and 5.9 Mg C ha^{-1} yr^{-1} (Malhi *et al.* 1998). Vetter *et al.* (2005) studied the carbon sequestration potential of managed coniferous forest using BIOME-BGC model and found that young and old stands had the highest changes in the biomass C accumulation and matured forest stands $\geq$80 years old turned from being carbon neutral to carbon sink. They stated that nitrogen deposition was responsible for the increase of net ecosystem production (NEP) of forests at higher elevations. CO_2 fertilization was the main factor increasing NEP of forests in the middle and lower elevations. Cuevas *et al.* (1991) found that the tropical pine plantation and broad-leaved secondary forests had similar net primary productivity. However, the plantation forest allocated most of its production to the aboveground compartment (>90%), while the secondary forests allocated upto 50% to the belowground compartment.

Dixon *et al.* (1994) estimated that the global forest vegetation and soils contain about 1146 petagrams of carbon with 359 and 787 petagrams of carbon in vegetation and soils, with approximately 37% of this carbon in low-latitude forests, 14% in mid-latitudes and 49% in high latitude forests.

Winjum *et al.* (1992) used an extensive database compiled from the literature to estimate the amount of C sequestered per unit area over a 50 years period for five major forestry practices in the tropics: reforestation, afforestation, natural regeneration, silviculture and agroforestry practices and found that natural regeneration had the highest mean storage carbon. Nilsson and Schopfhauser (1995) studied the C sequestration potential through a global afforestation program considering above and belowground biomass, rates of forest growth, economic and political factors, and even species-specific C accumulation rates. They found that significant C accumulation would take on the order of 40 to 50 years from the initiation of a global afforestation program, and that a maximum C fixation rate of 1.5 Gt of C annually would be reached about 60 years after the initiation of the programme. One hundred years after the initiation of the project, 104 Gt of C would be sequestered.

Baishya *et al.* (2009) studied the distribution pattern of carbon in tropical natural semi-evergreen forest and sal plantation forest in Meghalaya, and found that carbon sequestration potential in the sal plantation forest (231 Mg C ha^{-1}) was higher than the natural old-growth broad-leaved forest (162 Mg C ha^{-1}) due to site factors and adopted management practices.

Baishya and Barik (2011) studied the biomass and carbon pool in different compartments of 65 - 80 years old sub-tropical old-growth *Pinus kesiya* forest in Meghalaya and revealed that the carbon pool and carbon sequestration potential of the forest was high with carbon sequestration value of 205.7 Mg C ha^{-1} and NPP of 17.5 Mg ha^{-1} yr^{-1}, signifying the importance of the species for future management strategies. The above- and belowground carbon contribution to the total ecosystem carbon was 91.2% and 8.8%, respectively. They attributed that high productivity of *P. kesiya* may be due to high net assimilation rate and prolonged photosynthetic activity, and higher uptake of nutrients due to rapid turnover of nutrients (Das and Ramakrishnan, 1987).

Kaul *et al.* (2010) estimated the carbon sequestration potential of sal (*Shorea robusta* Gaertn. f.), Eucalyptus (*Eucalyptus tereticornis* Sm.), poplar (*Populus deltoides* Marsh), and teak (*Tectona grandis* Linn. f.) forests in India by using dynamic growth model (CO2 FIX). They concluded that long-term total carbon storage ranges from 101 to 156 Mg C ha^{-1}, with the largest carbon stock in the living biomass of long rotation sal forests (82 Mg C ha^{-1}). The net annual carbon sequestration rates were greater for fast growing short rotation poplar (8 Mg Cha^{-1}yr^{-1}) and Eucalyptus (6 Mg C ha^{-1}yr^{-1}) plantations followed by moderate growing teak forests (2 Mg Cha^{-1}yr^{-1}) and slow growing long rotation sal forests (1 Mg C ha^{-1}yr^{-1}). They found that by extending rotation length from the recommended 120 to 150 years increases the average carbon stock of forest ecosystem can be increased by 12%.

Tanabe *et al.* (2003) assessed the above- and below-ground biomass and net primary production of a 85-year-old matured *Pinus densiflora* forest established on a lava surface of Mt. Fuji in central Japan and found that 74% of the total biomass was in the aboveground compartment. The aboveground NPP was 61% of the total NPP under the condition of low nitrogen concentration and high soil C/N ratio.

Terakunpisut *et al.* (2007) reported an AGB value of 275 Mg ha^{-1} for the tropical rain forests of Thailand. Brown and Lugo (1984) reported AGB values of 170 Mg ha^{-1} for the broadleaved forests of tropical America, 260 Mg ha^{-1} for tropical Africa, 215 Mg ha^{-1} for tropical Asia and 150 Mg ha^{-1} for total tropics. Whittaker (1996) reported 500-600 Mg ha^{-1} of AGB for the undisturbed deciduous forest in the southern Appalachain Mountains.

The large trees contributed 49% to the total AGB in natural forest (Brown 1996; Brown *et al.* 1995; Brown and Lugo, 1992; Clark and Clark, 1996) who reported up to 50% contribution to AGB by the large trees (> 70 cm dbh). On the other hand, Brown *et al.* (1997) reported that smaller trees contribute to most AGB in forests with <300 Mg ha^{-1} aboveground biomass. Analyses have shown that forests with reduced biomass have either had their large trees removed by past human disturbance or represent regenerating secondary forests which do not yet have large trees. The distribution of biomass in large trees, therefore, could be an indicator of the presence or absence of past anthropogenic disturbance (Brown 1996). Iverson *et al.* (1993) studied the carbon sequestration potential of tropical Asia and conclude that degraded forest can be potential carbon stores if adequate protection is provided.

Lloyd *et al.* (1984) studied broad-scale impacts of harvesting on coastal redwood carbon stores by estimating carbon fluxes from 1910 through 1984 using historic inventory data from 4 country of northern California and found that the average-site carbon levels have declined from 629 Mg ha^{-1} in 1910 to 152 Mg ha^{-1} in 1984. This represents a net loss of 76% of carbon stores and indicates that the north costal region of California may have been a net source of atmospheric CO_2 from 1910 to 1984. Poffenberger *et al.* (2001) reported sequestration levels varying between 0.5 to 3.4 tonnes per hectare per year in community managed teak forests of Harda forest division of Madhya Pradesh (India).

Bhat *et al.* (2003) studied the carbon stock dynamics in the tropical rainforests of the Uttar Kannada district in Western Ghats, India and found that there was increased carbon assimilation in the reserve forests than the inner forests. Ramachandran *et al.* (2007) reported a value of 307 Mg ha^{-1} for the tropical evergreen forests of eastern coast of Tamil Nadu, India. The AGB values of 607.7 Mg ha^{-1} and 468 Mg ha^{-1}, respectively were reported for tropical wet evergreen forest and tropical semi evergreen forest of Western Ghats of India by Rai (1981) and Swamy (1989). Muller (1982) reported an AGB of 330 Mg $ha^{-1}$for the tropical forests of eastern hardwood region of USA. Brown and Lugo (1982) also reported AGB for the tropical rain forests in Malaysia (225 - 446 Mg ha^{-1}), Cameroon (238 - 341 Mg ha^{-1}) and Sri Lankan tropical rain forests (153 - 221 Mg ha^{-1}).

Haripriya (2003) studied the carbon budget of the Indian forest ecosystems and found that the Indian forest sector acted as a source of 12.8 TgC for the year 1994 and brought to notice the need to evaluate the country's forest sector in influencing the global atmospheric carbon levels. Lal and Singh (2000) reported that estimated annual carbon uptake increment by Indian forests and plantations have been able to remove about 0.12 Gt of CO_2 from the atmosphere in the year 1995.

Jana *et al.* (2009) studied the carbon sequestration rate and aboveground biomass carbon potential of four young species of *Shorea robusta*, *Albizzia lebbek*, *Tectona grandis* and *Artocarpus integrifolia*. Ravindranath *et al.* (1997) reported the Indian forests based on the forest cover of the year 1986 could sequester around 5 TgC (1 Tg = Tera gram, 10^{-12} g). A study reported by Warran and Patwardhan (2008) on carbon sequestration potential of trees and estimated the standing biomass in India to be 8375 million tons (Mt) for the year 1986, of which the carbon storage would be 4178 Mt. The total carbon stored in forests, including soil is estimated to be 9578 Mt (Warran and Patwardhan, 2008). Haripriya (2003) noted that on an average biomass carbon of the forest ecosystems in India for the year 1994 was 46 Mg C ha^{-1}, of which nearly 76% was in aboveground biomass and the rest in fine and coarse root biomass. The average carbon stock for the country was 24.94 t C ha^{-1} in 1984 and 24.54 t C ha^{-1} in 1994.

Das and Ramakrishnan (1987) studied the biomass and productivity studies of 5-22 years old *Pinus kesiya* forest in Meghalaya and reported a total aboveground NPP of 30.1 Mg $ha^{-1}yr^{-1}$ in the younger and 20.1 Mg $ha^{-1}yr^{-1}$ in older forest. Chaturvedi and Singh (1982, 1987) and Rana *et al.* (1989) calculated the aboveground NPP value of 6.1-15.6 Mg $ha^{-1}yr^{-1}$ for *Pinus roxburghii* forest in the central Himalayas of India. Karizumi (1974) studied the ecosystem level NPP in the *Pinus densiflora* forest in Japan and reported an aboveground NPP of 10.0 - 14.6 Mg $ha^{-1}yr^{-1}$ and 3 - 6 Mg $ha^{-1}yr^{-1}$ as belowground NPP.

Ravindranath *et al.* (2006) studied the impact of climate change on forests in India using climate projection models under A2 and B2 scenarios. They concluded that under the climate projection for the year 2085, 77% and 68% of the forested grids in India are likely to experience shift in forest types under A2 and B2 scenario, respectively. They also found that there is a shift towards wetter forest types in the north-eastern region and drier forest types in the north-western region of India in the absence of anthropogenic disturbances. Increasing atmospheric CO2 concentration and climate warming could also result in a doubling of net primary productivity under the A2 scenario and nearly 70% increase under the B2 scenario.

Flint and Richards (1994) studied the carbon storage in the vegetation of India from the year 1880 onwards and reported a decreasing trend. Changes of land-use in the past were dominated by losses of fertile land and degradation of forests. Bhadwal and Singh (2002) studied the carbon sequestration potential in forestry options under different land-use scenarios in India using the LUCS model for a period of 50 years from 2000 and found that the carbon sequestered in aboveground vegetation of India will be more than double by the year 2050. The amount of carbon sequestered by the plantations will be the maximum. Thus, the plantation-based scenario was identified as the best land-use pattern which fulfils the demands in future in a sustainable manner on a long-term basis.

The conversion of forests to agriculture and other human uses lead to a net release of CO_2 to the atmosphere (Clarke 1982; Houghton *et al.* 2000; Likens *et al.* 1970; Malhi *et al.* 1999; Palm *et al.* 1986; Singh *et al.* 1991; Wagai *et al.* 1998). The land-use changes that influence soil C release within the tropics can have larger implications for global C cycling (Hannah *et al.* 1994; Houghton 1994; Olson *et al.* 1983). Changes in soil C following deforestation have become a global concern both in terms of sustainable production at local and regional scale (Tiessen *et al.* 1994) and the global consequences relating to increased emissions of CO_2 from terrestrial systems (Houghton 1991). Many important global and regional soil C budgets are available for some biomes (Batjes 1996; Bhadwal and Singh, 2002; Eswaran *et al.* 1993; Gupta and Rao, 1994; Post *et al.* 1982; Schlesinger 1997; Singh *et al.* 1985), and a few have directly examined the effect of more subtle land-use changes on soil surface CO_2 flux (Wagai *et al.* 1998).

Soil respiration is a major flux in the carbon cycle, second in magnitude to gross primary productivity, which ranges from 100-120 Pg C yr^{-1} and equal or greater than the estimated global terrestrial net primary productivity of 50-60 Pg C yr^{-1} (Ajtay *et al.* 1979; Bolin 1983; Box 1978; Olson *et al.* 1983). Despite its importance in the global carbon cycle, the magnitude of soil respiration as affected by different land-uses is poorly quantified. The CO_2 emission through respiration, both by the vegetation itself and decomposition of organic matter increases with global warming (Grace and Rayment, 2000). Giardina and Ryan (2000) studied the decomposition pattern in Finnish soils and concluded that decomposition of organic matter is not very sensitive to temperature over long periods of time. Valentini *et al.*

(2000) studied the CO_2 flux in the forest of Europe forests and found that respiration is a more important component of carbon balance than low temperature in northern latitudes. Grace and Rayment (2000) argued that carbon fluxes in the tropics are larger than those in temperate and northern forests.

Possible adverse consequences of climate change resulting from increasing levels of atmospheric CO_2 have drawn attention to the inventory and dynamics of carbon in the biosphere (Chan 1982; Kellogg 1982). Carbon exchange between the terrestrial ecosystems and the atmosphere is one of the key processes that need to be assessed in the context of the Kyoto Protocol (IGBP Terrestrial Carbon Working Group 1998).

About 500 billion tons of carbon are stored in vegetation worldwide. Deforestation and forest degradation alone accounts for 17.4% of the world's greenhouse gas emissions. The problem is especially acute in tropical and subtropical forests where carbon stocks are decreasing at an alarming rate of 1-2 billion tons a year. Initiatives in forest conservation and enhancement which take into account the livelihood concerns of poor and socially marginalized people dependent on the forests can help to address this situation. In this context, 'Reducing Emissions from Deforestation and Forest Degradation (REDD)' has recently received special attention in the climate-change debate. When properly designed, REDD schemes can provide a sound bridging mechanism in the transition towards a low carbon economy. They can contribute to improving rural livelihoods, promoting good forest governance, delivering biodiversity objectives, and increasing resilience and adaptive capacities to climate change. Carbon accounting in a forest is one of the most crucial steps for successful implementation of REDD projects. The process needs to meet international standards and, at the same time, be manageable in a cost-effective manner within the local context. Reducing emissions from deforestation and forest degradation (REDD) has now gained focussed attention in international climate negotiations. Evolving discussions on REDD have brought forests to the forefront of both climate-change mitigation and adaptation. Among others, successful REDD programs require reliable, accurate, and cost-effective methods for measurement and monitoring of forest carbon storage. For implementation of Kyoto Protocol till 2012, and implementation of REDD+ beyond 2012, the quantification of carbon in forests is extremely important.

However, most forest types of north-east India have not been studied with an aim to quantify their carbon stock and flux rates.

CHAPTER III

STUDY SITES

The study was conducted in the state of Meghalaya which lies between 25°02′ and 26°07′ N latitude, and 89°49′ and 92°50′ E longitude with an area of 22, 429 km^2 (Fig.3.1). Being a part of the northeastern Indian bio-geographic zone (Rodgers and Panwar, 1988), it constitutes the meeting place of paleo-arctic, Indo-Malayan and Indo-Chinese bio-geographic realms. The topography of the state is variable and the elevation ranges from 50 m to 2040 m a.s.l.

Tropical forest of Nongkhyllem wildlife sanctuary and reserve forest

The study site is located near Lailad which is about 79 km north of Shillong, the state capital of Meghalaya. The sanctuary lies between 25°45' – 26°00' N latitude and 91°45' – 92°00' E longitude. The Nongkhyllem wildlife sanctuary and Nongkhyllem reserve forest are continuous. The Nongkhyllem reserve forest was constituted in the year 1910 with 96.91 km^2 area. The Nongkhyllem wildlife sanctuary was carved out of Nongkhyllem reserve forest in the year 1981 for adequate protection of the floral and faunal species of the area and covers an area of 29 km^2 on steep hill slopes (20° to > 65°) with an elevation ranging from 205 to 297 m. The Umtrew river is the major river in the study area and it marks the western boundary of the Sanctuary and the Reserve Forest. The river Umran forms the dividing boundary between the sanctuary and the reserve forest in the east and joins river Umtrew through the northern half of the sanctuary. The river Umling forms the northern boundary of the reserve forest and joins river Umtrew near the office of the sanctuary.

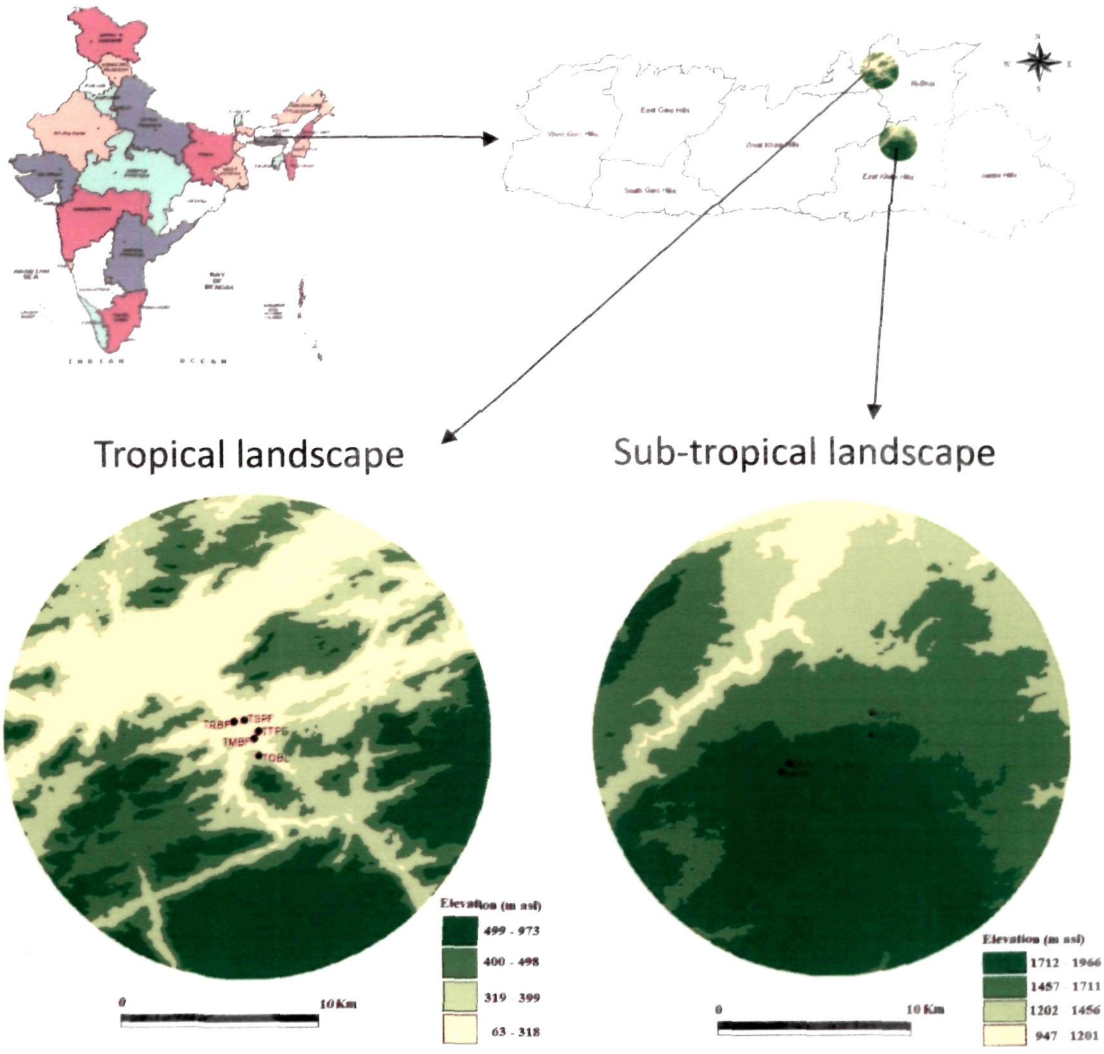


Fig. 3.1: Map showing location of study sites

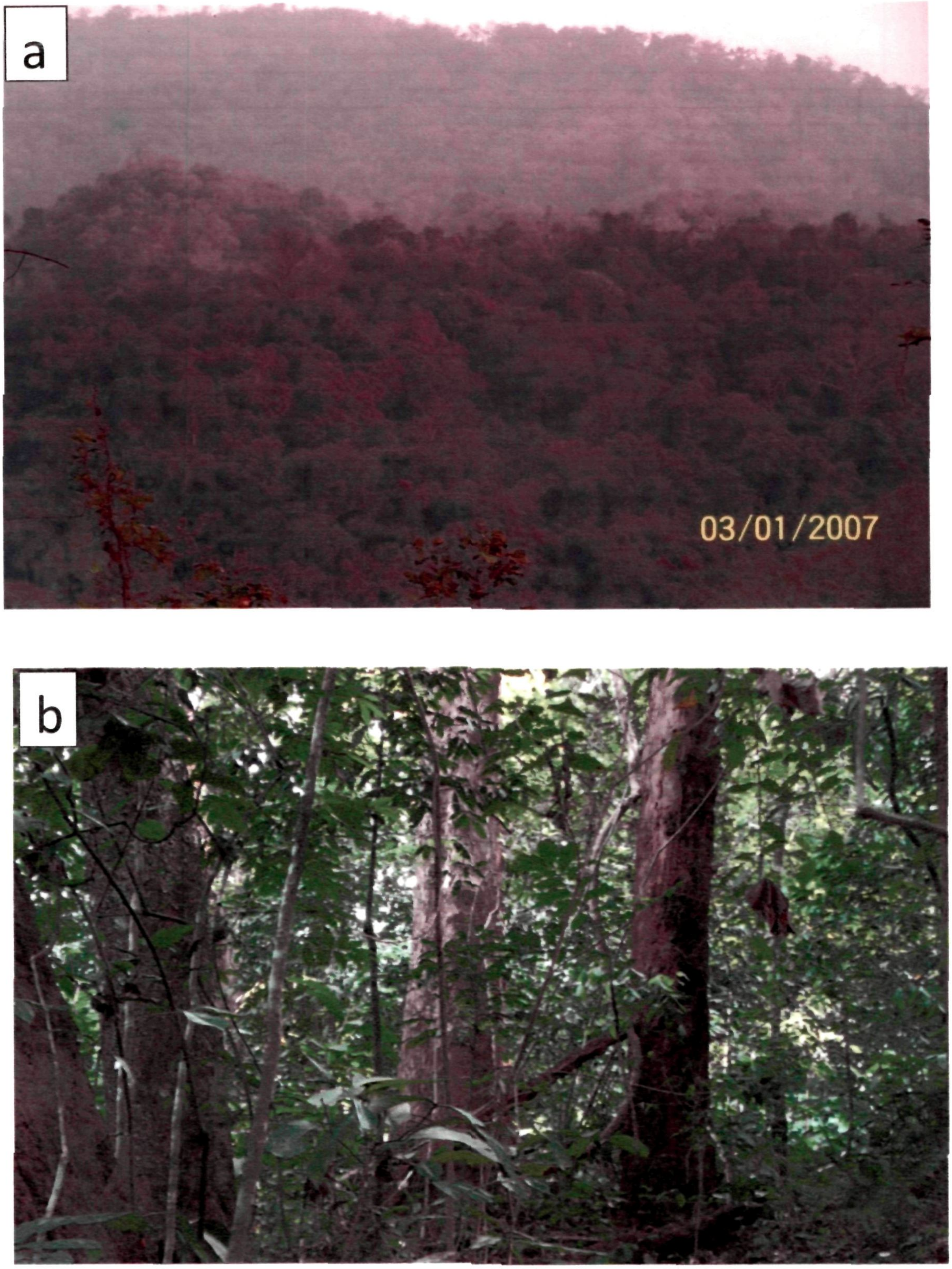


Plate 1: An overview (a) and interior view (b) of old-growth broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

Plate 2: Interior view of regenerating broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

Plate 3: Interior view of teak (*Tectona grandis*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

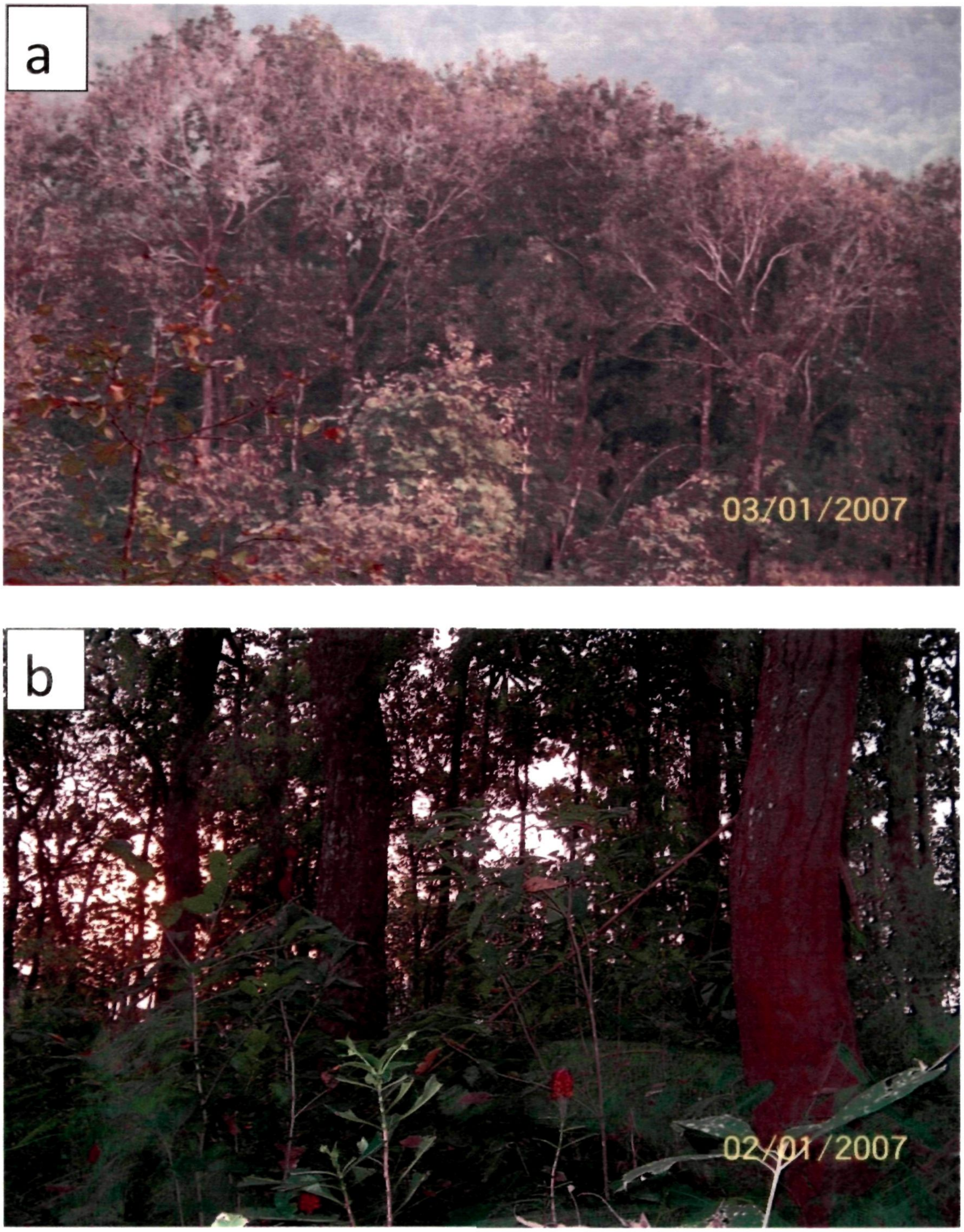


Plate 4: An overview (a) and interior view (b) of sal (*Shorea robusta*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

Plate 5: An overview of mixed bamboo forest in tropical landscape of Nongkhyllem wildlife sanctuary

Plate 6: Interior of old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Plate7: Interior view of old-growth pine (*Pinus kesiya*) forest in montane sub-tropical landscape of Upper Shillong

Plate8: An overview of regenerating pine (*Pinus kesiya*) forest in montane sub-tropical landscape of Upper Shillong

Plate 9: An overview of regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Table 3.1 Forest characteristics of the tropical and montane sub-tropical forest ecosystems

Tropical forest ecosystem	Age (year)	Area demarcated (ha)	Area sampled (ha)	Altitude (m.a.s.l)	Longitude	Latitude
Old-growth broad-leaved forest (TOBF)	80	20	3	280-297	E 091°46.546'- E 091°46.561'	N 25°55.578‘- N 25°55.592'
Regenerating broad-leaved forest (TRBF)	20-22	10	3	210-220	E 091°46.198'- E 091°46.219'	N 25°56.026‘- N 25°56.045’
Teak plantation forest (TTPF)	35-40	10	3	205-214	E 091°46.679'- E 091°46.685'	N 25°56.004‘- N 25°56.023'
Sal plantation forest (TSPF)	40-45	20	3	230-249	E 091°46.453'- E 091°46.472'	N 25°56.062'- N 25°56.134'
Mixed bamboo forest (TMBF)	65-70	20	3	225-236	E 091°46.468'- E 091°46.481'	N 25°56.617'- N 25°56.625'
Montane sub-tropical forest ecosystems						
Old-growth broad-leaved forest (SOBF)	65-80	20	3	1780-1791	E 091°53.329'- E 091°53.335'	N 25°32.929‘- N 25°32.942'
Old-growth pine forest (SOPF)	65-80	20	3	1672-1692	E 091°53.338'- E 091°53.366'	N 25°33.456'- N 25°33.486'
Regenerating pine forest (SRPF)	15-17	10	3	1890-1907	E 091°51.231'- E 091°51.247'	N 25°32.212'- N 25°32.231'
Regenerating broad-leaved forest (SRBF)	12-15	10	3	1915-1920	E 091°51.042'- E 091°51.076'	N 25°32.008'- N 25°32.043'

The study was conducted in five forest types *viz.*, tropical old-growth broad-leaved forest (TOBF) (25°55.578' N and 091°46.546' E), tropical regenerating broad-leaved forest (TRBF) (25°56.026' N and 091°46.198' E), tropical teak plantation forest (*Tectona grandis* L. f) (TTPF) (25°56.023' N and 091°46.481' E), tropical sal plantation forest (*Shorea robusta* Gaertn.f.) (TSPF) (25°56.062' N and 091°46.453' E), and tropical old-growth mixed bamboo forest (TMBF) (25°56.625' N and 091°46.679' E) (Table 3.1). These five forest types were selected because tropical semi-evergreen forests constitute the majority of natural tropical forests of the sanctuary, and sal and teak plantation forests dominate the plantation forest in the region. The regenerating broad-leaved forest is the result of past anthropogenic disturbance for jhum cultivation and timber and was selected because of their growth potential following disturbance. The bamboo forests dominate the tropical landscape with high growth rate and having high socio-economic importance for the local people. The tropical old-growth broad-leaved forest and mixed bamboo forest are part of the humid tropical forest of Nongkhyllem wildlife sanctuary. The sal and teak plantation forests are part of the Nongkhyllem reserve forest. The regenerating broad-leaved forest is located opposite to the office of the sanctuary towards the west of the river Umtrew at old Lailad. The sal and teak plantation forest were created during the year 1972 and are well-protected. On the other hand, the natural forest is an old growth forest, where the past disturbance in the form of selective logging took place at least 80 years ago prior to its declaration as a wildlife sanctuary. The forest is currently undisturbed and is characterized by dense canopy cover.

Montane sub-tropical forest of Upper Shillong

The study site is located at upper Shillong which is about 15 km from Shillong, the state capital of Meghalaya. The study was conducted in four forests *viz.*, sub-tropical old-growth broad-leaved forest (SOBF), sub-tropical old-growth pine forest (SOPF), sub-tropical regenerating pine forest (SRPF) and sub-tropical regenerating broad-leaved forest (SRBF). The pine forest was selected because of its dominance in the region and the broad-leaved forest was the original primary vegetation of the region and due to disturbances it is now present only in patches. The regenerating broad-leaved and regenerating pine forests were selected because of their regenerating potential after disturbance. The old-growth broad-leaved forest is located in the Laitkor protected forest (25°32.935' N and 091°53.329' E). The old-growth pine forest is part of the Malki protected forest (25°33.486' N and 091°53.338' E) and is natural, continuous, well-protected, 65-80 years old and is dominated by *Pinus kesiya* Royle. ex. Gordon. The forest is classified as Assam sub-tropical pine forest (Champion and Seth, 1968). The regenerating broad-leaved forest (25°32.008' N and 091°51.056' E) is about 12-15 years old since last disturbance after the primary forest of upper Shillong Sacred groove was cleared. The regenerating pine forest (25°32.223' N and 091°51.231' E) is a 15-17 years old degraded pine forest with *Pinus kesiya* as the dominant species (Table 3.1).

Geology

Meghalaya represents the remnant of the ancient plateau of Pre-Cambrian Indian peninsula and forms a prominent geomorphic unit stretching across the Garo, Khasi and Jaintia hills in east-west direction. The central and northern part of Meghalaya plateau is made up of "Archaean Gnessic Complex" which is essentially highly metamorphosed crystalline rocks of Pre-Cambrian origin. This complex is made up of gneisses (biotic gneiss, biotite granulite, quartz-sillimanite gneiss, cordierite, garnet and chondrodite) and Schistose (mica schist, quartz-sillimanite schist and metabasite) members of varying composition (Anon 1974; Murthy *et al.* 1976).

Climate

The climate of the state is monsoonic and is directly influenced by the southwest monsoon originating from the Bay of Bengal. The climatic variables like temperature, rainfall and humidity vary widely from place to place in the state due to wide variation in topography. Based on the climatic conditions, the year may be divided into spring, summer, autumn and winter seasons. The summer (May to September) is characterized by relatively high temperature, occasional thunderstorms with high rainfall and high wind velocity. The rainy season commences with the onset of southwest monsoon in May and continues up to September. Three fourth of the total annual rainfall is received during this season. The rainy season is followed by a brief autumn during October and November. Rainfall and temperature sharply decline during this period. The winter season extends from December to February. Morning fog and frost, and dry weather are the characteristic features of this season. March to April represent the spring season.

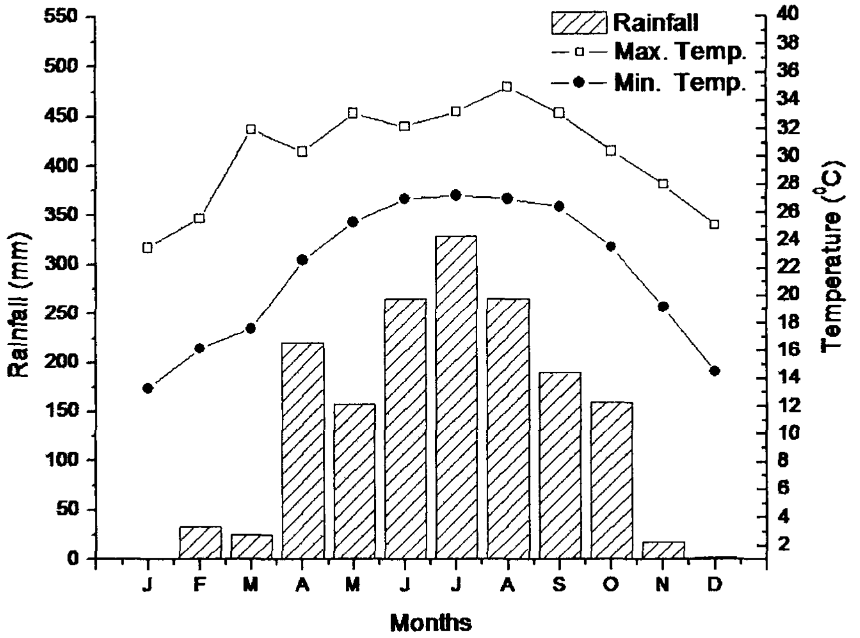


Fig.3.2 Mean annual rainfall and temperature in tropical landscape of Nongkhyllem wildlife sanctuary during 2005–2008

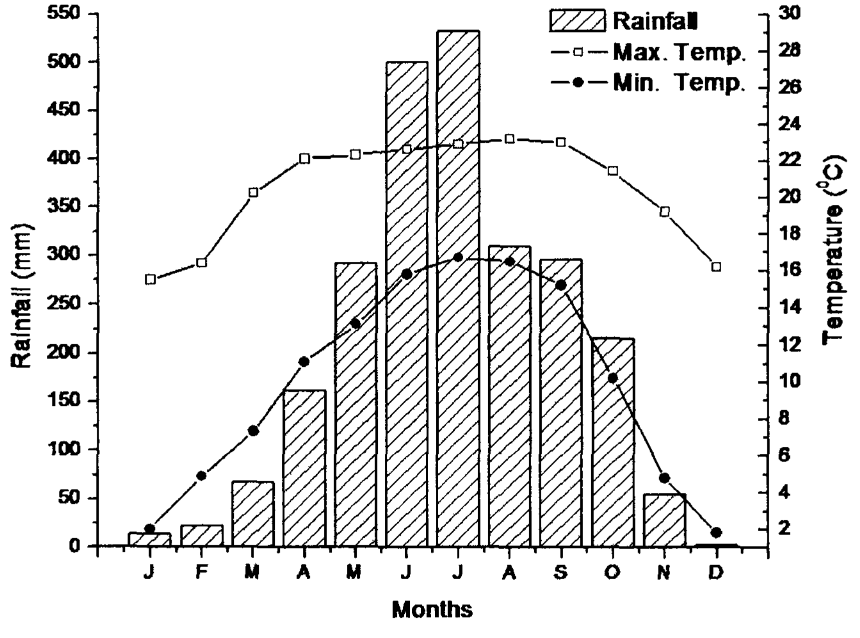


Fig.3.3 Mean annual rainfall and temperature in montane sub-tropical landscape of Upper Shillong during 2005–2008

The annual rainfall varies widely in the state ranging from about 60 mm in the month of December to over 10,070 mm in the month of June. Highest amount of rainfall is received during May to September.

The meteorological data were collected from the nearest meteorological station of the district agriculture office situated at Nongpoh, the headquarters of Ri-Bhoi district for the tropical landscape and from Upper Shillong meteorological station for the montane sub-tropical landscape. The total average annual rainfall during the year 2005-2008 was 1662mm for the tropical landscape and 2466mm for the montane sub-tropical landscape. The mean monthly temperature varied from a maximum of 35°C in the month of August to a minimum of 13.3°C in January in the tropical landscape and a maximum temperature of 23.3°C in August and minimum temperature of 1.8°C was recorded in December in the montane sub-tropical landscape (Fig. 3.2 & 3.3).

Soil

The soils of Meghalaya are derived from the underlying gneisses, schists and granites. They have been grouped under latosol (Oxisol) type (Pascoe 1950). In general, soils are highly leached, acidic and deficient in essential nutrients like nitrogen, phosphorus and potassium. The soils are acidic (pH 3 - 6.5) in nature. Acidity is attributed to the leaching of cations like calcium, magnesium and potassium from the soil due to high rainfall and undulating topography. Total Kjeldahl nitrogen content is low (ca. 0.15%) at degraded sites and fairly high (ca. 0.94%) in the undisturbed forests. Soils are deficient in available phosphorus (15 $\mu g\ g^{-1}$) and medium in exchangeable potassium (250 $\mu g\ g^{-1}$), and rich in organic carbon (1.62%). Soils are not suitable for intensive cultivation due to their poor base (35%) saturation (Singh 1996).

Vegetation

Forests

The forests of Meghalaya were classified by Kanjilal *et al.* (1934), Champion and Seth (1968), Balakrishnan (1981), Haridasan and Rao (1985), Rao and Hajara (1986) and Chauhan and Singh (1992). The forest cover of the state in 2003 was 16,988 km^2, or 75.7% of the total geographical area of the state (FSI 2005). The latest forest cover of the state in 2007 was 17321 sq. km, or 77.2% of the total geographical area (FSI 2009). There was a net gain of 116 km^2 forest cover between the two assessment years.

The tropical forest occurring below 1000 m elevation may be either evergreen or semi-evergreen type depending on the dominance of evergreen and deciduous trees in the canopy. The montane sub-tropical forest found above 1000 m is either broad-leaved or needle-leaved. Small pockets of montane sub-tropical evergreen broad-leaved forest are found where rainfall is relatively high and soil moisture condition remains favorable for most part of the year, while those areas which receive relatively less annual rainfall support semi-evergreen forest. Pine forests have developed as a stable secondary community on the disturbed evergreen and semi-evergreen montane sub-tropical broad-leaved forest sites, which are seasonally dry and nutrient-poor.

The primary tropical and montane sub-tropical forests of the state have been destroyed to a great extent by age-old tradition of shifting agriculture, which is extensively practiced in the state even today. As a result of this and other anthropogenic activities, extensive degradation of the forest has taken place in the state. The degraded forestlands support a variety of successional communities ranging from weed-dominated communities on recently abandoned Jhum fields to pine forest and grassland on frequently burnt and nutrient-deficient sites.

The major forest types of Meghalaya are tropical semi-evergreen, tropical moist deciduous, montane sub-tropical evergreen, montane sub-tropical semi-evergreen and montane sub-tropical pine forests.

Tropical semi-evergreen forest

This forest occupies the north-eastern and Northern slopes of the state, typically up to elevations of 1200m, where annual rainfall is 150-200 cm with a comparatively cooler winter. The numbers of species here are fewer than the evergreen zone. The deciduous species in the forest are *Callicarpa arborea, Careya arborea and Dillenia pentagyna.* There is a clear stratification of the trees in these forests.

Tropical moist deciduous forest

This forest type is represented by sal-bearing forest at low elevational areas of Garo hills, where annual rainfall is less than 150 cm. Along with *Shorea robusta*, other tree species found in the forest are *Bauhinia variegata, Calliandra sp., Callicarpa arborea, Cordia grandis, Dillenia scabrella, Dysoxylum binectariferum, Embelia floribunda, Lagerstroemia parviflora, Mallotus philippensis, Picrasma javanica, Schima wallichii, Sterculia villosa, Styrax serrulatum, Tectona grandis and Terminalia myriocarpa.* The under storey is composed of *Ardisia neriifolia*, *Clerodendrum viscosum*, *Digitaria sp.*, *Desmodium sp.*, *Eupatorium adenophorum*, *Gleichenia sp.*, *Melastoma malabathricum*, *Psychotria monticola*, *Pongamia sp.*, *Sabia purpurea*, and *Vandelia sp.* A distinct second storey is clearly observed in this type of forest.

Montane sub-tropical evergreen forest

It generally occurs above 1200 m a.s.l., where average annual rainfall ranges between 300 and 500 cm and temperature shows a noticeable difference between summer and winter season. The ground frost is common in December and January. The trees are generally short statured not exceeding >25 m height. Buttressed trunks and lianas are rare. Stratification is indistinct in the valleys, but it is clear at hilltops. The shrubby and herbaceous layers are clearly seen in the forest. Epiphytes, mosses and liverworts are abundant. The forest floor is spongy due to presence of thick litter layers and a dense network of fine roots.

Tall scattered trees of *Castanopsis sp., Betula sp., Engelhardtia spicata, Exbuklandia populnea, Ficus elastica, Lithocarpus elegans, Mangleitia insignis and Prunus nepalensis,* constitute the canopy layer of the forest. Occasionally, *Schima wallichii* is also seen. *Daphne involucrata*, *Helicia nilagirica*, *Ligustrum robustum, Michelia punduana*, *Quercus glauca*, *Symplocos racemosa, Viburnum foetidum*, *V. simonsii* and *Vernonia volkamerifolia* are plant species common in the sub-canopy layer. Tree ferns are common in the forest.

The common shrub species found in the forest include, *Ardisia sp*., *Baliospermum micranthum*, *Camellia caudate, Clerodendrum sp*., *Eurya japonica, Goniothalamus sesquiopedalis, Ixora subsessilis*, *Neillia thorsiflora*, *Psychotria sp*., *Sarcandra glabra*, *Sarcococca saligna*, *Saurauria sp.*, and members of Acanthaceae and Araliaceae.

The forest floor is covered with mosses such as *Selaginella* and angiosperm species such as *Begonia palmata, Begonia rubro-venia, Didymocarpus punduana, Disporum sp*., *Elatostema rupestre, Senecio griffithii, Sonerilla sp*., *Impatiens sp*., and members of Araceae Commelianaceae and Zingiberaceae. *Clematis sp*., *Dioscorea sp*., *Melodinus sp*., *Smilax sp*., and species of Menispermaceae and Cucurbitaceae are common climbers in the forest.

Montane sub-tropical semi-evergreen forest

The altitudinal limits of distribution and climatic conditions prevailing in the montane sub-tropical semi-evergreen forest area are similar to those of evergreen forest. A transitional zone between tropical and montane sub-tropical forests is distinguishable at certain places between 1000-1400 m.

The common canopy (25 m height) species are *Engelhardtia spicata, Castanopsis indica, Sapindus rarak, Paramichelia baillionii, Elaeocarpus floribundus, Meliosma wallichii, Diospyros undulata, Ficus altissima* and *Vitex glabrata.* The sub-canopy in the forest is composed of small trees of *Vitex vestita, Quercus semicarpifolia, Casearia vareca, Micromelum integrimum, Photinia arguta, Symplocos cochinchinensis* and *Xylosma controversum.*

Lianas are less frequent and they are represented by *Mucuna macrocarpa, Tetrastigma obovatum* and *Celastrus campionii.* The shrub layer has fewer species. The common shrubs are *Crotalaria assamica, Boehmaria platyphylla, Capparis acutifolia, Lyonia ovalifolia, Randia griffithii, Mussaenda glabra, Desmodium sp.*, *Maesa tetrandra* and *Clerodendrum sp.* The herbaceous layer is sparse. The common flowering plants of this layer are *Pilea umbrosa, Galinsoga parviflora, Anisadenia khasiana, Curcuma sp.*, *Polygalla sp.*, *Acanthus leucostachys*, *Pouzolzia hirta, Hedychium sp.*, and many plants of Asteraceae.

Montane sub-tropical pine forest

The forest is confined to the central upland of Shillong plateau between ca., 1000-2000 m a.s.l. The climatic conditions are similar to those of evergreen and semi-evergreen forests. It occurs in Khasi and Jaintia hills above 800 m (either as pure or mixed stands) on nutrient-poor soil. The forest is exposed to annual winter fire when ground vegetation is almost completely dry. Besides annual fire, other biotic disturbances such as fuelwood collection, timber extraction and grazing are common in the forest. As a result of these activities, the forest has been fragmented into small patches.

The average height of pine trees ranges between 20 and 35 m, however, on degraded sites the height may be less. Few scattered trees of broad-leaved species are often associated with pine. A few small trees or large shrubs are found scattered in the forest forming the sub-canopy layer. Annual fire prevents establishment of shrubs and other woody elements. However, weeds and perennial grasses form dense undergrowth during monsoon.

Pine forest is very poor in tree species composition. At places it forms a mixed stand with *Schima wallichii, Prunus undulata, Prunus cerrasoides, Rhus javanica, Quercus dealbata, Q. glauca, Q. griffithii, Lyonia ovalifolia, Rhododendron arboreum, Alnus nepalensis* and *Exbucklandia populnea.* The shrubby undergrowth includes *Rubus ellipticus, R. khasianus, R. rugosus, Myrsine semiserrata, Osbeckia crinita, Desmodium* sp., *Eupatorium* sp., *Lantana camara* and *Bidens pilosa.*

Grassland

In Meghalaya grasslands are not climax type and have developed on degraded sites after the destruction of natural forests. The species of flowering plants encountered in such grasslands are *Arundinella bengalensis, Arundinella khasiana, Arundinella nepalensis, Imperata cylindrical and Setaria glauca.* Dabadghao and Shankarnarayan (1973) have given an account of the grassland communities found on the hills of Meghalaya. According to them, the principal species in the grasslands at 1200-1600m a.s.l. are *Arundinella khasiana, Dimeria fuscescens, Eragrostiella leioptera and Ischaemum barbatum*, while those found at 800-1200m a.s.l. have a mixture of species from the plains and hills. *Arundinella bengalensis* is the dominant species here. Above 1800m a.s.l. the grassland developed on an abandoned jhum field is dominated by *Eragrostiella leioptera* and the community is similar to that occurring at 1200-1600 m a.s.l.

CHAPTER IV

METHODS AND MATERIALS

The estimation of carbon pool in the tropical and montane sub-tropical forest ecosystem was undertaken through the measurement/estimation of the following components viz., soil organic carbon (SOC), microbial biomass carbon (MBC), herb and shrub biomass, litter pool, aboveground biomass (AGB) and belowground biomass (BGB). The estimation of carbon flux includes litterfall, carbon dioxide efflux (CO_2), and net primary productivity (NPP). The methods for each of the above components are described below.

FIELD SAMPLING

In each of the tropical and montane sub-tropical forest types, depending upon the availability 10-20 ha of forest area for each type was demarcated. Six permanent plots each of size 250 m x 20 m (0.5 ha) were demarcated randomly within each forest type. In each plot, all trees with $\geq$ 5cm DBH were tagged, measured and identified. The girth of each individual tree was measured. The tree species were identified with the help of regional flora (Haridasan and Rao, 1985-87; Joseph 1982; Kanjilal *et al.* 1934-40). The ASSAM herbarium at Botanical Survey of India, Shillong was consulted for confirmation. Frequency, density, basal area and IVI were calculated following Misra (1968). Diversity indices i.e. Shannon's diversity index, Simpson's dominance index, and α diversity were determined. For depicting tree population structure in different landscape element, all trees were grouped into eight diameter classes i.e. >5-10, >10-20, >20-30, >30-40, >50-60, >60-70, >70-80, >80-90, >90-100 and >100 cm. For the regenerating broad-leaved forest of tropical and montane sub-tropical landscape and regenerating pine forest of montane sub-tropical landscape where the tree diameter ranges were low, trees were grouped into 2-4 diameter classes.

SOIL SAMPLING

Soil samples were collected from each forest type at seasonal interval over a period of three years i.e. from 2005 – 2008. The four seasons were categorized as follows: May-September (summer/rainy), October-November (autumn), December-February (winter) and March-April (spring). Soils were collected from randomly located five points from each forest type using a steel auger (11.46 cm diameter) from the surface (0-10 cm) and subsurface (10-20 cm) layer. The soil samples so collected were mixed thoroughly to obtain one composite sample for each forest type. Fresh soil was used for analysis of soil moisture content, soil respiration and microbial biomass C. The remaining soil was air-dried and sieved through 2 mm sieve and stored for analysis of soil organic carbon, total kjeldahl nitrogen, available phosphorus and exchangeable potassium.

SOIL ANALYSIS

Bulk density and porosity were determined following Anderson and Ingram (1993). After clearing the soil surface, a 5 cm diameter thin-sheet metal tube of known weight and volume was inserted into the soil surface. The soil was excavated from around the tube and the soil beneath the tube bottom was cut. The excess soil was trimmed from the tube ends. The soil samples were dried at 105°C until constant weight. The bulk density and porosity were determined using the following formulae:

Bulk density (g/cm^3) = (W2 –W1) / V

Where, W2 and W1 = dry and fresh weight of soil, V = volume of the metal cube

Total porosity was calculated from bulk density of soil and the particle density (assuming it to be 2.65 g/cm^3 for most mineral soils).

Total porosity (%) = {1 – (bulk density / particle density)} x 100

Soil texture was determined by Bouyoucos hydrometer method following Allen *et al.* (1974) using calgon /sodium hexametaphosphate as a dissolving agent.

$$\text{Clay (\%)} = \frac{A\,(g^{-1}) \times 100}{(50 - \text{moisture wt})\,g} - 1$$

$$\text{Silt + clay (\%)} = \frac{A\,(g^{-1}) \times 100}{(50 - \text{moisture wt})\,g} - 1$$

Silt (%) = (silt + clay) (%) – clay (%)

Where A was hydrometer readings after 5 hours and B was hydrometer readings after 4 min 48 sec, (1 = calgon correction factor).

The soil textural class was determined using the soil texture triangle.

Water holding capacity of the soil was determined by Keen's cup method by using copper cups of 5.6 cm internal diameter and 1.6 cm height (Piper 1942). Soil moisture content was determined by taking 10 g fresh sieved soil (Allen *et al.* 1974). Soil pH was determined electrometrically using a digital pH meter (SYSTRONICS-335) in 1:2.5 w/v suspension of soil in deionised water (Anderson and Ingram, 1993). Additional soil samples were collected upto 1 m depth for studying soil organic carbon pool in different tropical and montane sub-tropical forest ecosystems during 2007-2008.

For relating SOC with various biological and chemical soil properties, microbial biomass carbon, available phosphorus, exchangeable potassium and TKN were estimated.

Soil Organic Carbon (SOC)

Soil organic carbon was determined by colorimetric method following Anderson and Ingram (1993). One gram of the air dried and grounded soil sample was added to 10 ml 5% potassium dichromate solution to completely wet the soil. To it 20 ml concentrated H_2SO_4 was added from a fast burette and the mixture was gently swirled. The soil solution was allowed to cool and then 50 ml barium chloride was added and allowed to stand overnight, so as to leave a clear supernatant solution. An aliquot of the supernatant solution was transferred into a colorimetric cuvette and optical density was measured at 600 nm.

% organic carbon = (K x 0.1) / (W x 0.74)

Where, K = carbon concentration in the sample obtained from standard graph and W= weight of soil.

Soil organic matter was obtained by multiplying the soil organic carbon content by 1.724 assuming that the SOM contains 58% carbon (Allen *et al.* 1974).

The carbon stock density of soil organic carbon is calculated as (Pearson *et al.* 2007)

SOC = %C x BD x d

where,

SOC = soil organic carbon stock per unit area (Mg C ha^{-1})

BD = soil bulk density [g cm^{-3}],

d= soil depth [cm] and

%C = carbon concentration [%].

Total Kjeldahl Nitrogen (TKN)

Total kjeldahl nitrogen was determined by Kjeldahl digestion, distillation and titration method (Jackson 1973) by digesting 1g of air-dried soil samples at 350°C with concentrated sulphuric acid using kjeltabs (TECATOR) as catalysts, on a block digester for two hours. Distillation and titration were done simultaneously in a TECATOR KJELTEC AUTO 1030 ANALYSER. 10 ml of sample solution and 10 ml of 40% NaOH was allowed to distill and collected in 5 ml 1% boric acid indicator. The collected distillate was titrated against M/140 N HCL until endpoint was achieved.

$$N (\%) = \frac{T - \text{blank} \times \text{sample volume}}{10^2 \times \text{aliquot volume} \times \text{sample weight}}$$

Available Phosphorus (P)

Available phosphorus was determined after extracting soil phosphorus in 0.5 M sodium bicarbonate solution by ammonium-molybdate blue method (Allen *et al.* 1974). 10 ml of the sample solution was taken in a 50 ml volumetric flask and was diluted upto two thirds full. To it 2 ml of ammonium chloride reagent was added and mixed properly. 2 ml of stannous chloride reagent was added and mixed uniformly and diluted to volume and left for 30 minutes. The optical density was measured at 700 nm. The concentration of Phosphorus in the aliquot was calculated from the standard curve.

P (%) = C (mg) x solution volume (ml) / 10 x aliquot (ml) x sample wt (g)

Where, C = mg P obtained from standard graph

Exchangeable Potassium

Exchangeable Potassium was determined by using flame photometric method (Allen *et al.* 1973). The soil samples were extracted with 1 M ammonium acetate solution (pH 7.0) and the concentration of Potassium in ppm was determined using a flame photometer with K filter.

$$K\ (\%) = \frac{C\ (\text{ppm}) \times \text{solution volume (ml)}}{10^4 \times \text{sample wt (g)}}$$

Where, C = concentration of K in ppm.

Microbial biomass carbon (MBC)

MBC was determined by chloroform fumigation extraction method (Vance *et al.* 1987). Six sub-samples of 10 g ± 0.01 g each were drawn from each composite sample, three of them were fumigated by saturating with 10 ml (alcohol-free) chloroform liquid and kept for 24 hours and the remaining three were not fumigated. After fumigation, chloroform was removed from the samples by evaporation. Microbial biomass was extracted from both fumigated and non-fumigated samples with 50 ml of 0.5 M K_2SO_4 by shaking for 30 minutes. The extracts were filtered through Whatman filter paper No. 42 and the filtrates were used for the determination of microbial biomass carbon and nitrogen.

The organic C in the extracts of fumigated and non-fumigated soil samples was determined by digesting 4 ml filtered extract with 0.0667 M $K_2Cr_2O_7$ (1 ml) and 5 ml of H_2SO_4 (98 % acid) for 30 minutes. The digested sample was titrated with acidified ferrous ammonium sulphate solution using 0.3 ml (3 - 4 drops) of indicator (o-phenanthroline monohydrate and ferrous sulphate hexahydrate). The MBC was calculated as:

$$MBC = 2.64\ Ec$$

Where, Ec is the difference between the amount of organic C in the K_2SO_4 extract of fumigated and non-fumigated soils, both expressed as $\mu g\ g^{-1}$ dry soil and 2.64 is the relationship between biomass C as measured by fumigation incubation method and amount of C extracted by 0.5 M K_2SO_4 after chloroform treatment.

PLANT BIOMASS ESTIMATION METHOD

Estimation of carbon content in plant biomass

The oven-dried plant components were used for determination of carbon content. 1 g of oven dried material of different plant components was taken in silica crucible and allowed to heat at 550°C for 6 hours in a muffle furnace. The carbon content was calculated as 50% of ash free mass.

Estimation of herb and shrub biomass

The biomass of shrubs and herbs were estimated through harvest method following Misra (1968). Herbs were sampled from 10 plots of 1m x 1m size and shrubs from 5m x 5m plots at every three month interval for biomass and productivity estimation.

Developing tree biomass models

Estimation *Pinus kesiya* tree biomass in old-growth forest for model development

Forty trees of *Pinus kesiya* were selected in three forest stands of Laitkor reserve forest, Rait Laban reserve forest and Upper Shillong community forests, where trees were felled by the local people for construction purpose. The trees selected for Riat Laban reserved forest stand were from the adjacent community forest area, which is continuous with the reserve since felling of tree is banned inside the reserve. The trees selected in the two community forest stands were from the peripheral areas of the stands. Five to six trees were selected for each of the seven diameter classes of *Pinus kesiya* i.e. >5-10, >10-20, >20-30, >30-40, >40-50, >50-60 and 60-70 cm that represented minimum and maximum diameter range of the species in the forest. The DBH of the felled trees was measured. The age of the pine forest was determined by counting the annual growth rings in circular sections taken from the above mentioned 40 sample trees. The counting of the rings was done in the sections taken at

30 cm from the base of the tree. The mean value represented the age of the forest stand. The trees were separated into stem, branch, twig, needle, reproductive part, and root components, and the fresh weight of each component was taken. Three replicate samples of 2 kg each for each component were oven dried at 80°C till constant weight was achieved. For estimation of BGB, the roots of each cut tree were excavated as completely as possible and separated into fine roots (<2 mm diameter) and coarse roots (>2 mm diameter). Both the coarse and fine roots of each cut tree were weighed in the field. The portion of the tree stump that remains underground was treated as a part of the coarse root. The root samples in triplicate were brought to the laboratory and oven dried at 80°C till constant weight was achieved.

Estimation of *Pinus kesiya* tree biomass in regenerating forest for model development

Thirteen trees were randomly selected for felling from the regenerating forest stand. The trees were harvested and sorted into the different components as done in old growth forest stand. Four to five trees were selected for each of the three diameter classes of *Pinus kesiya* i.e. <10, >10-15 and >15-20 cm that represented minimum and maximum diameter range of the species in the forest. The DBH of the felled trees was measured and the biomass for each component was estimated following the methods used in old growth forest stand.

Estimation of regenerating broad-leaved tree biomass for model development

Twenty eight trees of broad-leaved forest stand at Upper Shillong were randomly selected for felling. The selected trees were harvested and sorted into different components as done in old growth *Pinus kesiya* forest. Fourteen trees were selected for each of the two diameter classes of i.e. <10, >10-15 cm that represented minimum and maximum diameter range of the species in the forest. The DBH of the felled trees was measured. The biomass for each component was estimated following the methods used in old growth *P. kesiya* forest stand.

Estimation of *Dendrocalamus hamiltonii* culm biomass for model development

Fifty four culms of *Dendrocallamus hamiltonii* in Nongkhyllem wildlife sanctuary were selected for felling from the mixed bamboo forest stands. Twenty seven culms were selected from each of the two diameter classes of i.e. <10, >10-15 cm that represented minimum and maximum diameter range of the species in the bamboo forest. All these 54 culms were randomly selected from 20 clumps and a maximum of 3 culms per clump were harvested. The harvested bamboos were sorted into different components i.e. culm, leaves, branches/twigs and rhizomes. The DBH of the felled culms was measured. After felling, the culms were subdivided into leaves, branches, culm and rhizome. Fresh weight of the components was taken in the field and sub-samples from each component were brought to the laboratory in plastic bags. The sub-samples were then oven dried at 80°C until constant weight. The dry weight data were log transformed and allometric model was developed which was then used to determine the biomass of the bamboo culms in the mixed bamboo forest based on the DBH data collected from each of the bamboo culms.

Estimation of *Teinostachyum dullooa* culm biomass for model development

Twenty six culms of *Teinostachyum dullooa* in Nongkhyllem wildlife sanctuary were selected for felling from the bamboo forest stands. Thirteen culms were selected from each of the two diameter classes of i.e. <10, >10-15 cm that represented minimum and maximum diameter range of the species in the bamboo forest stand. All these 26 culms were randomly selected from 8 clumps and a maximum of 4 culms per clump were harvested. The harvested bamboos were sorted into different components i.e. culm, leaves, branches/twigs and rhizomes. The DBH of the felled culms was measured. The processing of the harvested bamboo components follows the standard method used for biomass estimation of *Dendrocalamus hamiltonii*.

Development and evaluation of allometric models

Regression models were developed considering tree/bamboo DBH as independent variable and stem, branch, twig, needle, reproductive part, root, total aboveground and total tree biomass as dependent variables. The DBH and dry weight values were log transformed and nonlinear regression models were fitted for different tree components as well as for total tree biomass. For selecting the best-fit models, the coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) of the allometric equations were compared with those of existing models developed by earlier workers (Brown 1997; Delrio *et al.* 2008; Ter-Mikaelian and Korzukhin, 1997).

Several models were evaluated for goodness of fit for *Pinus kesiya* tree and bamboo biomass estimation. The models for both AGB and BGB were of the form:

$\text{Log}(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$

These models were developed based on several models published by earlier workers and comparing the respective goodness of fits (Table 4.1). For determining broad-leaved tree biomass, the models developed by Chambers *et al.* (2001a) for AGB and Cairns *et al.* (1997) for BGB estimation were used. The allometric model fitted for broadleaved tree aboveground component (Y_1) was: $\ln(Y_1) = [- 0.37 + 0.333*\ln(D) + 0.933*\ln(D)^2 - 0.122*\ln(D)]^3$ and that for belowground component (Y_2) was: $Y_2 = \exp [- 1.0587 + 0.8836*\ln(AGB)]$. These two models were selected based on R^2, SD, SSE, MSE, and RMSE values.

Estimation of total biomass and carbon of the forest

The total forest biomass was estimated by adding the biomass of the following components: (i) herbs and shrubs (ii) litter, (iii) tree aboveground biomass and (iv) tree belowground biomass. The mean biomass values obtained from the 6 permanent plots each in tropical and montane sub-tropical landscapes were presented.

ESTIMATION OF CARBON FLUX

Litterfall

Litterfall was estimated at seasonal interval over a period of three years from summer, 2005 to spring, 2008 (n=60). Prior to the commencement of sampling, 1m x 1 m x 0.15 m litter traps were randomly placed in each permanent plot during spring season of 2005. The litter traps were made of bamboo culms to check litter loss through runoff water during rainy season. Five litter traps were placed within each of the different tropical and montane sub-tropical forest type. The litter present in each litter trap were collected and brought to the laboratory. The samples were washed under a fine jet of water to remove adhered soil particles. The collected litter components were segregated into five fractions viz., leaf, twig, branch, bark and reproductive parts and oven dried at 80°C until constant weight. Litter biomass and annual litter production values were based on the data recorded from summer 2005 to spring 2008.

In situ leaf litter decomposition

Leaf litter decomposition in each forest type was studied using litterbag technique (Gilbert and Bocock, 1960). The nylon mesh (2 mm) litterbags (15 x 15 cm) were used for the study. Newly senesced leaves from each of the forest type were collected during peak litterfall period (March) and air-dried in the laboratory. 10 g of the air-dried materials were placed in each litter bag and stitched with nylon thread. For each landscape element 40 litter bags were prepared. The bags containing leaf litter were randomly placed on the last week of May, 2005on the forest floor after removal of organic layer from the ground. At each sampling season, five litter bags were retrieved at 90, 180, 270 and 360 days interval. The litter bags were brought to the laboratory and gently washed to remove any adhering soil particles and oven dried at 80°C till constant weight for determination of dry mass.

Litter turnover

Annual decomposition rate constant (k) was calculated from the data on the percent mass remaining using the negative exponential decay model (Olson 1963)

$k = \ln (x/x_o)/t$

Where, x_o=initial dry weight; x=weight remaining at the end of the investigation and t is the time in years. The time required for 50 % (t_{50}) and 99 % (t_{99}) decay was calculated as $t_{50} = 0.693/k$ and $t_{99} = 5/k$.

Litter turnover rate (k_L) was calculated by using the mathematical model of Reiners and Reiners (1970)

$k_L = L/X_L$

where, L= annual litterfall (kg ha^{-1} yr^{-1}) and X_L = mean annual standing crop (kg ha^{-1})

Turnover time (T) was calculated as a reciprocal of turnover rate

$T = 1/k_L$, where, T = time in year.

CO_2 efflux

The soil CO_2 efflux rate was determined following the method of McFadyen (1970). One kg of fresh forest soil samples were taken in rectangular shaped glass jars having 20 ml of 0.1N Potassium hydroxide (KOH). The soil samples were kept for 24 hours. The carbon dioxide fixed by the alkali was titrated against 0.1 N Hydrochloric acid (HCL) using phenolphthalein as an indicator.

The CO_2 evolved was expressed on per hour basis.

CO_2 mg/kg/hr = (B-V) *N*E

where, B = Volume of acid titrated without soil sample

V = Volume of acid titrated with soil sample

N = Normality of acid

E = Equivalent weight of CO_2 (E=22)

ESTIMATION OF NPP

The NPP of the forest was determined from the NPP estimates for each component i.e. tree, shrub, herb and litter in the six permanent sample plots. The NPP was estimated for all the components for three consecutive years i.e. 2005 - 2008, and the mean values were presented. Only the positive increments in each of the component were taken into account. As already mentioned, the biomass for the different forest types were estimated by applying the allometric equations developed in this study, and for other broad-leaved tree species it was estimated using the equations of Chambers *et al.* (2001a) for AGB and Cairns *et al.* (1997) for BGB. The standing tree biomass component of NPP was estimated by subtracting biomass estimated in July, 2005 from that of June, 2006 and biomass in July, 2006 from that of June, 2007, and biomass in July, 2007 from that of June, 2008, respectively and the mean biomass were presented. The aboveground NPP was determined by summing the tree biomass component of NPP and annual litter production measured during the same time interval (Kira and Shidei, 1967). The annual root production was measured by sampling roots using a soil augar in four seasons each year. The roots were washed and segregated into fine and coarse roots and the biomass was determined for each component after oven drying the samples at 80°C till constant weight was achieved. The annual root production was measured by summing up the positive increments in live root biomass and concurrent positive increment in the dead root biomass during the successive samplings (Persson 1978). The NPP for shrubs and herbs was estimated using the biomass data for the same time interval as standing tree biomass component.

Table 4.1 Allometric models used for estimation of biomass for broad-leaved tree species in the tropical and montane sub-tropical forest ecosystem

Model No	Regression equation	R^2	Reference
AGB1	$Y= 42.69 - 12.800(D) + 1.242(D^2)$	0.87	FAO. 3.2.3 (1997)
AGB2	$Y= \exp \{- 2.134 + 2.530*\ln (D)\}$	0.80	FAO. 3.2.4 (1997)
AGB3	$Y= 21.297 - 6.953 (D) + 0.740(D^2)$	0.87	FAO. 3.2.5 (1997)
AGB4	$Y= \exp [- 3.114 + 0.972*\ln (D^2H)]$	0.87	Brown *et al.* (1989) (>5 cm)
AGB5	$Y= \exp [- 2.409 + 0.952*\ln (pD^2H)]$	0.88	Brown *et al.* (1989) (>10 cm)
AGB6	$Y= \exp (- 2.00 + 2.42) \ln (D)$	0.82	Chave *et al.* (2001)
AGB7	$Y= \exp [- 0.37 + 0.33*\ln (D) + 0.933*\ln(D)^2 - 0.122* \ln(D)^3]$	0.93	Chambers *et al.* (2001a)
AGB8	$Y= 1.276 + 0.034 (D^2*H)$	0.86	Brown and Iverson 1992
AGB9	$Y= 38.4908 - 11.7883 (D) + 1.1926 D^2$	0.88	Brown *et al.* (1989)
AGB10	$Y= 13.2579 - 4.8945 (D) + 0.6713 (D^2)$	0.86	Brown *et al.* (1989)
BGB1	$Y=\exp [- 1.0587 + 0.8836* \ln (AGB)]$	0.83	Cairns *et al.* (1997)

CHAPTER V

CARBON SEQUESTRATION IN TROPICAL LANDSCAPE

INTRODUCTION

Tropical forests harbor most of the world's biodiversity (CBD 2011) and store large amount of carbon (Miller *et al.* 2004), yet very little is known about the levels of carbon storage in many tropical forests (Baishya *et al.* 2009). The rates of carbon accumulation by different forest types provide empirical evidence to the fact that that the tropical landscape serves as sinks of atmospheric carbon. The estimates of carbon stock are also important for scientific and management issues such as forest primary productivity and nutrient cycling. The estimation of carbon in tropical forest landscape includes C in soil, microbial biomass, litter, aboveground and belowground biomass compartments.

Soil carbon (C) pools affect productivity and sustainability of forest ecosystems. Soil C content impacts overall soil properties, nutrient cycling and consequently have a critical role in global C cycling. While the physical properties of soil play an important role in determining the extent of biological processes in the soil (De Vos *et al.* 1994), the chemical characteristics determine the overall soil quality (Hassink 1997). Therefore, soil physico-chemical characteristics such as soil temperature, moisture, bulk density, water holding capacity, pH, total organic C, N and P have a bearing on overall carbon cycle through influencing soil microbial populations and their activities, and regulating the productivity through impacting uptake of water and nutrients by roots.

The forest soil C pools form a large and dynamic reservoir of C, and work as a potential sink for atmospheric CO_2. Due to the importance of soil C pools in nutrient cycling and their role in global C balance, there has long been an interest in understanding the effect of forest soil management on soil C pools (Ussiri and Johnson, 2007). Soil organic C pools consist of various fractions varying in degree of decomposition, recalcitrance, and turnover rate. Forest management practices affect these fractions differently (Ghani *et al.* 2003).

The soil microbial biomass constitutes a transformation matrix for all natural organic materials in the soil and acts as a labile reservoir of plant available nutrients (Lajtha, 1997; Seely and Zogg *et al.* 2000). It increases with forest productivity (Myrold *et al.* 1989) and is influenced by forest management (Bauhus and Barthel, 1995). Tree species directly affect soil microbial communities through variation in litter quality, root activities, complexity and amount of organic inputs into the soil (Hattenschwiler 2005). Measurement of microbial biomass has been used in studies on carbon flow, nutrients cycling and plant productivity (Voroney *et al.* 1989). The role of microbial biomass in improving the soil fertility, primary production, and organic matter decomposition has been studied by Parkinson and Coleman (1991).

The net primary production of a forest ecosystem returns to the soil environment as plant litter and subsequently enters the decomposition pathway (Swift *et al.* 1979). Litter production and decomposition play an important role in maintaining the fertility of forest soils. Litter originating from both aboveground and belowground compartments is the major pathway of supply of energy and N to soil in most terrestrial types (Swift *et al.* 1981). Management practices can cause drastic changes in litter production by modifying species composition and productivity. Climate change may affect litterfall through changing tree phenology and tree species, which are often consequences of changing precipitation pattern and mean annual temperature (Condit *et al.* 1996). The activities are influenced by numerous factors such as the chemical decomposition and physical structure of the detritus and the climatic factors such as temperature, humidity, soil aeration and pH (Couteaux *et al.* 1995).

Most litter production studies have been carried out in the temperate forests. Very few studies were conducted in the tropical forests (Bray and Gorham, 1964; Singh 1980). Litter flux and the associated elements released through litter decay play a fundamental role in maintaining the sustainability of natural and plantation forest types. Soil C sequestration is another important function associated with litter dynamics (Lemma *et al.* 2007). Elevated atmospheric CO_2 have increased litterfall and productivity in several forest types (Allen *et al.* 2000; DeLucia *et al.* 1999; Finzi *et al.* 2001; Schlesinger and Lichter, 2001; Zak *et al.* 2003). Decomposition of forest litter is the primary means of transferring nutrients into forms available for plant uptake, and is one of the most crucial processes in the biogeochemical cycle of the forest types (DeCatanzaro and Kimmins, 1985).

Respiration is one of the largest and most important carbon dioxide (CO_2) fluxes in terrestrial ecosystems (Valentini *et al.* 2000), with many components such as CO_2 efflux from soil, leaves, stems, and branches. Soil respiration has been shown to make up 55–85% of the whole ecosystem respiration in different forest types (Davidson *et al.* 2006; Law *et al.* 1999; Pilegaard *et al.* 2001). Because of growing concern about the global carbon budget, it is essential to understand how soil respiration fluxes contribute to the overall exchange of carbon between vegetation and atmosphere. Soil respiration is the amount of carbon dioxide produced due to metabolic activities of microorganisms. Most of the differences in soil respiration can be attributed to the differences in microbial activity. The production of biomass and accumulation of organic matter in soil affect soil microbial populations, thus increasing the potential for soil respiration (Aghasi *et al.* 2011).

Most studies on NPP estimation consider only the increment in AGB and litterfall and completely ignore the belowground component. For estimating ecosystem level NPP of a forest, time series biomass data for tree, shrub, herb and litter components are pre-requisites. Cairns *et al.* (1997) argued that the approach of allometric modelling should be more realistic than root/shoot ratio for estimating tree belowground biomass (BGB). The carbon pool of a forest ecosystem varies with age (Clark *et al.* 2004; Kurz and Apps, 1995). While young and middle-aged forest stands act as active carbon sinks (Valentini *et al.* 2000), old stands are moderate to small C sinks or even C sources depending on the forest type and species composition (Desai *et al.* 2005; Knohl *et al.* 2003; Law *et al.* 2003; Malhi *et al.* 1999). Most NPP studies world-wide have been carried out in relatively younger stands and data on carbon content and NPP in old-growth forests are limited (Delrio *et al.* 2008). However, such data for different components other than tree are not available for the tropical forests of the region and as such works on total ecosystem level NPP estimation are limited.

In view of the carbon data gap in the tropical landscapes, the present study analyzes the seasonal and spatial distribution pattern of soil organic carbon, microbial biomass carbon, litterfall, aboveground biomass, belowground biomass, ecosystem level biomass and carbon, and ecosystem level NPP in different tropical forest types. We studied pools and fluxes of C in five tropical forest types of Meghalaya *viz.*, old-growth broad-leaved, regenerating broad-leaved, teak plantation, sal plantation and mixed bamboo forests. Some selected soil physical and chemical parameters were also studied to assess their impact on carbon pools and fluxes and vice-versa.

RESULTS

The tropical landscapes had five forest types. These are (1) old-growth broad-leaved forest, (2) regenerating broad-leaved forest, (3) teak plantation, (4) sal plantation, and (5) mixed bamboo forest.

Tree species composition and community characteristics

Old-growth broadleaved forest

Total number of tree species was 94 and *Schima wallichii* was the dominant species (IVI 25.88) and *Polyalthia jenkinsii* and *Diospyros variegate* (IVI 22.68 and 22.24, respectively) were the co-dominant species. The basal area of trees in this forest was 65.13 m^2 ha^{-1}. *Schima wallichii* had the highest basal area of 12.52 m^2 ha^{-1}. *Polyalthia jenkinsii* had the highest density in the forest with 64 individuals ha^{-1}. The total density of the forest was 1090 individual ha^{-1}. Based on A/F ratio, 92.6 % of total tree species exhibited clumped distribution and 7.4% exhibited random distribution pattern (Table 5.1).

Table 5.1 Tree species composition, density, frequency, basal area, IVI and A/F ratio in tropical old-growth broad-leaved forest of Nongkhyllem wildlife sanctuary

Sl. No.	Species	Density	Frequency	Basal Area m^2 ha^{-1}	IVI	A/F
1	*Acacia concinna* (Willd.) DC.	8	6	0.03	1.57	0.22
2	*Actinodaphne obovata* (Nees) Blume	10	8	0.10	2.13	0.16
3	*Albizia odoratissima* (L.f.) Benth.	4	4	0.28	1.33	0.25
4	*Albizia procera* (Roxb.) Benth.	4	4	0.19	1.19	0.25
5	*Alseodaphne khasyana* (Meissn.) Kostern	6	6	1.70	3.95	0.17
6	*Alstonia scholaris* R.Br.	8	8	0.77	2.98	0.13
7	*Amoora wallichii* King	8	6	0.87	2.87	0.22
8	*Aphanamixis wallichii* (King) Haridasan & R.R.Rao	14	4	3.38	7.01	0.88
9	*Artocarpus chaplasha* Roxb.	4	2	0.59	1.54	1.00
10	*Bauhinia purpurea* Wall.	2	2	0.10	0.61	0.50
11	*Beilschmiedia assamica* Meisn.	30	18	0.46	5.84	0.09
12	*Sterculia villosa* Roxb.	2	2	0.52	1.24	0.50
13	*Butea parviflora* Roxb.	2	2	0.02	0.48	0.50
14	*Caesalpinia crista* L.	4	4	0.05	0.97	0.25
15	*Callicarpa arborea* Roxb.	2	2	0.03	0.49	0.50

16	*Calophyllum polyanthum* Wall.	4	4	1.96	3.90	0.25
17	*Casearia glomerata* Roxb.	14	12	2.25	6.33	0.10
18	*Castanopsis indica* A.DC.	14	12	3.45	8.16	0.10
19	*Castanopsis tribuloides* A.DC.	6	6	0.01	1.37	0.17
20	*Celastrus championi* Benth.	6	4	0.05	1.16	0.38
21	*Celastrus paniculatus* Willd.	4	4	0.02	0.92	0.25
22	*Chonemorpha fragrans* Alston	2	2	0.02	0.48	0.50
23	*Cinnamomum bejolghota* Sweet	4	2	0.01	0.65	1.00
24	*Cinnamomum pauciflorum* Nees	2	2	0.70	1.53	0.50
25	*Combretum punctatum* A.Rich.	10	4	0.12	1.63	0.63
26	*Combretum roxburghii* G.Don	6	4	0.02	1.11	0.38
27	*Cyathostemma argenteum* (Blume) J. Sincl.	8	4	0.02	1.29	0.50
28	*Dillenia indica* L.	4	4	0.50	1.66	0.25
29	*Dillenia pentagyna* Roxb.	22	18	0.96	5.87	0.07
30	*Diospyros variegata* Kurz	120	62	1.97	22.24	0.03
31	*Drimycarpus racemosus* Hook.f.	2	2	0.00	0.45	0.50
32	*Duabanga grandiflora* Walp.	6	6	0.92	2.75	0.17
33	*Dysoxylum binectariferum* Hiern.	6	6	0.08	1.47	0.17
34	*Dysoxylum gobara* (Buch.-Ham.) Merrill	36	32	1.18	9.34	0.04
35	*Elaeocarpus tectorius* Poir	58	44	1.41	13.31	0.03
36	*Elaeocarpus aristatus* Roxb.	12	10	0.19	2.72	0.12
37	*Entada phaseoloides* Merrill	4	4	0.02	0.92	0.25
38	*Eugenia kurzii* Duthie	4	4	0.17	1.16	0.25
39	*Eugenia tetragona* Wight	8	6	0.03	1.58	0.22
40	*Eurya acuminata* DC.	2	2	0.01	0.46	0.50
41	*Ficus altissima* Blume	2	2	0.36	1.00	0.50
42	*Ficus concinna* Miq.	2	2	0.33	0.95	0.50
43	*Ficus hirta* Vahl	2	2	0.01	0.46	0.50
44	*Ficus virens* Ait.	4	4	0.66	1.90	0.25
45	*Fissistigma polyanthum* (Hook.f. & Thomson) Merr.	10	10	0.03	2.28	0.10
46	*Garcinia cowa* Choisy	14	6	0.16	2.32	0.39
47	*Garcinia paniculata*Roxb.	2	2	0.01	0.46	0.50
48	*Glochidion hirsutum* Voigt	2	2	0.02	0.48	0.50
49	*Gmelina arborea* Roxb.	6	4	1.23	2.97	0.38
50	*Goniothalamus simonsii* Hook.f. & Thomson	32	6	0.09	3.87	0.89
51	*Grewia disperma* Rottl. ex Spreng	4	4	0.03	0.94	0.25
52	*Hibiscus macrophyllus* Roxb.	6	4	0.63	2.05	0.38
53	*Knema linifolia* (Roxb.) Warb.	12	12	0.12	2.87	0.08
54	*Lasianthus lucidus* Blume	2	2	0.01	0.46	0.50
55	*Lagerstroemia parviflora* Roxb.	4	4	0.42	1.54	0.25
56	*Leea alata* Edgew.	90	52	0.89	16.50	0.03
57	*Lithocarpus elegans* (Blume) Hatus	2	2	0.02	0.48	0.50
58	*Litsea khasyana* Meissn.	22	14	0.96	5.34	0.11
59	*Litsea laeta* Benth. & Hook.f.	6	6	0.14	1.55	0.17
61	*Litsea monopetala* (Roxb.) Pers.	16	14	0.13	3.53	0.08
62	*Litsea salicifolia* Hook.f.	2	2	0.01	0.46	0.50
63	*Macaranga denticulata* Muell.Arg.	10	10	0.59	3.15	0.10

64	*Macropanax undulatum* Seem	2	2	0.04	0.51	0.50
65	*Magnolia pterocarpa* Roxb	2	2	0.02	0.48	0.50
66	*Mesua ferrea* L	60	38	3.90	16.51	0.04
67	*Michelia oblonga* Wall	2	2	0.81	1.68	0.50
68	*Miliusa roxburghiana* Hook f & Thomson	4	2	0.31	1.11	1.00
69	*Millettia caudata* Baker	6	4	0.02	1.11	0.38
70	*Millettia cinerea* Benth	6	4	0.05	1.15	0.38
71	*Millettia pachycarpa* Benth	4	4	0.01	0.92	0.25
72	*Ostodes paniculata* Blume	4	4	0.02	0.93	0.25
73	*Phlogacanthus thyrsiflorus* Nees	2	2	0.01	0.46	0.50
74	*Polyalthia jenkinsii* Benth & Hook f	108	64	2.80	22.68	0.03
75	*Polyalthia simiarum* Benth & Hook f ex Hook f	6	6	0.39	1.94	0.17
76	*Pterospermum acerifolium* Benth	4	2	0.01	0.65	1.00
77	*Pycnarrhena planiflora* Hook f & Thomson	8	4	0.05	1.34	0.50
78	*Kydia calycina* Roxb	4	2	0.01	0.65	1.00
79	*Sapindus rarak* DC	2	2	0.01	0.46	0.50
80	*Sarcosperma griffithii* Hook f	26	16	1.07	6.15	0.10
81	*Schefflera hypoleuca* Harms	2	2	0.03	0.49	0.50
82	*Schima wallichii* Choisy	38	24	12.52	25.88	0.07
83	*Shorea robusta* A DC	8	8	1.79	4.53	0.13
84	*Spondias axillaris* Roxb	4	4	1.47	3.16	0.25
85	*Spondias pinnata* Kurz	2	2	0.70	1.53	0.50
86	*Syzygium diospyrifolium* (Wall ex Duthie) S N Mitra	2	2	0.01	0.46	0.50
87	*Terminalia chebula* Retz	4	2	0.32	1.13	1.00
88	*Terminalia citrina* Roxb ex Flem	4	4	0.24	1.27	0.25
89	*Terminalia myriocarpa* Heurck & Muell Arg	14	14	3.79	8.96	0.07
90	*Tetrastigma leucostaphylum* (Dennst) Alston	18	16	0.14	3.98	0.07
91	*Toona ciliata* M Roem	6	4	0.71	2.16	0.38
92	*Uvaria lurida* Dalz & Gibs	6	2	0.06	0.90	1.50
93	*Vitex peduncularis* Wall	6	6	1.85	4.19	0.17
94	*Wendlandia ligustrina* Wall	2	2	0.01	0.46	0.50
	Total	1090	756	65.18	300.00	

Regenerating broad-leaved forest

In the regenerating broad-leaved forest, the total number of tree species was 34 and *Actinodaphne obovata* (IVI 54.45) and *Artocarpus chaplasha* (IVI 52.36) were the dominant species. *Polyalthia jenkinsii* (IVI 25.29) was the co-dominant species. The total tree basal area of this forest was 9.92 m^2 ha^{-1}. *Actinodaphne obovata* had the highest basal area of 2.86 m^2 ha^{-1}. *Artocarpus chaplasha* had the highest density in the forest with 302 individuals ha^{-}

^{1}.The total density of the forest was 1728 individuals ha^{-1}. Based on A/F ratio, 91.2 % of total tree species exhibited clumped distribution and 8 8% exhibited random distribution pattern (Table 5.2)

Table 5.2 Tree species composition, density, frequency, basal area, IVI and A/F ratio in tropical regenerating broad-leaved forest of Nongkhyllem wildlife sanctuary

Sl. No.	Species	Density	Frequency	Basal Area m^2 ha^{-1}	IVI	A/F
1	*Actinodaphne obovata* (Nees) Blume	216	76	2 86	54 45	0 04
2	*Artocarpus chaplasha* Roxb	302	48	2 64	52 36	0 13
3	*Artocarpus heterophylla* Lam	102	2	0 19	8 14	25 50
4	*Bauhinia purpurea* Wall	6	4	0 02	1 29	0 38
5	*Bischofia javanica* Blume	40	16	0 03	5 34	0 16
6	*Callicarpa arborea* Roxb	102	32	0 03	11 71	0 10
7	*Casearia glomerata* Roxb	12	4	0 01	1 44	0 75
8	*Cassia fistula* L	98	36	0 07	12 59	0 08
9	*Celastrus paniculatus* Willd	12	6	0 00	1 75	0 33
10	*Cinnamomum tamala* T Nees and Eberm	50	34	0 25	11 26	0 04
11	*Dillenia indica* L	32	22	0 03	5 92	0 07
12	*Dillenia pentagyna* Roxb	18	10	0 01	2 88	0 18
13	*Diospyros variegata* Kurz	8	4	0 01	1 22	0 50
14	*Drimycarpus racemosus* Hook f	12	6	0 01	1 79	0 33
15	*Duabanga grandiflora* Walp	4	2	0 01	0 64	1 00
16	*Dysoxylum binectariferum* Hiern	6	4	0 01	1 11	0 38
17	*Elaeocarpus aristatus* Roxb	22	6	0 11	3 40	0 61
18	*Ficus altissima* Blume	4	2	0 01	0 65	1 00
19	*Ficus concinna* Miq	4	2	0 01	0 66	1 00
20	*Ficus fulva* Spreng	24	10	0 26	5 71	0 24
21	*Ficus hirta* Vahl	28	12	0 29	6 59	0 19
22	*Ficus maclelandi* Alli	4	2	0 01	0 65	1 00
23	*Ficus virens* Ait	22	10	0 06	3 57	0 22
24	*Hibiscus macrophyllus* Roxb	16	8	0 02	2 55	0 25
25	*Kydia calycina* Roxb	8	4	0 01	1 23	0 50
26	*Leea alata* Edgew	34	16	0 32	8 01	0 13
27	*Litsea laeta* Benth & Hook f	10	4	0 03	1 53	0 63
28	*Mesua ferrea* L	66	28	0 32	11 86	0 08
29	*Polyalthia jenkinsii* Benth & Hook f	184	70	0 25	25 29	0 04
30	*Schefflera hypoleuca* Harms	126	44	0 13	16 20	0 07
31	*Schima wallichii* Choisy	12	6	1 35	15 33	0 33
32	*Shorea robusta* A DC	36	20	0 28	8 35	0 09
33	*Syzygium diospyrifolium* (Wall Duthie) S N Mitra	80	14	0 27	9 81	0 41
34	*Tetrastigma leucostaphylum* (Dennst) Alston	28	14	0 07	4 73	0 14
	Total	1728	578	9 92	300 00	

Teak plantation forest

Total number of tree species was 33 and teak (*Tectona grandis*) was the dominant species (IVI 179.52) Being a plantation forest, *Dysoxylum gobara* was the second most important tree but with much lesser IVI (28 31) than *Tectona grandis* The total tree basal area of this forest was 38.31 m^2 ha^{-1}. *Tectona grandis* had the highest basal area of 36.02 m^2 ha^{-1} and density of 504 individuals ha^{-1}. The total density of the forest was 864 individuals ha^{-1} Based on A/F ratio, 93 9 % of total tree species exhibited clumped distribution and 6.1% exhibited random distribution pattern (Table 5.3).

Table 5.3 Tree species composition, density, frequency, basal area IVI and A/F ratio in tropical teak (*Tectona grandis*) plantation forest of Nongkhyllem wildlife sanctuary

Sl. No.	Species	Density	Frequency	Basal Area m^2 ha^{-1}	IVI	A/F
1	*Beilschmiedia assamica* Meisn	20	16	0 07	6 85	0 08
2	*Callicarpa arborea* Roxb	2	2	0 01	0 80	0 50
3	*Calophyllum polyanthum* Wall	14	12	0 16	5 31	0 10
4	*Castanopsis tribuloides* A DC	2	2	0 01	0 80	0 50
5	*Dillenia pentagyna* Roxb	14	12	0 05	5 01	0 10
6	*Diospyros variegata* Kurz	8	4	0 02	2 06	0 50
7	*Duabanga grandiflora* Walp	14	12	0 06	5 04	0 10
8	*Dysoxylum gobara* (Buch -Ham) Merrill	108	54	0 43	28 31	0 04
9	*Dysoxylum binectariferum* Hiern	18	16	0 08	6 65	0 07
10	*Elaeocarpus tectorius* Poir	6	4	0 03	1 86	0 38
11	*Eurya acuminata* DC	2	2	0 01	0 79	0 50
12	*Ficus altissima* Blume	2	2	0 00	0 79	0 50
13	*Fissistigma polyanthum* (Hook f & Thomson) Merr	4	4	0 01	1 58	0 25
14	*Grewia disperma* Rottl ex Spreng	2	2	0 01	0 79	0 50
15	*Lasianthus lucidus* Blume	2	2	0 00	0 79	0 50
16	*Leea alata* Edgew	2	2	0 00	0 73	0 50
17	*Litsea monopetala* (Roxb) Pers	40	30	0 10	13 05	0 04
18	*Macaranga denticulate* Muell Arg	4	4	0 01	1 58	0 25
19	*Maesa indica* Hook f	4	4	0 01	1 57	0 25
20	*Mesua ferrea* L	16	16	0 05	6 32	0 06
21	*Millettia pachycarpa* Benth	2	2	0 01	0 79	0 50
22	*Ostodes paniculata* Blume	10	6	0 10	3 05	0 28
23	*Polyalthia simiarum* Benth & Hook f ex Hook f	6	6	0 01	2 36	0 17
24	*Schefflera hypoleuca* Harms	2	2	0 02	0 81	0 50
25	*Schima wallichii* Choisy	2	2	0 01	0 80	0 50
26	*Sterculia villosa* Roxb	4	4	0 01	1 58	0 25
27	*Syzygium diospyrifolium* (Wall ex Duthie) S N Mitra	18	18	0 05	7 10	0 06

28	*Tectona grandis* L f	504	100	36 02	179 52	0 05
29	*Terminalia myriocarpa* Heurck & Muell Arg	16	12	0 89	7 43	0 11
30	*Toona ciliata* M Roem	4	2	0 04	1 11	1 00
31	*Uvaria lurida* Dalz & Gibs	8	8	0 03	3 17	0 13
32	*Vitex peduncularis* Wall	2	2	0 00	0 78	0 50
33	*Wendlandia ligustrina* Wall	2	2	0 00	0 79	0 50
	Total	864	368	38 31	300 00	

Sal plantation forest

In the sal plantation forest, the total number of tree species was 67 and *Shorea robusta* was the dominant species (IVI 121 22). *Schima wallichii* and *Litsea khasyana* (IVI 17.69 and 17.19, respectively) were the next two important species. The total tree basal area of this forest was 93.01 m^2 ha^{-1}. *Shorea robusta* had the highest basal area of 64.63 m^2 ha^{-1} and highest density of 386 individuals ha^{-1}. The total density of the forest was 1104 individuals ha^{-1} Based on A/F ratio, 91 % of total tree species exhibited clumped distribution and 9% exhibited random distribution pattern (Table 5 4).

Table 5.4 Tree species composition, density, frequency, basal area, IVI and A/F ratio in tropical sal (*Shorea robusta*) plantation forest of Nongkhyllem wildlife sanctuary

Sl. No.	Species	Density	Frequency	Basal area m^2 ha^{1}	IVI	A/F
1	*Acacia concinna* (Willd) DC	4	4	0 02	1 07	0 25
2	*Actinodaphne obovata* (Nees) Blume	6	6	0 02	1 59	0 17
3	*Albizia odoratissima* (L f) Benth	10	8	0 05	2 33	0 16
4	*Albizia procera* (Roxb) Benth	4	4	0 01	1 06	0 25
5	*Alseodaphne petiolaris* (Meisn) Hook f	2	2	0 01	0 54	0 50
6	*Antidesma acuminatum* Wight	2	2	0 00	0 53	0 50
7	*Antidesma nigricans* Tul	2	2	0 00	0 53	0 50
8	*Artocarpus chaplasha* Roxb	44	28	4 12	13 20	0 06
9	*Bauhinia purpurea* Wall	2	2	0 02	0 54	0 50
10	*Beilschmiedia assamica* Meisn	18	12	0 20	3 90	0 13
11	*Butea parviflora* Roxb	2	2	0 02	0 54	0 50
12	*Callicarpa longifolia* Lam	2	2	0 05	0 58	0 50
13	*Calophyllum polyanthum* Wall	16	8	0 11	2 94	0 25
14	*Cassia fistula* L	2	2	0 01	0 54	0 50
15	*Castanopsis tribuloides* A DC	14	8	1 11	3 84	0 22
16	*Celastrus championi* Benth	2	2	0 01	0 53	0 50

17	*Celastrus paniculatus* Willd	8	4	0.03	1.45	0.50
18	*Combretum dasystachyum* Kurz	2	2	0.00	0.53	0.50
19	*Combretum roxburghii* G Don	2	2	0.00	0.53	0.50
20	Sp A Dieng pessia (Local name)	2	2	0.03	0.55	0.50
21	Sp B Dieng rymmai (Local name)	2	2	0.10	0.63	0.50
22	Sp C Dieng tyrsing (Local name)	2	2	0.22	0.76	0.50
23	*Dillenia indica* L	10	8	0.04	2.32	0.16
24	*Dillenia pentagyna* Roxb	8	6	0.50	2.28	0.22
25	*Diospyros variegata* Kurz	34	24	0.15	7.35	0.06
26	*Dysoxylum binectariferum* Hiern	18	16	0.05	4.42	0.07
27	*Dysoxylum gobara* (Buch -Ham) Merrill	10	8	0.76	3.09	0.16
28	*Elaeocarpus aristatus* Roxb	38	24	2.40	10.13	0.07
29	*Elaeocarpus tectorius* Poir	44	30	0.32	9.47	0.05
30	*Eurya acuminata* DC	20	12	0.16	4.03	0.14
31	*Ficus hirta* Vahl	2	2	0.00	0.53	0.50
32	*Ficus squamosa* Roxb	2	2	0.67	1.24	0.50
33	*Ficus virens* Ait	2	2	2.43	3.14	0.50
34	*Fissistigma polyanthum* (Hook f & Thomson) Merr	4	4	0.02	1.07	0.25
35	*Gmelina arborea* Roxb	2	2	0.55	1.12	0.50
36	*Hibiscus macrophyllus* Roxb	2	2	0.06	0.58	0.50
37	*Knema linifolia* (Roxb) Warb	14	10	0.05	3.03	0.14
38	*Kydia calycina* Roxb	2	2	0.01	0.53	0.50
39	*Lagerstroemia parviflora* Roxb	8	6	1.89	3.79	0.22
40	*Leea alata* Edgew	4	4	0.01	1.06	0.25
41	*Acacia caesia* (L) Willd	2	2	0.01	0.54	0.50
42	*Litsea khasyana* Meissn	102	44	0.39	17.19	0.05
43	*Litsea meissneri* Hook f	24	14	0.07	4.65	0.12
44	*Litsea monopetala* (Roxb) Pers	16	8	0.08	2.90	0.25
45	*Macropanax dispermum* Kuntze	6	2	0.01	0.90	1.50
46	*Maesa montana* A DC	8	6	0.01	1.77	0.22
47	*Mesua ferrea* L	24	12	0.12	4.36	0.17
48	*Mesua floribunda* (Wall) Kosterm	2	2	0.01	0.53	0.50
49	*Mezoneurum cucullatum* Wight & Arn	8	8	0.02	2.11	0.13
50	*Millettia pachycarpa* Benth	2	2	0.01	0.53	0.50
51	*Ostodes paniculata* Blume	4	4	0.01	1.06	0.25
52	*Persea villosa* (Roxb) Kosterm	2	2	0.00	0.53	0.50
53	*Polyalthia jenkinsii* Benth & Hook f	40	30	0.78	9.60	0.04
54	*Polyalthia simiarum* Benth & Hook f ex Hook f	14	12	0.15	3.48	0.10
55	*Schefflera hypoleuca* Harms	2	2	0.02	0.54	0.50
56	*Schefflera wallichiana* Harms	2	2	0.01	0.54	0.50
57	*Schima wallichii* Choisy	46	34	7.17	17.69	0.04
58	*Shorea robusta* Gaertn f	386	98	64.63	121.22	0.04
59	*Spondias pinnata* Kurz	12	8	1.34	3.90	0.19
61	*Sterculia coccinea* Jack	4	2	0.03	0.73	1.00
62	*Stereospermum chenoloides* DC	2	2	0.17	0.70	0.50
63	*Syzygium praecox* (Roxb)	2	2	0.01	0.53	0.50

	Rathakr. & N C Nair					
64	*Terminalia citrina* Roxb ex Flem	2	2	0.01	0.53	0.50
65	*Terminalia myriocarpa* Heurck & Muell Arg	10	6	1.73	3.79	0.28
66	*Trevesia palmata* Vis	2	2	0.00	0.53	0.50
67	*Uvaria lurida* Dalz & Gibs	4	2	0.02	0.73	1.00
	Total	1104	584	93.01	299.46	

Mixed bamboo forest

In the mixed bamboo forest, *Dendrocalamus hamiltonii* and *Teinostachyum dullooa* were the two bamboo species. *Dendrocalamus hamiltonii* was the dominant bamboo species (IVI 239) and had 180 clumps ha^{-1} with density of 3420 culms ha^{-1} whereas, *Teinostachyum dullooa* was the co-dominant (IVI 61) and had 44 clumps ha^{-1} with density of 1524 culm ha^{-1}. *Dendrocalamus hamiltonii* had a maximum basal area of 51.9 m^2 ha^{-1} out of the total basal area of 53.1 m^2 ha^{-1} in both the bamboo species (Table 5.5). The total number of tree species was 73 and *Shorea robusta* was the dominant species (IVI 37.4) and *Ficus virens* (IVI 33.2) was the co-dominant species. The total tree basal area of this forest was 61.8 m^2 ha^{-1}. *Ficus virens* had the highest basal area of 18.7 m^2 ha^{-1} in the mixed bamboo forest. *Beilschmiedia assamica* had the highest density in the forest with 98 individuals ha^{-1}. The total density of the forest was 794 individuals ha^{-1}. Based on A/F ratio, both the bamboo species exhibited 100% clumped distribution pattern (Table 5.5). The tree species exhibited 94.5 % clumped distribution and 5.5% exhibited random distribution pattern (Table 5.6).

Table 5.5 Bamboo species composition, density, frequency, basal area, IVI and A/F ratio in tropical mixed bamboo forest of Nongkhyllem wildlife sanctuary

Sl. No.	Species	Density	Frequency	Basal area ha^{1}	IVI	A/F
1	*Dendrocalamus hamiltonii* Nees Arn ex Munro	3420	72	51 94	238 99	0 66
2	*Teinostachyum dullooa* Gamble	1524	28	1 16	61 01	1 94
	Total	4944	100	53 10	300 00	2 60

Table 5.6 Tree species composition, density, frequency, basal area, IVI and A/F ratio in tropical mixed bamboo forest of Nongkhyllem wildlife sanctuary

Sl. No.	Species	Density	Frequency	Basal Area m^{2} ha^{1}	IVI	A/F
1	*Acacia concinna* (Willd) DC	2	4	0 01	0 96	0 13
2	*Actinodaphne obovata* (Nees) Blume	4	4	0 01	1 22	0 25
3	*Albizia odoratissima* (L f) Benth	2	2	0 27	1 04	0 50
4	*Albizia procera* (Roxb) Benth	6	6	0 06	1 90	0 17
5	*Alstonia scholaris* R Br	8	8	0 47	3 16	0 13
6	*Amoora wallichii* King	2	2	0 03	0 65	0 50
7	*Artocarpus chaplasha* Roxb	8	8	2 22	6 00	0 13
8	*Beilschmiedia assamica* Meisn	98	56	3 70	28 09	0 03
9	*Bridelia retusa* A Juss	2	2	0 03	0 65	0 50
10	*Caesalpinia crista* L	4	4	0 01	1 22	0 25
11	*Calophyllum polyanthum* Wall	4	4	0 09	1 35	0 25
12	*Casearia glomerata* Roxb	10	10	1 29	5 09	0 10
13	*Castanopsis indica* A DC	2	2	0 06	0 69	0 50
14	*Castanopsis tribuloides* A DC	10	8	0 18	2 95	0 16
15	*Celastrus paniculatus* Willd	4	4	0 02	1 24	0 25
16	*Celastrus championi* Benth	2	2	0 02	0 64	0 50
17	*Chonemorpha fragrans* Alston	2	2	0 02	0 64	0 50
18	*Cinnamomum bejolghota* Sweet	2	2	0 01	0 61	0 50
19	*Cinnamomum pauciflorum* Nees	4	2	0 23	1 23	1 00
20	*Combretum punctatum* A Rich	4	4	0 05	1 28	0 25
21	*Combretum roxburghii* G Don	10	4	0 09	2 10	0 63
22	*Cyathostemma argenteum* (Blume) J Sincl	6	8	0 04	2 21	0 09
23	Desmos longiflorus	2	2	0 01	0 61	0 50
24	*Dillenia indica* L	6	6	0 09	1 96	0 64
25	*Dillenia pentagyna* Roxb	10	12	0 56	3 91	0 63
26	*Diospyros variegata* Kurz	24	16	0 35	6 37	0 09
27	*Drimycarpus racemosus* Hook f	2	2	0 01	0 62	0 05
28	*Duabanga grandiflora* Walp	8	4	0 80	3 00	0 50
29	*Dysoxylum binectariferum* Hiern	12	10	0 09	3 40	0 12
30	*Dysoxylum gobara* (Buch -Ham) Merrill	94	44	2 72	23 90	0 31
31	*Ficus altissima* Blume	2	2	0 02	0 64	0 50
32	*Ficus concinna* Miq	12	10	0 96	4 80	0 12
33	*Ficus hirta* Vahl	2	2	0 10	0 76	0 50
34	*Ficus virens* Ait	10	10	18 67	33 23	0 10

35	*Fissistigma polyanthum* (Hook.f. & Thomson) Merr.	12	12	0.03	3.66	0.08
36	*Garcinia cowa* Choisy	4	4	0.09	1.34	0.25
37	*Ficus altissima* Blume	2	2	0.04	0.66	0.50
38	*Garcinia morella* Desc.	6	4	0.02	1.48	0.38
39	*Garcinia paniculata* Roxb.	2	2	0.02	0.63	0.50
40	*Glochidion hirsutum* Voigt	2	2	0.04	0.67	0.50
41	*Goniothalamus simonsii* Hook.f. & Thomson	6	2	0.02	1.13	1.50
42	*Grewia disperma* Rottl. ex Spreng	4	4	0.02	1.23	0.25
43	*Hibiscus macrophyllus* Roxb.	14	6	1.41	5.09	0.39
44	*Ilex excelsa*Wall.	2	2	0.01	0.61	0.50
45	*Illigera khasiana* C.B.Clarke	2	2	0.03	0.65	0.50
46	*Knema linifolia* (Roxb.) Warb.	12	8	0.12	3.10	0.19
47	*Lasianthus lucidus* Blume	2	2	0.22	0.96	0.50
48	*Leea alata* Edgew.	68	44	0.65	17.29	0.04
49	*Litsea cubeba* Pers.	2	2	0.02	0.63	0.50
50	*Litsea khasyana* Meissn.	6	6	0.06	1.90	0.17
51	*Litsea laeta* Benth. & Hook.f.	14	12	0.07	3.97	0.10
52	*Litsea monopetala* (Roxb.) Pers.	36	24	0.25	9.11	0.06
53	*Macaranga denticulata* Muell.Arg.	4	2	0.01	0.77	1.00
54	*Macaranga indica* Wight	6	4	0.31	1.95	0.38
55	*Magnolia pterocarpa* Roxb.	2	2	0.02	0.63	0.50
56	*Mesua ferrea* L.	24	20	0.53	7.27	0.06
57	*Millettia pachycarpa* Benth.	2	2	0.11	0.78	0.50
58	*Persea villosa* (Roxb.) Kosterm.	6	6	0.02	1.83	0.17
59	*Polyalthia jenkinsii* Benth. & Hook.f.	16	8	0.30	3.80	0.25
60	*Randia wallichii* Hook.f.	4	4	0.01	1.22	0.25
61	*Schefflera venulosa* Harms.	4	4	0.05	1.29	0.25
62	*Schima wallichii* Choisy	60	42	14.05	37.42	0.03
63	*Shorea robusta* A.DC.	10	10	1.72	5.79	0.10
64	*Stercularia hamiltonii* (Kuntze) Adelb.	8	4	0.22	2.06	0.50
65	*Stereospermum chenoloides* DC.	26	18	6.59	17.09	0.08
66	*Syzygium diospyrifolium* (Wall. ex Duthie) S.N. Mitra	2	2	0.02	0.63	0.50
67	*Terminalia myriocarpa* Heurck & Muell.Arg.	6	6	0.48	2.58	0.17
68	*Tetrameles nudiflora* R.Br.	4	4	0.22	1.56	0.25
69	*Tetrastigma leucostaphylum* (Dennst.) Alston	16	16	0.18	5.09	0.06
70	*Toona ciliata* M. Roem.	10	8	0.03	2.70	0.16
71	*Uvaria lurida* Dalz. & Gibs.	2	2	0.04	0.66	0.50
72	*Vitex peduncularis* Wall.	2	2	0.04	0.67	0.50
73	*Wendlandia ligustrina* Wall.	4	2	0.40	1.50	1.00
	Total	794	576	61.76	300.00	

Among all the forest types, the regenerating broad-leaved forest had the greatest tree density (1728 individuals ha^{-1}) and species richness was highest in the old-growth broad-leaved forest (94 species). The Shannon diversity and Simpson dominance indices in the tropical landscape were high in the old-growth broad-leaved forest and follow the order: TOBF>TMBF>TSPF>TRBF>TTPF. The Species evenness index was highest in the regenerating broad-leaved forest and followed the order: TRBF>TMBF>TOBF>TSPF> TTPF. Among all tropical landscape elements, α diversity was highest in the old-growth broad-leaved forest and followed the order: TOBF>TMBF>TSPF>TTPS>TRBF (Table 5.7).

Table 5.7 Community indices for different tropical forest types of Nongkhyllem wildlife sanctuary (TOBF= Tropical old-growth broad-leaved forest; TRBF=Tropical regenerating broad-leaved forest; TTPF=Tropical teak plantation forest; TSPF=Tropical sal plantation forest; TMBF=Tropical mixed bamboo forest)

Parameters	TOBF	TRBF	TTPF	TSPF	TMBF
Species richness	94	34	33	67	73
Dominance	0.0	0.1	0.4	0.1	0.1
Shannon -Weiner diversity index	3.8	2.9	1.8	2.9	3.6
Simpson dominance index	1.0	0.9	0.6	0.9	1.0
Evenness	0.5	0.5	0.2	0.3	0.5
α diversity	24.3	6.0	6.8	15.4	20.7

PHYSICO-CHEMICAL PROPERTIES OF SOIL

Soil texture

The soil texture at both surface (0-10 cm) and sub-surface (10-20 cm) soil layers was sandy in all the forest types except the subsurface (10-20 cm) layer of sal plantation and mixed bamboo forests, where it was sandy loam (Table 5.8).

Table 5.8 Proportion of soil particles and soil texture class in different tropical forest types of Nongkhyllem wildlife sanctuary

Forest types	Depth (cm)	Proportion of soil particle			Soil textural class
		Clay (%)	Silt (%)	Total Sand (%)	
TOBF	0-10	10.2	1.0	88.8	Sandy
	10-20	11.3	2.0	86.7	Sandy
TRBF	0-10	11.2	2.0	86.7	Sandy
	10-20	13.3	4.1	82.6	Sandy
TTPF	0-10	10.2	0.9	88.9	Sandy
	10-20	11.2	2.0	86.8	Sandy
TSPF	0-10	13.3	1.0	85.7	Sandy
	10-20	17.3	4.1	78.6	Sandy loam
TMBF	0-10	12.1	6.1	81.8	Sandy
	10-20	12.2	8.1	79.7	Sandy loam

Water holding capacity

The water holding capacity was highest in the old-growth broad-leaved forest, and teak and sal plantation forests had the lowest water holding capacity among all the forest types (Fig. 5.1). Water holding capacity decreased significantly ($p<0.01$) with depth in all the forest types. Two-way ANOVA showed significant ($P< 0.01$) variation in the water holding capacity of the soil due to forest type ($F = 1689.7$) and soil depth ($F = 128.7$).

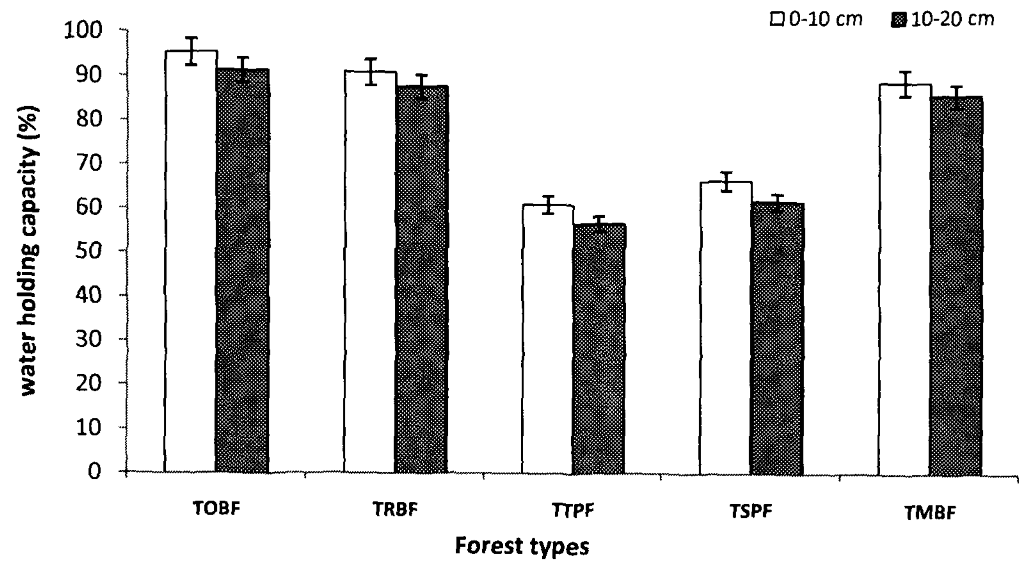


Fig. 5.1 Water holding capacity (%) in surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane landscape of Nongkhyllem wildlife sanctuary. The values are mean (±SE) of five replicate samples

Bulk density and Porosity

Bulk density was highest in the old-growth broad-leaved forest (1.5 g cm^{-3}) and lowest in the regenerating broad-leaved forest (1.2 g cm^{-3}) (Table 5.9). The bulk density increased with depth in all the forest types. Two-way ANOVA showed a significant ($P < 0.01$) variation in the bulk density of soil due to forest type (F = 268.2) and soil depth (F = 49.8). Conversely, soil porosity was highest in the regenerating broad-leaved forest (53.5%) and lowest in the old-growth broad-leaved forest (44.9%) (Table 5.9). Soil porosity declined with increase in depth in all the forest types. Two-way ANOVA showed a significant ($P < 0.01$) variation in the porosity of the soil due to forest type (F = 97.5).

Table 5.9 Bulk density (g cm^{-3}) and porosity (%) in surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary. The values are mean (±SE) of five replicate samples

Forest types	Depth (cm)	Bulk density (g cm^{-3})	Porosity (%)
TOBF	0-10	1.5	44.9
	10-20	1.5	43.0
TRBF	0-10	1.2	53.6
	10-20	1.3	52.5
TTPF	0-10	1.3	50.2
	10-20	1.3	48.7
TSPF	0-10	1.3	50.6
	10-20	1.4	49.1
TMBF	0-10	1.3	52.5
	10-20	1.4	52.4

Soil moisture content (SMC)

The SMC was highest during summer season and lowest during winter season (Fig. 5.2). When compared among the forest types, both the plantation forests had lower soil moisture content than the natural forests. Three-way ANOVA revealed a significant ($P< 0.01$) variation in the SMC due to forest type, depth and season (Table 5.10).

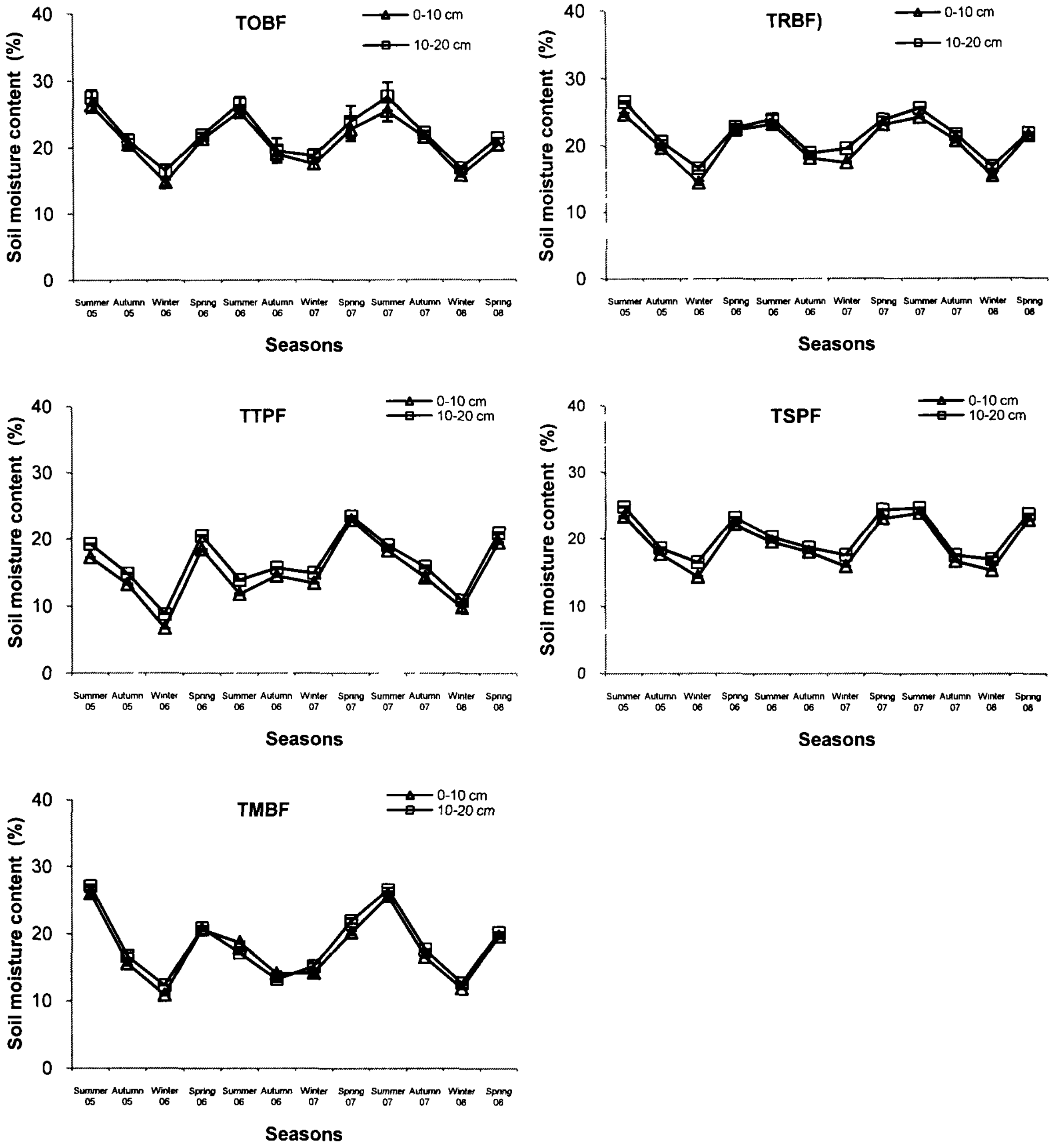


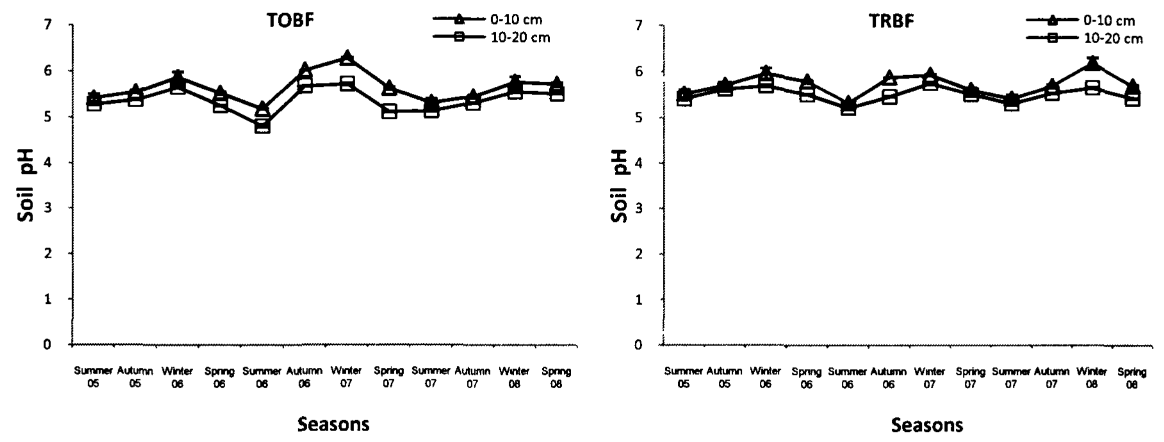
Fig. 5.2 Seasonal variation of soil moisture content (%) in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.10 Three way ANOVA showing effect of forest type, season and soil depth on soil moisture content (%) in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary (*p<0.01, **p<0.01)

Variation due to	df	SS	MS	F	p
Forest type	4	517.80	129.45	860.80**	0.000
Depth	1	37.17	37.17	247.20**	0.000
Season	11	1485.67	135.06	898.11**	0.000
Forest type*Depth	4	2.33	0.58	3.87*	0.009
Forest type*Season	44	271.99	6.18	41.10**	0.000
Depth*Season	11	4.65	0.42	2.81*	0.007
Forest type*Depth*Season	44	6.62	0.15	1.00	0.500

Soil pH

The soil pH was acidic ranging from pH value of 4.2 to 6.3. The plantation forests and the mixed bamboo forests had lower pH values than the two natural forests (Fig.5.3). The difference in pH between surface (0-10 cm) and sub-surface (10-20 cm) soil layers in the tropical forest types was significant (p<0.01). The soil was significantly more acidic in summer season than other seasons (p<0.01).



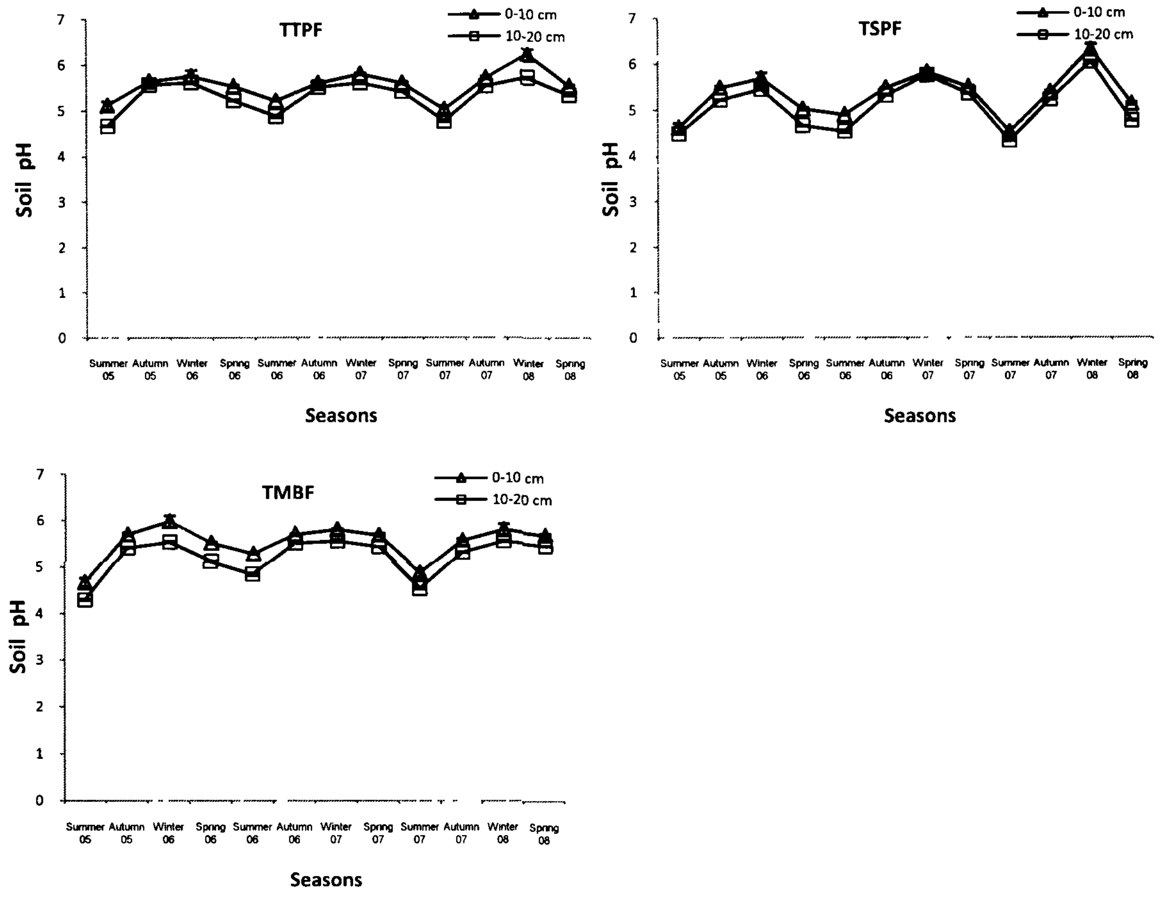


Fig. 5.3 Seasonal variation of soil pH in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Total Kjeldahl Nitrogen (TKN)

The highest TKN value was recorded during winter season and lowest during the summer season in all the forest types. The highest TKN value was recorded for the mixed bamboo forest (0.3%) and it was lowest in the sal plantation forest (0.2%) (Fig.5.4). The TKN declined significantly (p<0.01) with the increase in soil depth. Three-way ANOVA showed significant variation in TKN due to forest type, season and depth (p<0.01) (Table 5.11).

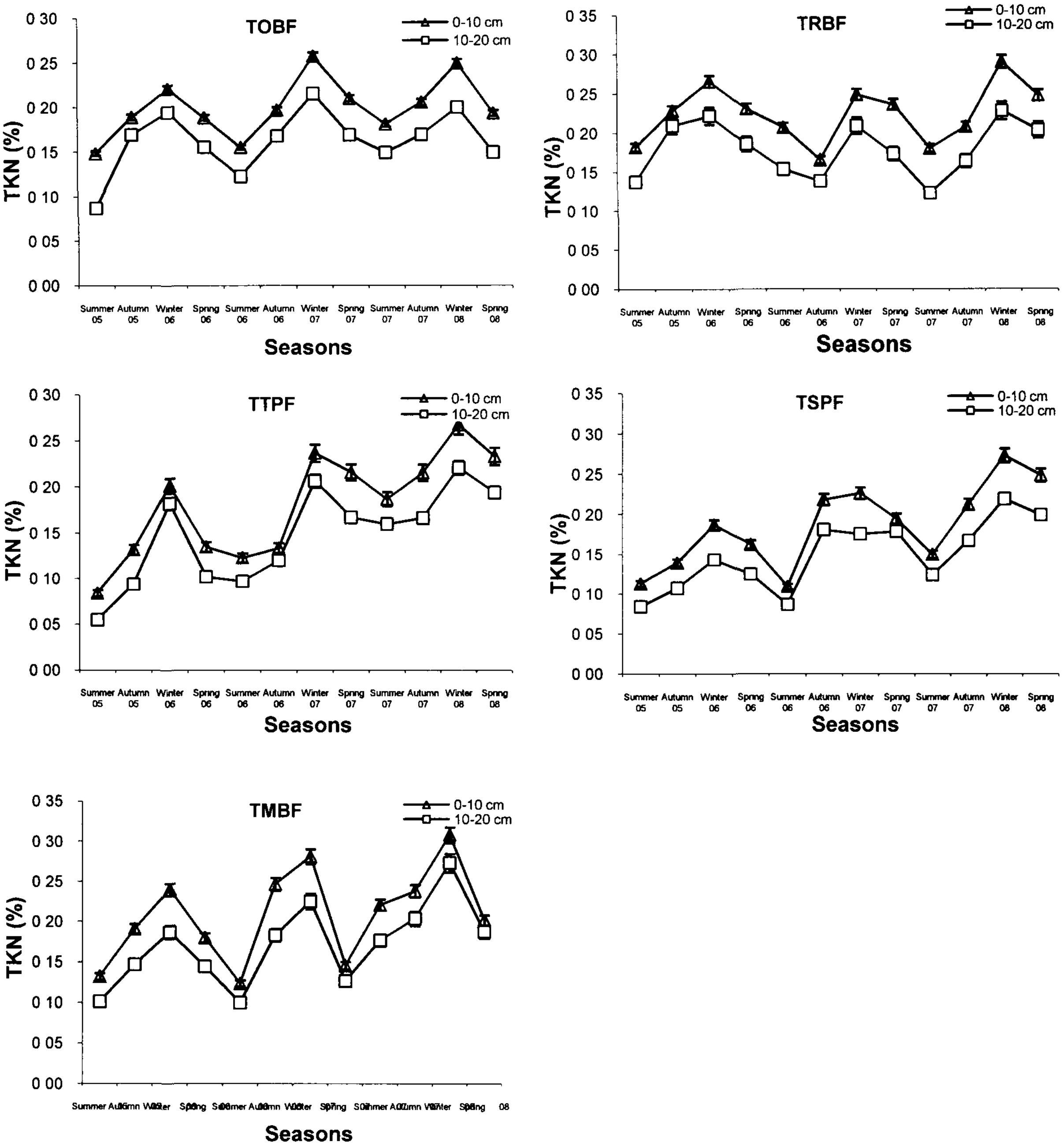


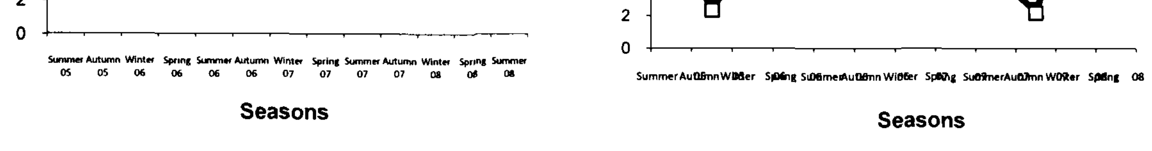
Fig. 5.4 Seasonal variation of total kjeldahl nitrogen (TKN, %, ±SE) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.11 Three-way ANOVA showing the effect of forest type, season and depth on total kjeldahl nitrogen in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary ($p<0.01$)**

Variation due to	df	SS	MS	F	*p*
Forest type	4	0.02	0.01	70.95**	0.000
Depth	1	0.04	0.04	526.11**	0.000
Season	11	0.17	0.02	191.09**	0.000
Forest type*Depth	4	0.00	0.00	1.36	0.262
Forest type*Season	44	0.06	0.00	16.05**	0.000
Depth*Season	11	0.00	0.00	0.80	0.644
Forest type*Depth*Season	44	0.00	0.00	1.00	0.500

Available Phosphorus (P)

The highest available Phosphorus value was obtained in the sal plantation forest (19.9 µg g^{-1}) and lowest in the teak plantation forest (2.3 µg g^{-1}). The values were greatest during the winter and lowest in the summer season (Fig. 5.5). The available Phosphorus declined significantly ($p<0.01$) with the increase in soil depth. Three-way ANOVA showed significant variation ($p<0.01$) due to forest type, depth and season (Table 5.12).



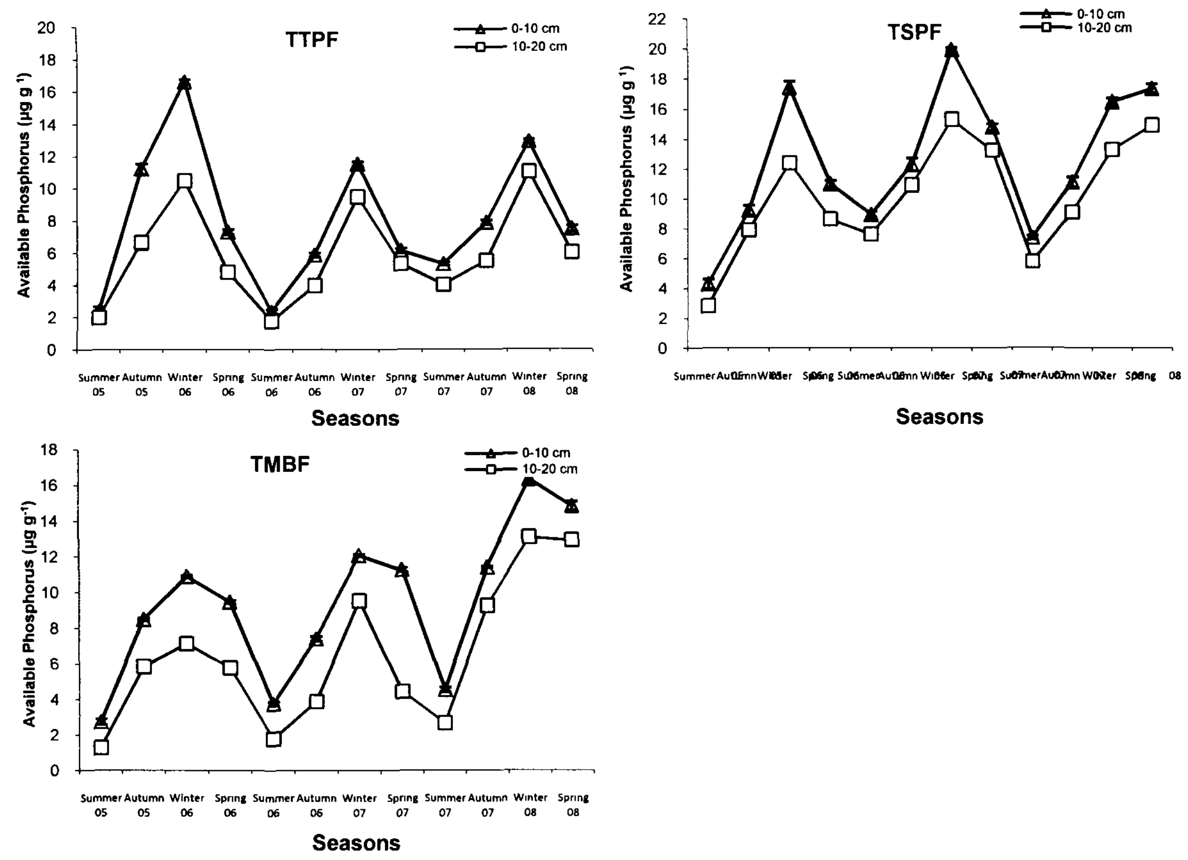


Fig. 5.5 Seasonal variation in available phosphorus (µg g^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.12 Three-way ANOVA showing the effect of forest type, season and depth on available Phosphorus in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary ($p<0.01$)**

Variation due to	df	SS	MS	F	*P*
Forest type	4	273.57	68.39	75.14**	0.000
Depth	1	188.37	188.37	206.94**	0.000
Season	11	990.22	90.02	98.90**	0.000
Forest type*Depth	4	2.25	0.56	0.62	0.651
Forest type*Season	44	622.36	14.14	15.54**	0.000
Depth*Season	11	21.18	1.93	2.11	0.039
Forest type*Depth*Season	44	40.05	0.91	1.00	0.500

Exchangeable Potassium (K)

The exchangeable K value was highest in the sal plantation forest (454.0 µg g^{-1}) and lowest in the teak plantation forest (117.5 µg g^{-1}). During winter season exchangeable K values was the highest and during summer it was the lowest in all the three years of study (Fig. 5.6). The exchangeable K declined significantly ($p<0.01$) with increase in soil depth. Three-way ANOVA showed significant variation due to forest type, depth and season ($p<0.01$) (Table 5.13).

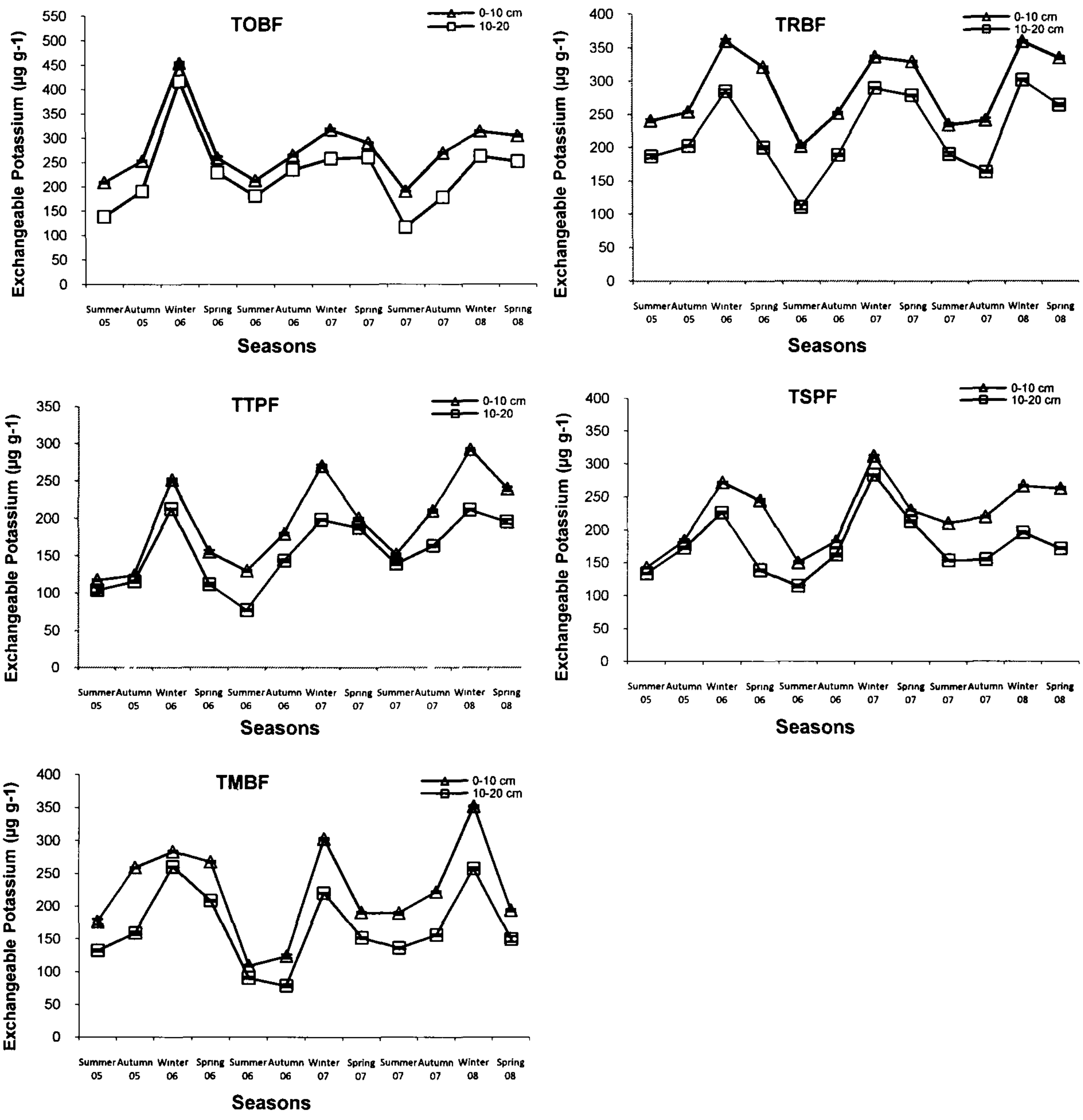


Fig. 5.6 Seasonal variation in exchangeable potassium (µg g^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.13 Three-way ANOVA showing the effect of forest type, season and depth on exchangeable Potassium in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary (*$p<0.01$, **$p<0.01$)

Variation due to	df	SS	MS	F	*p*
Forest type	4	102932.77	25733.19	57.09**	0.000
Depth	1	101128.91	101128.91	224.37**	0.000
Season	11	275441.06	25040.10	55.56**	0.000
Forest type*Depth	4	9857.07	2464.27	5.47*	0.001
Forest type*Season	44	111837.66	2541.77	5.64*	0.000
Depth*Season	11	7241.25	658.30	1.46	0.181
Forest type*Depth*Season	44	19831.51	450.72	1.00	0.500

SOIL ORGANIC CARBON (SOC) POOL

The SOC pool in the surface and sub-surface soil layer was largest in the old-growth broad-leaved forest (25.2 & 20.3 Mg ha^{-1}, respectively) and lowest in the teak plantation forest (10.1 & 9.1 Mg ha^{-1}, respectively). The values were greatest in summer season and lowest during winter season in all the forest types and at both the soil depths (Fig. 5.7). The mean seasonal SOC in the surface and sub-surface layers were largest in the old-growth broad-leaved forest (18.7 & 15.9 Mg ha^{-1} respectively) and lowest in the teak plantation forest (13.7 & 11.4 Mg ha^{-1}, respectively). The SOC pool declined significantly ($p<0.01$) with increase in the soil depth. Three-way ANOVA revealed significant variation due to forest type, season and depth ($p<0.01$) (Table 5.14).

The soil organic carbon pool upto a depth of 1 m varied significantly ($p<0.01$) among the four seasons in all the forest types. It was highest during summer and lowest during winter season (Fig. 5.8 – 5.12). SOC declined with the depth ($p<0.01$). The mean seasonal value of total SOC pool was highest in old-growth broad-leaved forest (83.2 Mg ha^{-1}) while the lowest SOC value was in teak plantation forest (41.7 Mg ha^{-1}) (Table 5.15). Three-way ANOVA revealed significant variation due to forest type, season and depth ($p<0.01$) (Table 5.16).

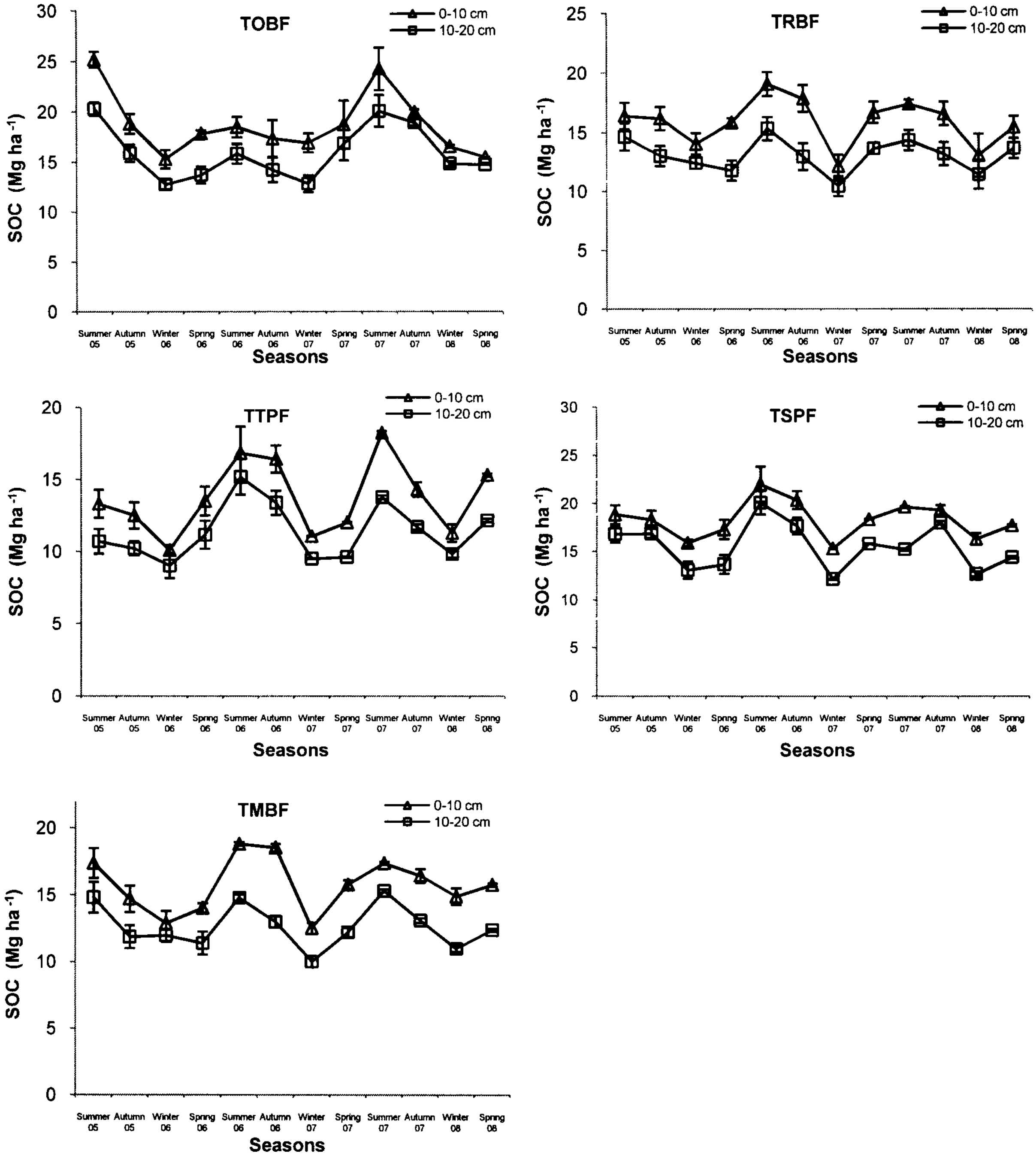


Fig. 5.7 Seasonal variation in the soil organic carbon (Mg ha^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.14 Three-way ANOVA showing effect of forest types, season and soil depth on soil organic carbon in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary (*$P<0.01$, **$p<0.01$)

Variation due to	df	SS	MS	F	p
Forest type	4	379.43	94.86	174.48**	0.000
Depth	1	231.00	231.00	424.90**	0.000
Season	11	363.20	33.02	60.73**	0.000
Forest type*Depth	4	1.56	0.39	0.72	0.586
Forest type*Season	44	146.13	3.32	6.11*	0.000
Depth*Season	11	9.24	0.84	1.54	0.150
Forest type*Depth*Season	44	23.92	0.54	1.00	0.500

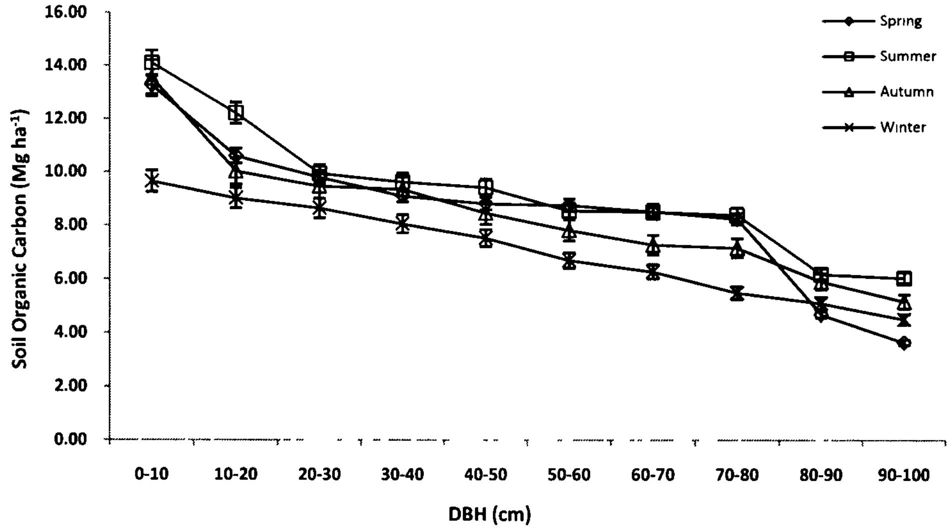


Fig. 5.8 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the old-growth broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

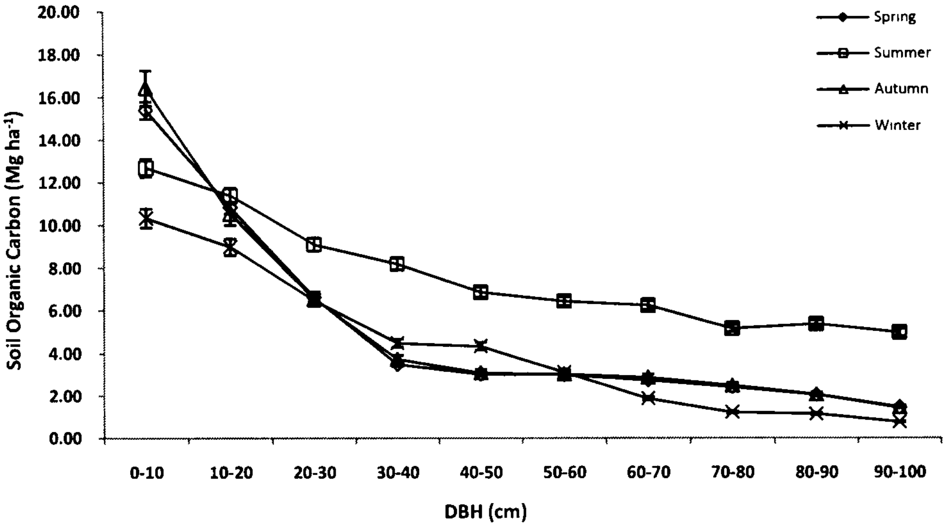


Fig. 5.9 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the regenerating broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

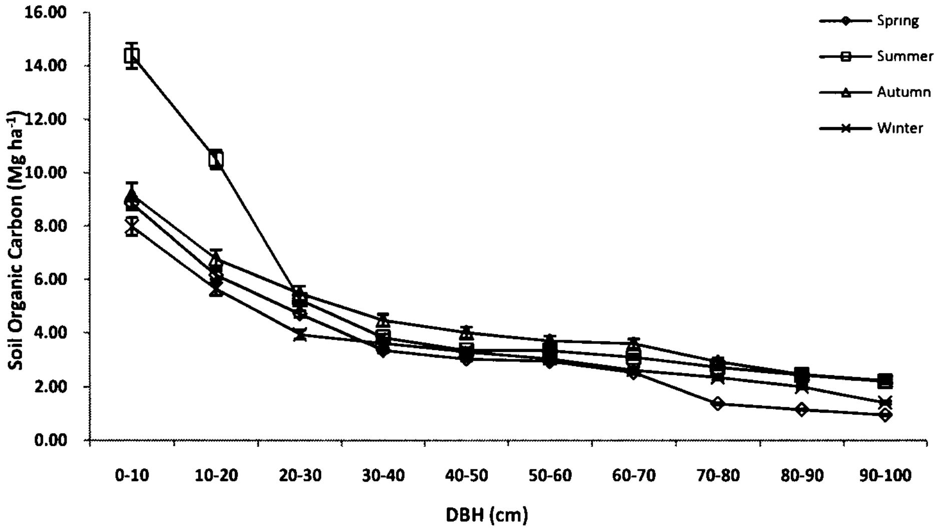


Fig. 5.10 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the teak plantation in tropical landscape of Nongkhyllem wildlife sanctuary

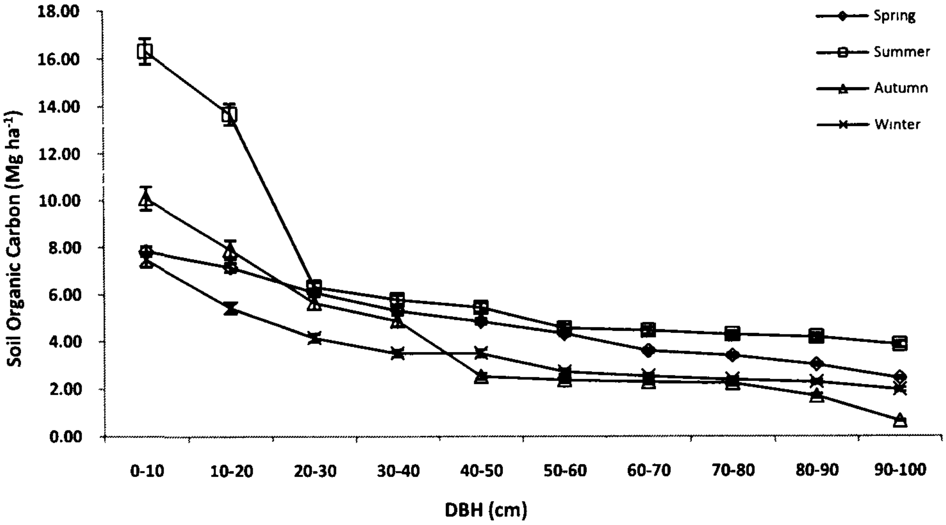


Fig. 5.11 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the sal plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

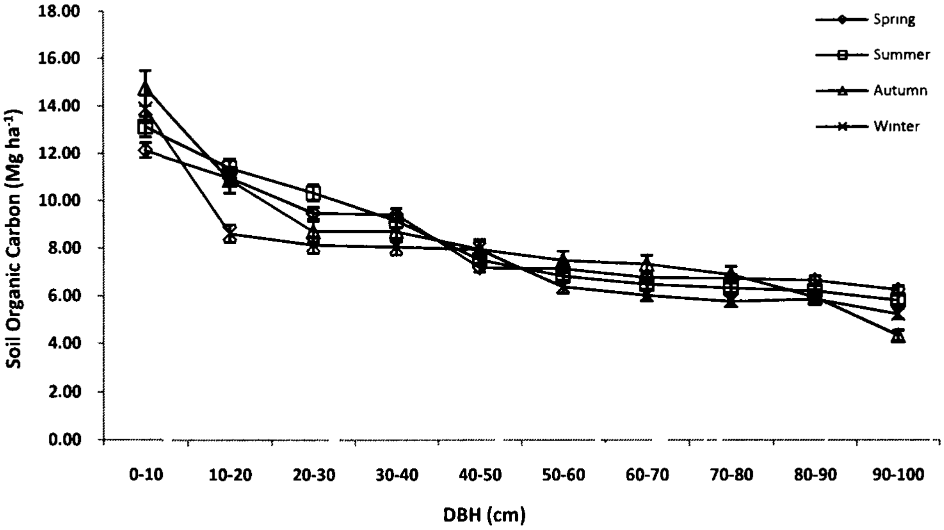


Fig. 5.12 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the mixed bamboo forest in tropical landscape of Nongkhyllem wildlife sanctuary

Table 5.15 Total SOC (Mg ha^{-1}) upto 1 m in different seasons in all tropical forest types of Nongkhyllem wildlife sanctuary

Forest type	Spring	Summer	Autumn	Winter	Mean
TOBF	85.3	92.8	84.0	70.8	83.2
TRBF	51.0	77.6	52.5	42.7	56.0
TTPF	35.1	51.1	44.8	35.9	41.8
TSPF	48.0	68.8	40.3	35.9	48.2
TMBF	82.7	83.2	83.0	75.9	81.2

Table 5.16 Three-way ANOVA showing effect of forest types, season and soil depth on soil organic carbon upto 1 m depth in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary (*$P<0.01$, **$p<0.01$)

Variation due to	df	SS	MS	F	p
Forest type	4	584.23	146.06	195.45**	0.000
Depth	9	1329.39	147.71	197.66**	0.000
Season	3	57.65	19.22	25.71**	0.000
Forest type*Depth	36	80.64	2.24	3.00*	0.000
Forest type*Season	12	121.62	10.13	13.56**	0.000
Depth*Season	27	66.31	2.46	3.29*	0.000
Forest type *Depth*Season	108	80.71	0.75	1.00	0.500

C/ N ratio in the soil

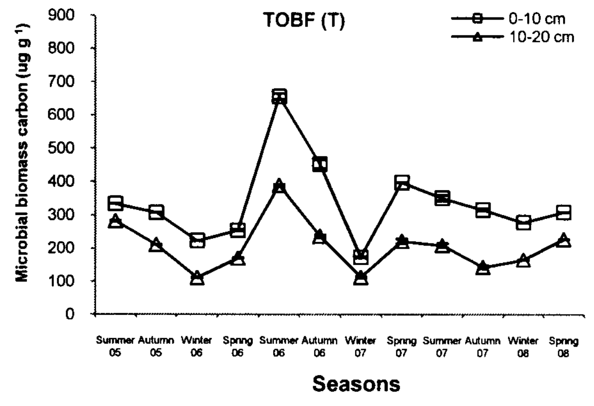
The C/N ratio was greater in the sub-surface soil layer than the surface layer. In the tropical landscape, highest C/N ratio was recorded in the mixed bamboo forest (9.6 in the surface layer and 10.3 in the sub-surface layer) and old-growth broad-leaved forest (9.6 in the surface layer and 9.9 in the sub-surface layer) and lowest in the regenerating broad-leaved forest (7.1 in the surface layer and 7.3 in the sub-surface layer) (Table 5.17).

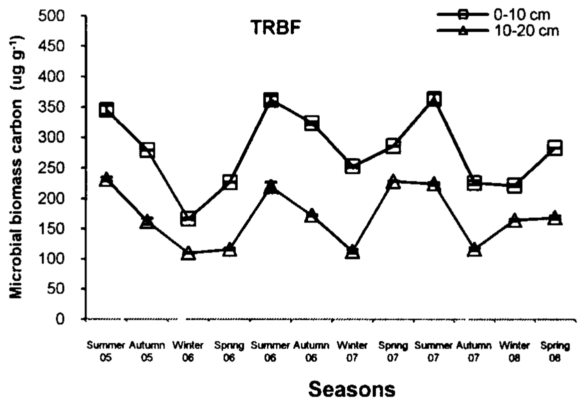
Table 5.17 Soil organic carbon, total kjeldahl nitrogen and C/N ratio in soils in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary. The values are mean of 12 seasons and five replicates each

Forest types	Depths (cm)	SOC (%)	TKN (%)	C/N
TOBF	0-10	1.91	0.20	9.55
	10-20	1.59	0.16	9.94
TRBF	0-10	1.55	0.22	7.05
	10-20	1.31	0.18	7.28
TTPF	0-10	1.37	0.18	7.61
	10-20	1.15	0.15	7.67
TSPF	0-10	1.58	0.21	7.52
	10-20	1.27	0.17	7.47
TMBF	0-10	1.83	0.19	9.63
	10-20	1.54	0.15	10.27

MICROBIAL BIOMASS CARBON POOL

The microbial biomass carbon (MBC) both in the surface and sub-surface soil layers peaked during summer season (679.6 & 470.3 μg g^{-1}, respectively) in the sal plantation forest and minimum during the winter season (166.6 & 110.2 μg g^{-1}) in the regenerating broad-leaved forest. The mean seasonal MBC in the surface and sub-surface layers were highest in the sal plantation forest (420.4 & 293.9 μg g^{-1}, respectively) and lowest in the regenerating broad-leaved forest (278.4 & 169.4 μg g^{-1}, respectively) (Fig. 5.13). Based on mean MBC concentration values in the surface soil layer, the sal plantation forest had significantly greater MBC than the mixed bamboo forest, old-growth broad-leaved forest, teak plantation forest and regenerating broad-leaved forest. The surface soil layer had significantly (p<0.01) higher values than the sub-surface layer in all the forest types. Three-way ANOVA showed significant variation due to forest type, season and depth (p<0.051 (Table 5.18).





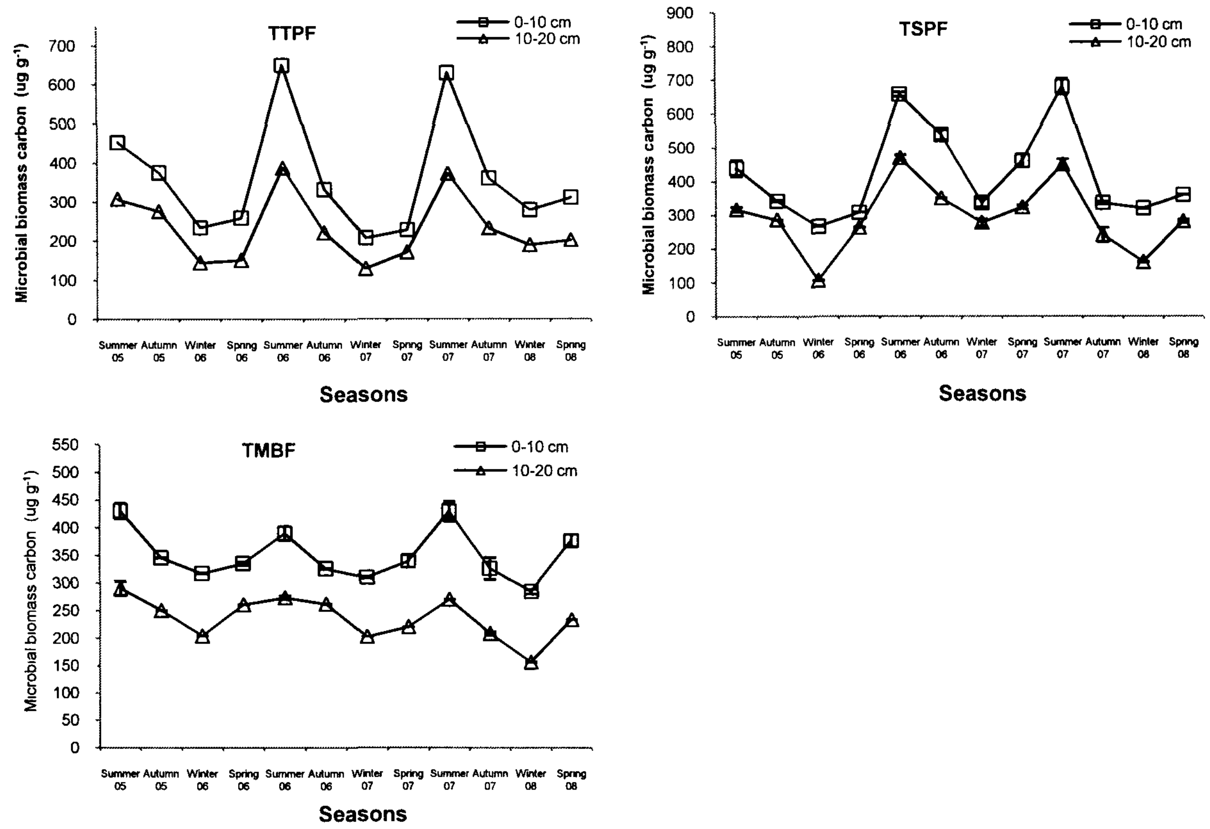


Fig. 5.13 Seasonal variation in the microbial biomass carbon (μg g^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.18 Three-way ANOVA showing the effect of forest type, season and depth on microbial biomass carbon in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary (*$p<0.01$, **$p<0.01$)

Variation due to	df	SS	MS	F	p
Forest type	4	188008.32	47002.08	33.70**	0.000
Depth	11	639403.13	58127.56	41.67**	0.000
Season	1	411980.34	411980.34	295.36**	0.000
Forest type*Depth	44	311556.16	7080.82	5.08*	0.000
Forest type*Season	4	8307.54	2076.89	1.49	0.222
Depth*Season	11	54306.50	4936.95	3.54*	0.001
Forest type*Depth*Season	44	61372.81	1394.84	1.00	0.500

Percentage contribution of MBC to SOC

During summer season the percentage contribution of MBC to SOC was highest in all the forest types and it was lowest during winter season. The percentage contribution was greater in the surface soil layer than that of sub-surface soil layer. The teak plantation forest had greater percentage contribution (4.9) followed by sal plantation forest (3.6), mixed bamboo forest (2.9), regenerating broad-leaved forest (2.7) and old-growth broad-leaved forest (2.7). In the sub-surface layer it was highest in the teak plantation forest (3.5) and lowest in regenerating broad-leaved forest (2) during summer season (Table 5.19).

Table 5.19 Percentage contribution of microbial biomass carbon to soil organic carbon in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary. The seasons are mean of three years from 2005 to 2008

Forest types	Depth (cm)	Summer	Autumn	Winter	Spring
TOBF	0-10	2.7	1.6	1.2	1.7
	10-20	2.1	1.1	0.7	1.4
TRBF	0-10	2.7	1.6	1.4	1.6
	10-20	2.0	1.0	1.0	1.4
TTPF	0-10	4.9	2.4	1.9	1.7
	10-20	3.5	1.9	1.4	1.5
TSPF	0-10	3.6	2.2	1.7	1.9
	10-20	3.2	1.7	1.2	1.9
TMBF	0-10	2.9	2.1	1.9	2.1
	10-20	2.4	1.8	1.5	1.8

LITTER POOL

The total litterfall in different forest types ranged between 6.5 to 10.8 Mg ha^{-1} yr^{-1}. The leaf litter contributed between 63.2 to 82.2 % to the total litter. The maximum (82.2%) contribution of leaf litter to the total litterfall was in the teak plantation forest. The sal plantation forest had the maximum litterfall (10.8 Mg ha^{-1} yr^{-1}), followed by old-growth broad-leaved (9.8 Mg ha^{-1} yr^{-1}), teak plantation (7.4 Mg ha^{-1} yr^{-1}), regenerating broad-leaved

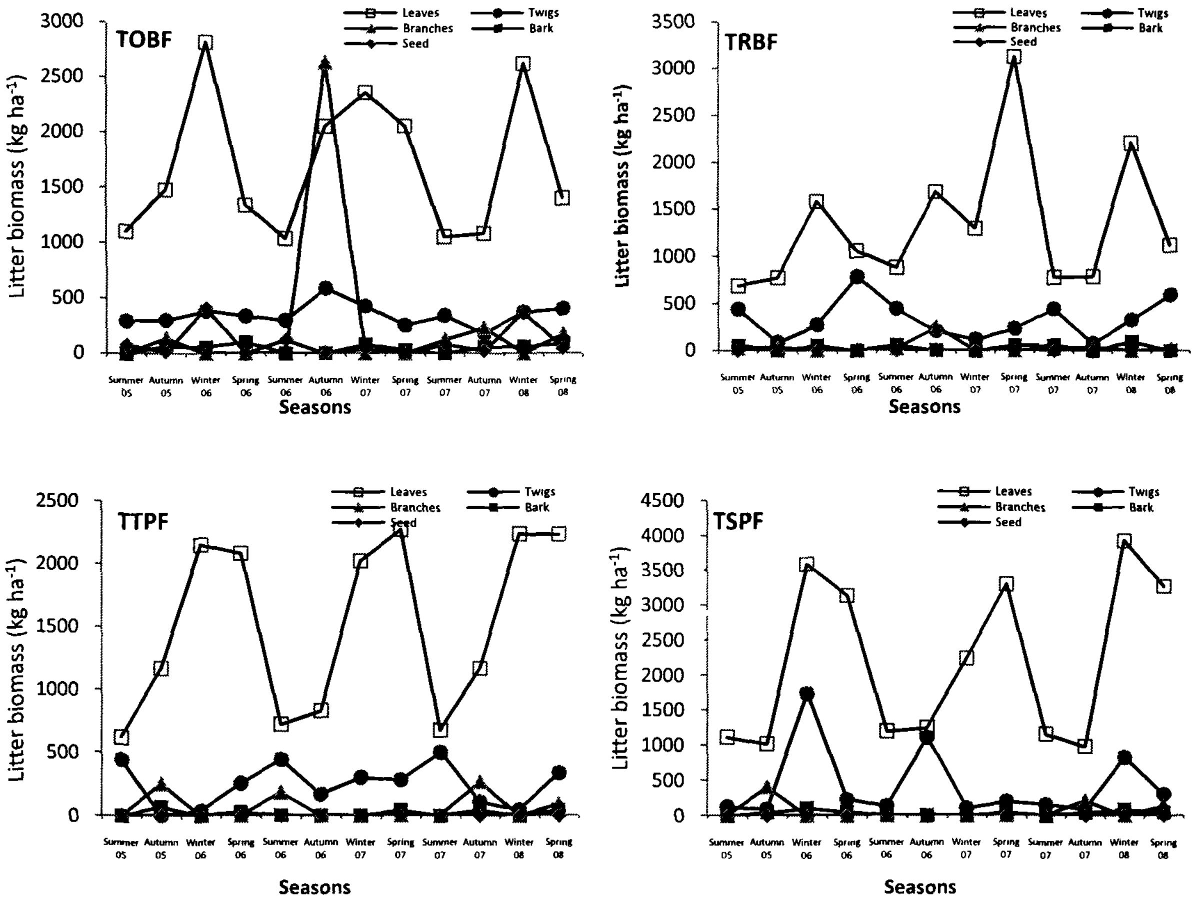
(6.9 Mg ha^{-1} yr^{-1}) and mixed bamboo (6.5 Mg ha^{-1} yr^{-1}) forests, respectively (Fig. 5.14 & Table 5.20). The litterfall was maximum during winter and spring seasons and minimum during summer season in all the forest types. Two-way ANOVA revealed significant variation in litterfall among forest types (ANOVA F = 4.95; $p<0.01$) and seasons (ANOVA F = 5.59; $p<0.01$).

Litter decomposition

Leaf litter showed three distinct phases of decomposition. The first phase showed a rapid rate of decomposition during the first 90 days (8 - 27% weight remaining). This phase was followed by a period of slow weight loss lasting for 180 days (5 – 40 % weight remaining). The third phase i.e. between 270 – 360 days, decomposition was rapid (0.5 – 9% weight remaining) (Fig. 5.15). The TMBF showed faster litter decomposition than the other forest types. The weight loss pattern followed the trend: TMBF> TSPF>TRBF>TOBF>TTPF. The litter decay constant (k) was of the order TMBF>TOBF>TSPF=TTPF>TRBF. The time required for 50% of litter to decay (t_{50}) varied among the forest types and ranged between 5.8 in TRBF and 11.5 in TMBF. Similarly, t_{99} ranged between 41.7 in TRBF and 83.3 in TMBF (Table 5.21). The litter turnover rate varied among the different forest types. The TOBF and TRBF showed high turnover rate (1.0 yr^{-1}) and low in TMBF (0.5 yr^{-1}) (Table 5.22). The litter turnover time varied from 1 - 2.2 yr^{-1} (Table 5.23). Based on litter turnover rate, the forest types may be arranged in the order: TMBF>TSPF>TTPF>TOBF>TRBF.

Table 5.20 Litterfall by different components (Mg $ha^{-1}yr^{-1}$) and their carbon content in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of 12 seasons and five replicate samples. The figures in parenthesis represent the percentage contribution of litter components to total litter

Forest types	Leaves	Twigs	Branches	Bark	Seed	Total	Total litter C
TOBF	6.8 (68.9)	1.4 (14.1)	1.1 (11.2)	0.2 (1.8)	0.4 (4.0)	9.8	4.6
TRBF	5.3 (76.8)	1.3 (19.3)	0.1 (2.0)	0.1 (1.8)	-	6.9	3.3
TTPF	6.1 (82.2)	1.0 (13.2)	0.3 (3.6)	0.1 (1.0)	-	7.4	3.5
TSPF	8.7 (80.7)	1.7 (15.8)	0.3 (2.3)	0.1 (1.2)	-	10.8	5.1
TMBF	4.1 (63.2)	0.9 (14.2)	1.5 (22.6)	-	-	6.5	3.1



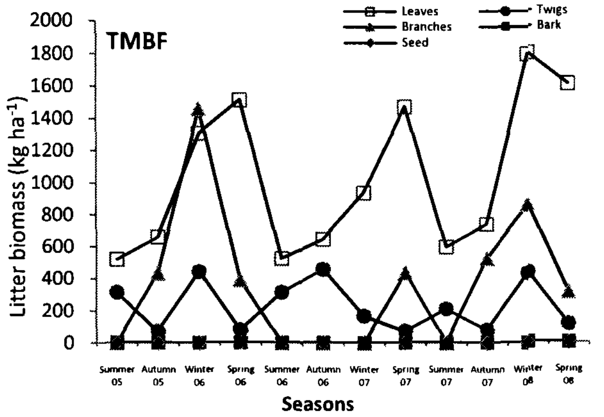


Fig. 5.14 Seasonal variation in litterfall (kg ha^{-1}) in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

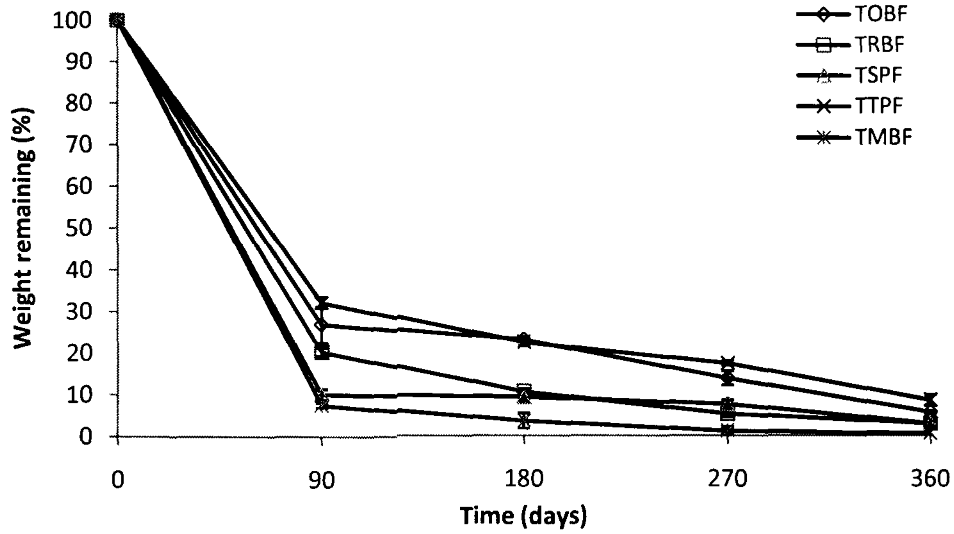


Fig. 5.15 Weight remaining (%) in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during *in situ* leaf litter decomposition. The values are mean (±SE) of five replicate samples

Table 5.21 Annual decay constant (k) of litter in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary

Parameters	TOBF	TRBF	TTPF	TSPF	TMBF
Decay constants					
k	0.08	0.12	0.09	0.09	0.06
t_{50}	8.66	5.78	7.37	7.88	11.55
t_{99}	62.50	41.67	53.19	56.82	83.33

Table 5.22 Litter turnover rate (k_L yr^{-1}) in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary

Forest	Litter turnover rate (k_L yr^{-1})					
types	Leaves	Twigs	Branches	Bark	Seed	Total
TOBF	0.9	0.6	1.3	1.6	2.4	1.0
TRBF	1.2	1.5	1.6	2.2	0.0	1.0
TTPF	1.0	1.8	1.9	2.3	0.0	0.8
TSPF	0.7	1.7	4.0	1.4	0.0	0.6
TMBF	0.4	1.0	1.3	0.0	0.0	0.5

Table 5.23 Litter turnover time (t yr^{-1}) in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary

Forest	Turnover time (t yr^{-1})					
types	Leaves	Twigs	Branches	Bark	Seed	Total
TOBF	1.1	2.1	0.9	0.6	0.5	1.0
TRBF	0.9	0.7	0.7	0.6	0.0	1.0
TTPF	1.0	0.7	0.4	0.6	0.0	1.2
TSPF	1.4	0.6	0.3	0.7	0.0	1.8
TMBF	2.6	1.0	0.8	0.0	0.0	2.2

ABOVEGROUND AND BELOWGROUND BIOMASS POOL

Old-growth broad-leaved forest

The allometric model of Chambers *et al.* (2001a) for broad-leaved tree aboveground biomass (AGB) was used to determine AGB of broad-leaved tree component in the old-growth broad-leaved forest. The model was:

$$Y_l = \exp\left[- 0.37 + 0.33 * \ln(D) + 0.933 * \ln(D)^2 - 0.122 * \ln(D)^3\right]$$

Similarly, the model of Cairns *et al.* (1997) for broad-leaved tree belowground biomass (BGB) was used to determine BGB of broad-leaved tree component in the forest. The model was:

$Y_2 = \exp[-1.0587 + 0.8836 * \ln(AGB)]$

where, Y= biomass/tree, D= diameter at breast height, AGB= aboveground biomass.

The same allometric models have been used for estimation of AGB and BGB for all the broad-leaved tree species in tropical landscape.

The carbon (%) derived for the different tropical forest types were used for determination of carbon content (Table 5.24).

Table 5.24 Carbon content (%) in different tree components in tropical landscape of Nongkhyllem wildlife sanctuary

Forest	Aboveground (%)					Belowground (%)		Litter (%)			
types	Leaf	Twig	Branch/ culm	Cone	Stem	Fine root	Coarse root	Leaf	Twig	Branch /culm	Misc.
TOBF	........................49........................					...47...		48	47	48	47
TRBF	........................48......................					...47...		47	47	48	47
TTPF	........................49....................					...48...		48	47	47	47
TSPF	........................49......................					...48...		47	47	49	47
TMBF	49	48	49	NA	49	...47...		48	48	50	48

Density diameter distribution

The density diameter distribution in the old-growth broad-leaved forest yielded a reverse J-shaped curve. The total tree density of the forest was 1090 individuals ha^{-1}. The diameter class >5-10 cm had the maximum number of trees (372 individuals ha^{-1}) contributing 34.1% to the total tree density in the forest. The higher diameter class i.e. >60 cm had 74 trees contributing 6.8% to the total tree density in the forest. The highest diameter class >100 cm had only 4 individuals ha^{-1} and contributed 0.4% to the total tree density in the forest (Fig. 5.16).

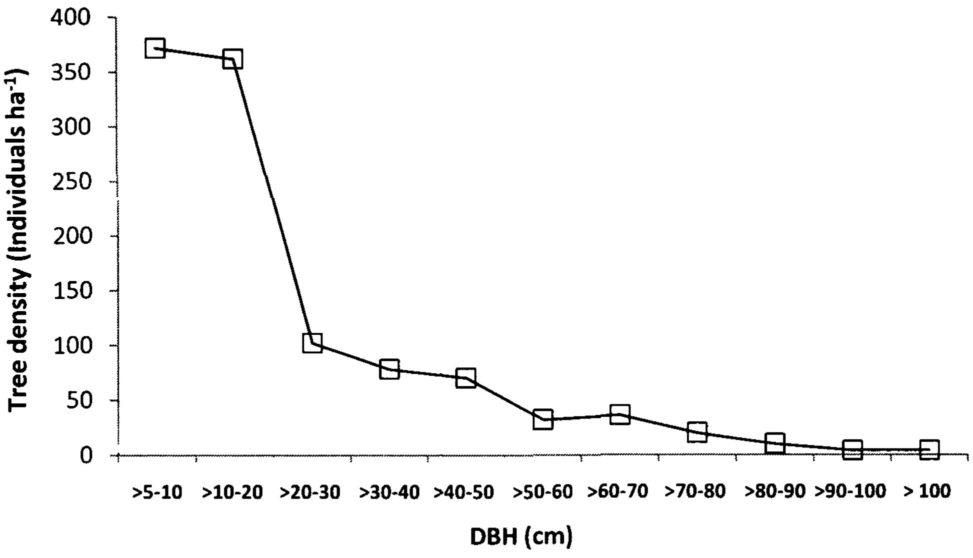


Fig. 5.16 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of old-growth broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

Aboveground biomass and carbon

Tree aboveground biomass in the old-growth broad-leaved forest was high in the >40-50 and >60-70 cm diameter classes. The higher diameter classes i.e. >60 cm contributed 42.5% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon were 313.8 Mg ha^{-1} and 153.8 Mg C ha^{-1}, respectively (Table 5.25).

Table 5.25 Aboveground biomass and biomass carbon in different diameter classes of old-growth broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class	Aboveground tree biomass		
(cm)	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	11.5	5.7	3.7
>10-20	28.2	13.8	9.0
>20-30	20.7	10.1	6.6
>30-40	38.2	18.7	12.2
>40-50	47.7	23.4	15.2
>50-60	34.1	16.7	10.9
>60-70	47.9	23.5	15.3
>70-80	34.9	17.1	11.1
>80-90	24.4	11.9	7.8
>90-100	16.8	8.2	5.4
> 100	9.5	4.7	3.0
Total	313.8	153.8	100.0

Belowground biomass and carbon

Tree belowground biomass in the old-growth broad-leaved forest was high in the >40-50 cm diameter class. The higher diameter classes i.e. >60 cm contributed 37.6% to the total tree below ground biomass in the forest. The total belowground biomass and carbon were 50.8 Mg ha^{-1} and 23.9 Mg C ha^{-1}, respectively (Table 5.26).

Table 5.26 Belowground biomass and carbon in different diameter classes of old- growth broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class (cm)	Belowground tree biomass (Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	2.7	1.3	5.3
>10-20	5.8	2.7	11.4
>20-30	3.8	1.8	7.5
>30-40	6.5	3.0	12.7
>40-50	7.7	3.6	15.1
>50-60	5.3	2.5	10.4
>60-70	7.1	3.3	14.0
>70-80	5.0	2.4	9.9
>80-90	3.4	1.6	6.7
>90-100	2.3	1.1	4.5
> 100	1.3	0.6	2.5
Total	50.8	23.9	100.0

Herb and Shrub biomass and carbon

The herb and shrub biomass were 12.4 and 37.1 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 5.8 and 17.1 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (323.7 Mg ha^{-1}) in the forest contributing only 0.004 and 0.012%, respectively (Table 5.27).

Table 5.27 Herb and shrub biomass and carbon and % contribution to total AGB in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary

Forest types	Herb biomass and carbon (kg ha^{-1})	(kg C ha^{-1})	Shrub biomass and carbon (kg ha^{-1})	(kg C ha^{-1})	%AGB Herb	%AGB Shrub
TOBF	12.43	5.84	37.11	17.81	0.004	0.012
TRBF	8.24	3.87	32.05	15.38	0.005	0.021
TTPF	11.05	5.18	40.12	19.26	0.006	0.020
TSPF	13.42	6.31	44.05	21.13	0.003	0.010
TMBF	7.45	3.50	23.12	11.10	0.002	0.006

Regenerating broad-leaved forest

Density diameter distribution

Total tree density of the forest was 1728 individuals ha^{-1}. The diameter class >10-20 cm had the maximum number of trees (950 individuals ha^{-1}) contributing 55% to the total density in the forest. The highest diameter class i.e. >30-40 cm had 26 trees contributing 1.5% to the total tree density in the forest (Fig. 5.17).

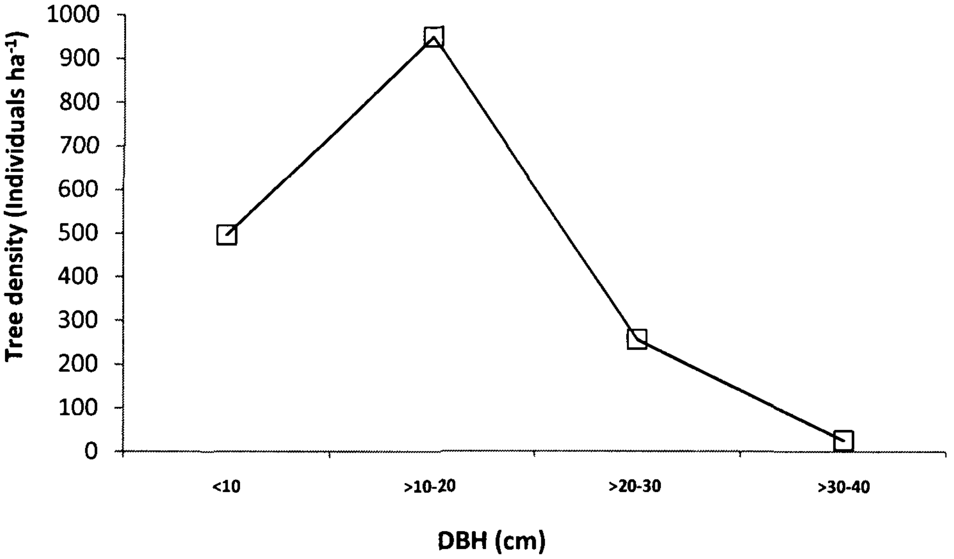


Fig. 5.17 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of regenerating broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

Aboveground biomass and carbon

In the regenerating broad-leaved forest, the tree aboveground biomass was high in the >10-20 cm diameter class indicating the young age of trees. This diameter class contributed 53.5% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon in the forest were 152.3 Mg ha^{-1} and 74.6 Mg C ha^{-1}, respectively (Table 5.28).

Table 5.28 Tree aboveground biomass and carbon in different diameter classes of regenerating broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class (cm)	Aboveground tree biomass		
	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	16.2	7.9	10.6
>10-20	81.5	39.9	53.5
>20-30	43.6	21.4	28.6
>30-40	11.1	5.4	7.3
Total	152.4	74.7	100.0

Belowground biomass and carbon

Tree below ground biomass in the regenerating broad-leaved forest was high in the >10-20 cm diameter class contributing 54.7% to the total tree belowground biomass in the forest. Total tree belowground biomass and carbon were 30.3 Mg ha^{-1} and 14.2 Mg C ha^{-1}, respectively (Table 5.29).

Table 5.29 Belowground biomass and carbon in different diameter classes of regenerating broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class (cm)	Belowground tree biomass		
	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	3.8	1.8	12.5
>10-20	16.6	7.8	54.7
>20-30	8.0	3.8	26.5
>30-40	1.9	0.9	6.2
Total	30.3	14.2	100.0

Herb and Shrub biomass and biomass carbon

The herb and shrub biomass were 8.2 and 32.1 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 3.9 and 15.4 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (159.34 Mg ha^{-1}) in the forest contributing only 0.005 and 0.021%, respectively (Table 5.27).

Teak plantation forest

Density diameter distribution

Total tree density of the forest was 864 individuals ha^{-1}. The diameter class >5-10 cm had the maximum number of trees (306 individuals ha^{-1}) contributing 35.4% to the total tree density in the forest. The highest diameter class >60-70 cm had 8 trees contributing 0.9% to the total tree density in the forest (Fig. 5.18).

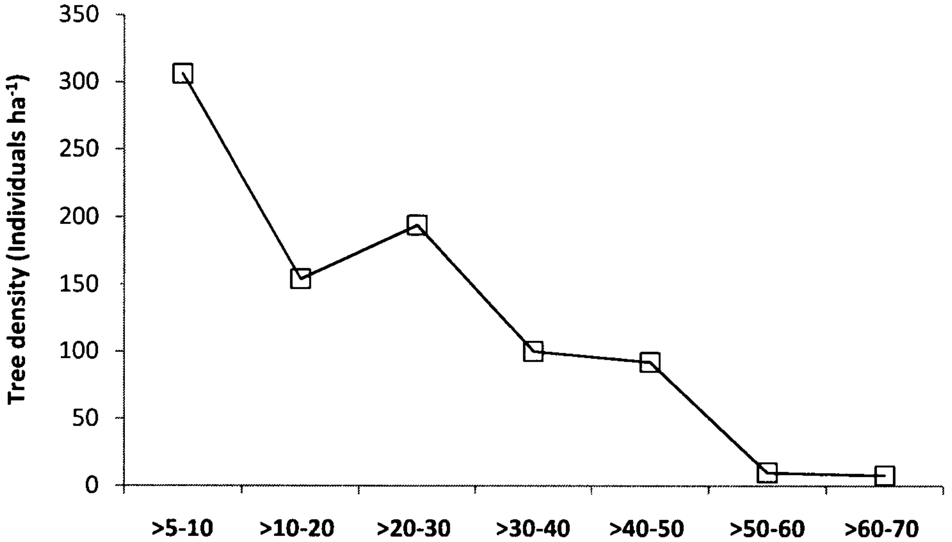


Fig. 5.18 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of teak plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

Aboveground biomass and carbon

Tree aboveground biomass in the teak plantation forest was high in the >40-50 cm diameter class. The higher diameter classes i.e. >60-70 cm contributed only 4.8% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon in the forest were 198.8 g ha^{-1} and 97.4 Mg C ha^{-1}, respectively (Table 5.30).

Table 5.30 Aboveground biomass and carbon in different diameter classes of teak (*Tectona grandis*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class	Aboveground tree biomass		
(cm)	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	7.7	3.8	3.9
>10-20	17.9	8.8	9.0
>20-30	44.6	21.9	22.5
>30-40	45.4	22.2	22.8
>40-50	61.8	30.3	31.1
>50-60	11.8	5.8	5.9
>60-70	9.6	4.7	4.8
Total	198.8	97.4	100.(

Belowground biomass and carbon

Tree belowground biomass in the teak plantation forest was high in the >40-50 cm diameter class. The highest diameter classes i.e. >60-70 cm contributed only 4.2% to the total tree belowground biomass in the forest. Total tree belowground biomass and carbon were 34.4 Mg ha^{-1} and 16.2 Mg C ha^{-1}, respectively (Table 5.31).

Table 5.31 **Belowground biomass and carbon in different diameter classes of teak (*Tectona grandis*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary**

DBH class (cm)	Belowground tree biomass		
	(Mg ha^{-1})	(Mg ha^{-1})	%
>5-10	1.9	0.9	5.4
>10-20	3.6	1.7	10.4
>20-30	8.1	3.8	23.7
>30-40	7.7	3.6	22.3
>40-50	10.0	4.7	28.9
>50-60	1.8	0.9	5.3
>60-70	1.4	0.7	4.2
Total	34.4	16.2	100.0

Herb and Shrub biomass and carbon

The herb and shrub biomass were 11.1 and 40.1 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 5.2 and 19.3 kg C ha^{-1}, respectively. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (206.2 Mg ha^{-1}) in the forest contributing only 0.006 and 0.02%, respectively (Table 5.27).

Sal plantation forest

Density diameter distribution

The density diameter distribution in the sal plantation forest yielded a reverse J-shaped curve. Total tree density of the forest was 1104 individuals ha^{-1}. The diameter class >5-10 cm had the maximum number of trees (362 individuals ha^{-1}) contributing 32.8% to the total tree density in the forest. The higher diameter class i.e. >60 cm had 108 trees contributing 9.8% to the total tree density in the forest. The highest diameter class >100 cm had only 2 individuals ha^{-1} and contributed 0.2% to the total tree density in the forest (Fig. 5.19).

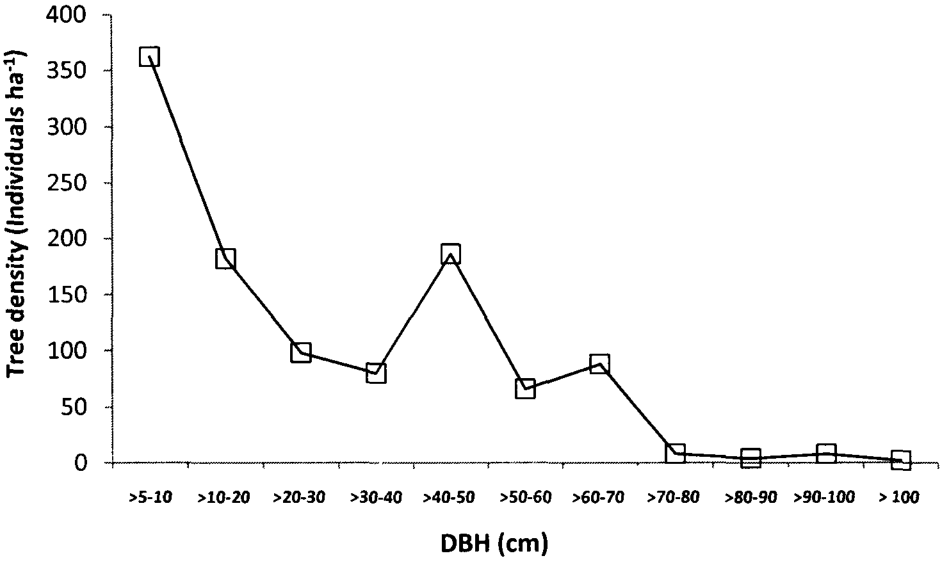


Fig. 5.19 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of tropical sal plantation forest of Nongkhyllem wildlife sanctuary

Aboveground biomass and carbon

Table 5.32 Aboveground biomass and carbon in different diameter classes of sal (*Shorea robusta*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class (cm)	Aboveground tree biomass		
	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	10.4	5.1	2.2
>10-20	12.0	5.9	2.6
>20-30	27.4	13.4	5.9
>30-40	42.5	20.8	9.2
>40-50	134.0	65.6	29.1
>50-60	86.8	42.5	18.8
>60-70	90.7	44.4	19.7
>70-80	15.6	7.7	3.4
>80-90	8.9	4.4	1.9
>90-100	23.4	11.5	5.1
> 100	9.7	4.7	2.1
Total	461.1	226.0	100.0

Tree aboveground biomass in the sal plantation forest was high in the >40-50 cm diameter class. The higher diameter classes i.e. >60 cm contributed 32.2% to the total tree aboveground biomass in the forest. Most trees in the lower diameter class i.e. >5-10 cm belong to other broad-leaved species present in the under canopy region of the sal plantation forest. Total tree aboveground biomass and carbon were 461.1 Mg ha^{-1} and 225.9 Mg C ha^{-1}, respectively (Table 5.32).

Belowground biomass and carbon

Tree belowground biomass in the sal plantation forest was higher in the >40-50 cm diameter class. The higher diameter classes i.e. >60 cm contributed 29.1% to the total tree belowground biomass in the forest. Total tree belowground biomass and carbon were 73.2 Mg ha^{-1} and 34.4 Mg C ha^{-1}, respectively (Table 5.33).

Table 5.33 Belowground biomass and carbon in different diameter classes of sal (*Shorea robusta*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

DBH class (cm)	Belowground tree biomass (Mg ha-1)	(Mg C ha-1)	%
>5-10	2.5	1.2	3.4
>10-20	2.5	1.2	3.4
>20-30	4.9	2.3	6.8
>30-40	7.2	3.4	9.8
>40-50	21.5	10.1	29.4
>50-60	13.4	6.3	18.2
>60-70	13.5	6.4	18.5
>70-80	2.2	1.1	3.1
>80-90	1.3	0.6	1.7
>90-100	3.2	1.5	4.4
> 100	1.1	0.5	1.5
Total	73.2	34.4	100.0

Herb and Shrub biomass and carbon

The herb and shrub biomass were 13.4 and 44.1 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 6.31 and 21.1 kg C ha^{-1}, respectively. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (472 Mg ha^{-1}) in the forest contributing only 0.003 and 0.01%, respectively (Table 5.27).

Mixed bamboo forest

The following model was fitted to estimate AGB and BGB of bamboo species:

$$\text{Log}(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$$

where, Y= AGB (kg/culm), a, b, c, and d are regression coefficients, and D is the culm diameter at breast height. All the measures of coefficients of the models for culm components were statistically significant ($P< 0.001$). Models were evaluated by comparing coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE). The dry weights of various culm components of both the bamboo species were obtained for the development of culm component regression models (Table 5.34 and 5.35) across a range of DBH. The component-wise models i.e. regression coefficients, along with R^2, SD, SSE, MSE and RMSE for both the species are presented in Tables 5.36 and 5.37. The goodness of fit of the models has been depicted in Fig. 5.20 for *Dendrocalamus hamiltonii* and Fig. 5.21 for *Teinostachyum dullooa.*

Table 5.34 Dry weight (kg culm^{-1}) of different components of *Dendrocalamus hamiltonii* used for developing regression models (AGB=aboveground biomass)

DBH (cm)	Dry wt. (kg culm^{-1})					
	Culm	Leaves	Branch/ twigs	Total rhizome	Total AGB	Total culm biomass
5.0	8.5	0.0	0.1	0.6	8.7	9.2
5.0	9.2	0.1	0.1	0.6	9.3	9.9
5.1	9.3	0.1	0.1	0.6	9.4	10.1
5.3	9.7	0.1	0.1	0.6	9.9	10.5
5.4	10.5	0.1	0.1	0.6	10.7	11.3
6.1	11.0	0.1	0.1	0.7	11.1	11.8
6.1	11.7	0.1	0.1	0.7	11.9	12.6
6.2	13.3	0.1	0.1	0.7	13.5	14.2
6.2	14.0	0.1	0.1	0.7	14.1	14.8
6.3	14.4	0.1	0.1	0.7	14.6	15.2
6.5	15.8	0.1	0.1	0.7	15.9	16.6
6.7	15.9	0.1	0.1	0.7	16.1	16.8

6.7	16.3	0.1	0.1	0.7	16.5	17.2
6.8	16.8	0.1	0.1	0.7	16.9	17.7
6.8	16.8	0.1	0.1	0.7	17.0	17.7
7.3	17.2	0.1	0.1	0.7	17.3	18.1
7.6	17.2	0.1	0.1	0.8	17.4	18.1
7.6	17.6	0.1	0.1	0.8	17.8	18.5
8.4	20.1	0.1	0.1	0.8	20.3	21.1
9.1	21.1	0.1	0.1	0.8	21.3	22.1
9.1	21.1	0.1	0.1	0.8	21.3	22.1
9.2	21.2	0.1	0.1	0.8	21.4	22.2
9.2	21.2	0.1	0.1	0.8	21.4	22.2
9.3	21.2	0.1	0.1	0.8	21.4	22.2
9.3	21.7	0.1	0.1	0.8	21.9	22.8
9.6	18.1	0.1	0.1	0.8	18.3	19.1
9.6	18.2	0.1	0.1	0.8	18.3	19.1
9.6	18.2	0.1	0.1	0.8	18.4	19.1
9.6	21.7	0.1	0.1	0.8	21.9	22.8
9.6	21.8	0.1	0.1	0.9	22.0	22.8
9.6	21.9	0.1	0.1	0.9	22.2	23.0
9.6	21.9	0.1	0.1	0.9	22.2	23.0
9.6	20.5	0.1	0.1	0.8	20.7	21.5
9.6	18.8	0.1	0.1	0.8	18.9	19.7
9.6	22.5	0.1	0.1	0.9	22.7	23.6
9.7	22.5	0.1	0.2	0.9	22.7	23.6
9.7	19.4	0.1	0.1	0.8	19.6	20.4
9.7	19.5	0.1	0.1	0.8	19.7	20.5
9.7	23.0	0.1	0.2	0.9	23.2	24.1
9.7	23.0	0.1	0.2	0.9	23.2	24.1
9.8	24.1	0.1	0.2	0.9	24.4	25.2
10.2	20.5	0.1	0.1	0.8	20.7	21.5
10.3	27.1	0.1	0.2	0.9	27.4	28.2
10.3	20.0	0.1	0.1	0.8	20.1	20.9
10.5	27.8	0.1	0.2	0.9	28.0	28.9
10.6	28.5	0.1	0.2	0.9	28.8	29.6
10.9	28.6	0.1	0.2	0.9	28.8	29.7
11.3	29.1	0.1	0.2	0.9	29.4	30.3
11.7	30.1	0.1	0.2	0.9	30.3	31.2
12.2	33.1	0.1	0.2	0.9	33.4	34.3
12.2	34.6	0.1	0.2	1.0	34.8	35.8
12.8	36.1	0.1	0.2	1.2	36.4	37.6
12.8	39.7	0.1	0.2	1.2	40.0	41.2
13.2	39.8	0.1	0.2	1.3	40.1	41.4

Table 5.35 Dry weight (kg culm^{-1}) of different components of *Teinostachyum dullooa* used for developing regression models (AGB=aboveground biomass)

DBH (cm)	Dry wt. (kg culm^{-1})					
	Culm	Leaves	Branch/ twigs	Total rhizome	Total AGB	Total culm biomass
2.6	4.4	0.0	0.1	0.2	4.4	4.7
2.7	4.4	0.0	0.1	0.2	4.5	4.7
2.8	4.4	0.0	0.1	0.2	4.5	4.7
2.9	4.5	0.0	0.1	0.2	4.5	4.8
3.1	4.5	0.0	0.1	0.2	4.6	4.8
3.1	4.5	0.0	0.1	0.2	4.6	4.8
3.3	4.5	0.0	0.1	0.2	4.6	4.8
3.4	4.5	0.0	0.1	0.2	4.6	4.9
3.4	4.6	0.0	0.1	0.2	4.7	4.9
3.9	4.6	0.0	0.1	0.2	4.7	4.9
4.0	4.6	0.0	0.1	0.3	4.7	4.9
4.1	4.6	0.0	0.1	0.3	4.7	5.0
4.5	4.8	0.0	0.1	0.3	4.9	5.2
4.6	4.8	0.0	0.1	0.3	4.9	5.2
4.7	5.1	0.0	0.1	0.3	5.2	5.5
5.0	5.2	0.0	0.1	0.3	5.3	5.6
5.1	5.3	0.0	0.1	0.3	5.4	5.7
5.4	5.4	0.0	0.1	0.4	5.5	5.9
5.6	5.6	0.0	0.1	0.4	5.7	6.1
5.7	5.6	0.0	0.1	0.4	5.7	6.1
6.1	5.7	0.0	0.1	0.4	5.8	6.2
6.2	6.0	0.0	0.1	0.4	6.1	6.5
6.3	6.2	0.0	0.1	0.4	6.3	6.6
6.4	6.2	0.0	0.1	0.4	6.3	6.6
6.6	6.5	0.0	0.1	0.4	6.6	7.0
6.8	6.7	0.0	0.1	0.4	6.8	7.2

Table 5.36 Regression coefficients (a, b, c and d), coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) in respect of the models for biomass estimation of individual culm components and total biomass of *Dendrocalamus hamiltonii* Nees & Arn. ex Munro in Nongkhyllem wildlife sanctuary. The model is of the form Log(Y) = a + b logD + c (logD)^2 + d (logD)^3, where Y= biomass of individual culm components/BGB/AGB/total culm expressed in dry weight (kg culm^{-1}) and D= diameter at breast height (n=54). The model validity is between 5.0 cm and 13.2 cm DBH

Dependent variables (Y)	Coefficients				R^2	SD	SSE	MSE	RMSE
	a	b	c	d					
Culm	-12.8989	45.3283	-49.2392	18.1654	0.94	0.15	0.07	0.001	0.038
Leaves	-5.2862	13.8337	-16.2540	6.5358	0.84	0.08	0.06	0.001	0.034
Twigs	3.3254	-14.9528	16.2134	-5.4545	0.84	0.10	0.09	0.002	0.043
Rhizome	-6.5288	21.3274	-24.0058	9.1321	0.92	0.07	0.02	0.001	0.020
Total AGB	-12.7333	44.8105	-48.6870	17.9692	0.91	0.15	0.07	0.001	0.038
Total culm biomass	-12.3185	43.5935	-47.4304	17.5306	0.91	0.15	0.07	0.001	0.036

Table 5.37 Regression coefficients (a, b, c and d), coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) in respect of the models for biomass estimation of individual culm components and total biomass of *Teinostachyum dullooa* Gamble in Nongkhyllem wildlife sanctuary. The model is of the form Log(Y) = a + b logD + c (logD)^2 + d (logD)^3, where Y= biomass of individual culm components /BGB/AGB/total culm expressed in dry weight (kg culm^{-1}) and D= diameter at breast height , AGB= aboveground biomass (n=26). The model validity is between 2.6 cm and 6.8 cm DBH

Dependent variables (Y)	Coefficients				R^2	SD	SSE	MSE	RMSE
	a	b	c	d					
Culm	0.6013	0.5142	-1.6407	1.6136	0.98	0.059	0.001	0.001	0.007
Leaves	-3.3690	7.0859	-10.9693	5.9447	0.98	0.066	0.002	0.001	0.009
Twigs	-2.2056	4.8983	-7.6037	4.0858	0.98	0.039	0.001	0.002	0.006
Rhizome	1.1903	-9.9747	17.0343	-8.8480	0.97	0.096	0.007	0.003	0.019
Total AGB	0.5883	0.6079	-1.7712	1.6705	0.94	0.058	0.001	0.001	0.007
Total culm biomass	0.7136	0.0305	-0.7471	1.0980	0.94	0.060	0.001	0.004	0.006

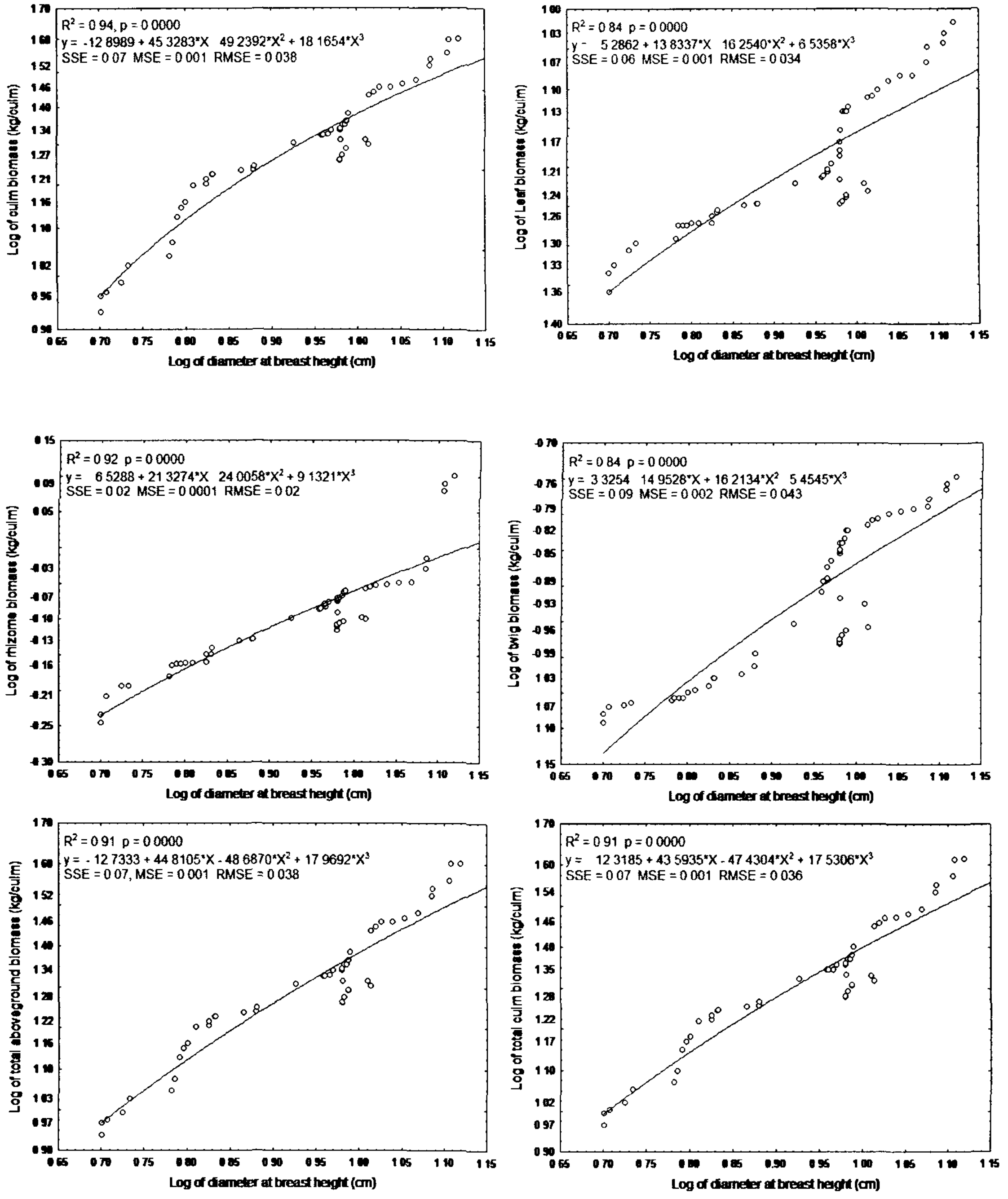


Fig. 5.20 Regression analyses between log of culm diameter at breast height (cm), and biomass of different culm components, total above ground biomass and total culm biomass of *Dendrocalamus hamiltonii* of mixed bamboo forest

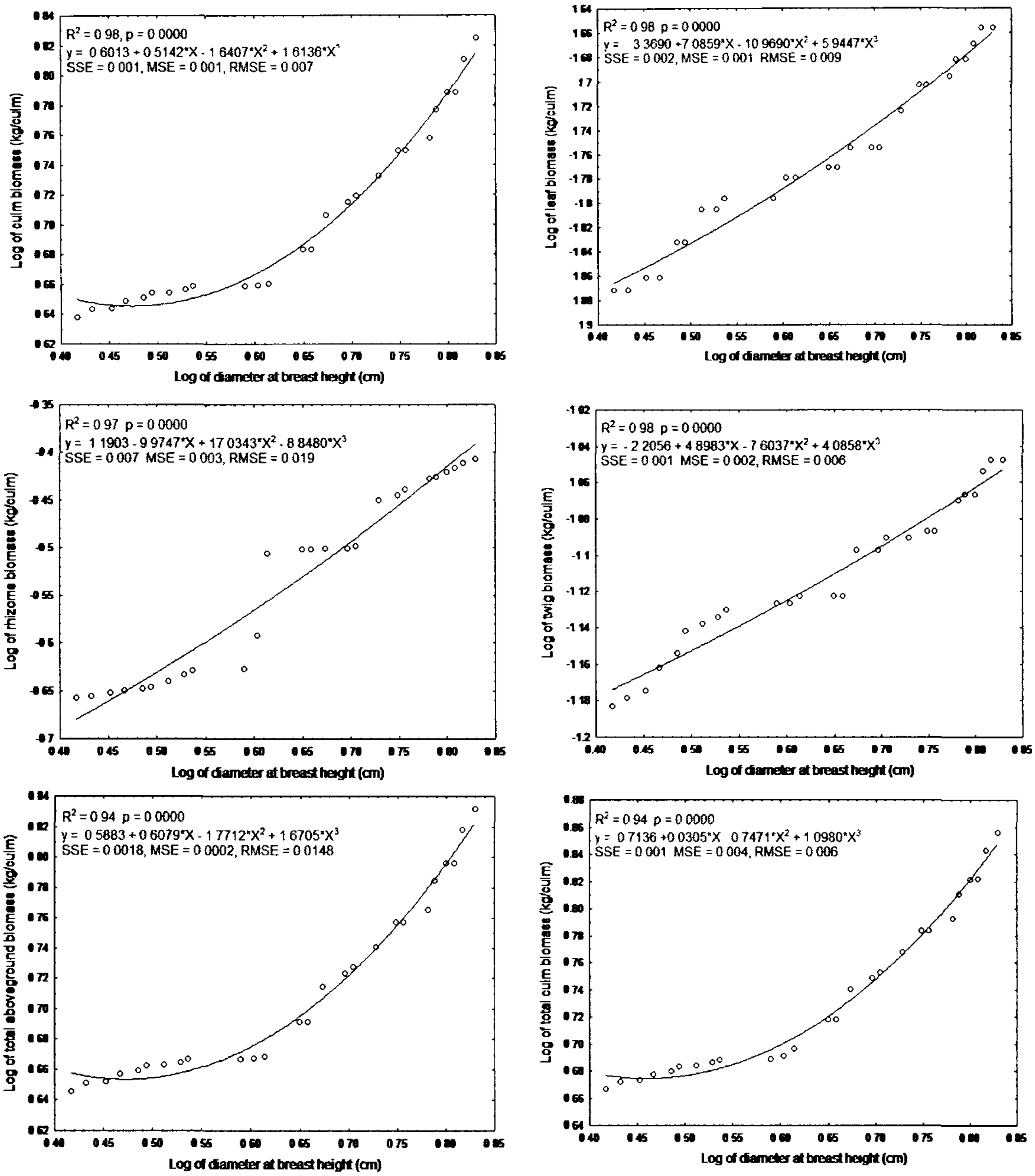


Fig. 5.21 Regression analyses between log of culm diameter at breast height (cm), and biomass of different culm components, total above ground biomass and total culm biomass of *Teinostachyum dullooa* of mixed bamboo forest

The mixed bamboo forest had both broad-leaved tree composition as well as bamboo. The allometric model of Chambers *et al.* (2001a) for broad-leaved tree aboveground biomass (AGB) was used to determine AGB of broad-leaved tree component in the mixed bamboo forest. The model was:

$$Y_1 = \exp [- 0.37 + 0.33*\ln (D) + 0.933*\ln(D)^2 - 0.122*\ln(D)^3]$$

Similarly, the model of Cairns *et al.* (1997) for broad-leaved tree belowground biomass (BGB) was to determine BGB of broad-leaved tree component in the forest. The model was:

$$Y_2 = \exp [- 1.0587 + 0.8836*\ln(AGB)]$$

where, Y= biomass/tree, D= diameter at breast height and AGB= aboveground biomass.

Density diameter distribution of bamboo and tree species

The density diameter distribution of trees in the mixed bamboo forest yielded a reverse J-shaped curve. The total tree density of the forest was 794 individuals ha^{-1}. The diameter class >10-20 cm had the maximum number of trees (270 individuals ha^{-1}) contributing 34% to the total tree density in the forest. The higher diameter class i.e. >60 cm had 53 trees contributing 6.7% to the total tree density in the forest. The highest diameter class >100 cm had only 10 individuals ha^{-1} contributing 1.3% to the total tree density in the forest (Fig. 5.22). The two bamboo species *viz.*, *Dendrocalamus hamiltonii and Teinostachyum dullooa* were under two diameter classes i.e. <10 cm and >10-15 cm. *Dendrocalamus hamiltonii* had 180 clumps ha^{-1} with density of 3420 culms ha^{-1} whereas, *Teinostachyum dullooa* had 44 clumps ha^{-1} with density of 1524 culm ha^{-1}. The <10 cm diameter class had 3238 culms ha^{-1} contributing 65.3% to the total bamboo culm density in the forest. The >10-15 cm diameter class constitutes only *Teinostachyum dullooa* with 1706 culms ha^{-1} and contributed 34.7% to the total bamboo culm density in the forest (Fig. 5.23).

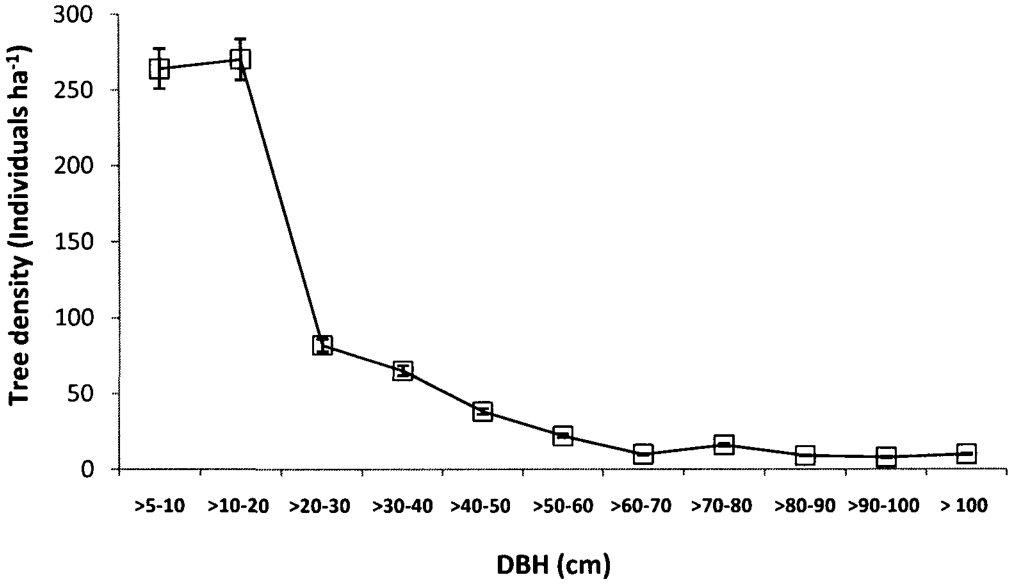


Fig. 5.22 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of mixed bamboo forest in tropical landscape of Nongkhyllem wildlife sanctuary

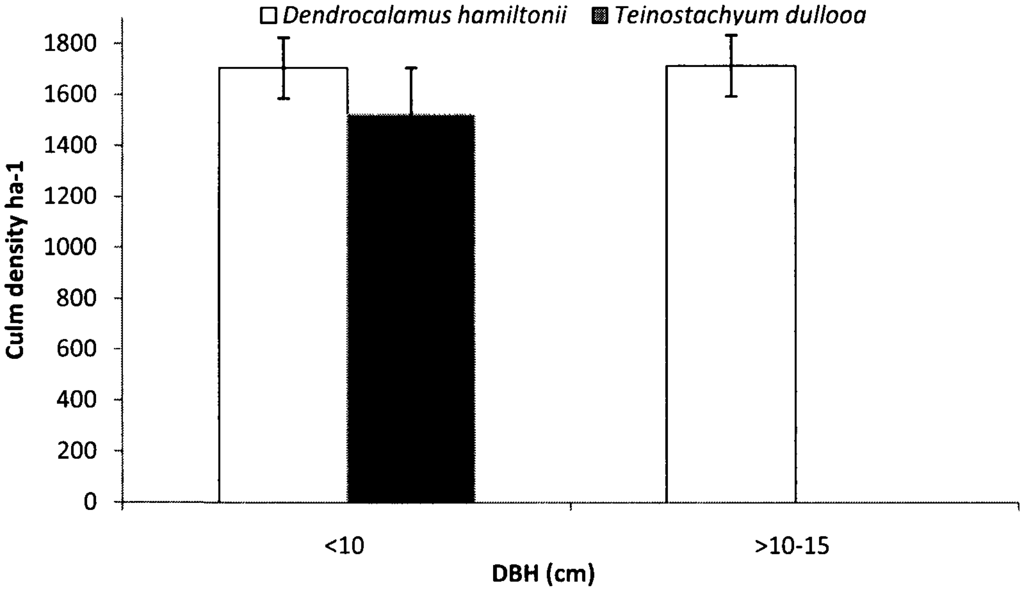


Fig. 5.23 Bamboo culm density distribution (culms ha^{-1}) in two diameter class (cm) of mixed bamboo forest in tropical landscape of Nongkhyllem wildlife sanctuary

Aboveground biomass and carbon in different diameter classes of trees and bamboos

The tree aboveground biomass in the mixed bamboo forest was high in the >100 cm diameter class. The higher diameter classes i.e. >60 cm contributed 52.4% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon were 273.4 Mg ha^{-1} and 134 Mg C ha^{-1}, respectively. The bamboo culm aboveground biomass in the mixed bamboo forest was high in the <10 cm diameter class. This diameter class contributed 79.1% to the total culm aboveground biomass in the forest. The total culm aboveground biomass and carbon were 87.8 Mg ha^{-1} and 43.9 Mg C ha^{-1}, respectively (Table 5.38).

Table 5.38 Aboveground biomass and carbon in different diameter classes of mixed bamboo forest in tropical landscape of Nongkhyllem wildlife sanctuary. A. Tree species biomass and B. Bamboo species biomass

A. Tree species aboveground biomass

DBH class	Aboveground tree biomass		
(cm)	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	7.8	3.8	2.8
>10-20	22.8	11.2	8.4
>20-30	19.8	9.7	7.2
>30-40	33.9	16.6	12.4
>40-50	24.6	12.0	9.0
>50-60	21.3	10.4	7.8
>60-70	14.9	7.3	5.5
>70-80	32.9	16.1	12.1
>80-90	15.2	7.4	5.6
>90-100	16.8	8.3	6.2
> 100	63.5	31.1	23.2
Total	273.4	134.0	100.0

B. Bamboo species aboveground biomass

DBH class	Bamboo biomass (Mg ha^{-1})				
(cm)	Culm	Leaves	Twigs	Total AGB	AGBC
<10	78.4	0.2	0.5	79.1	39.5
>10-15	8.5	0.0	0.1	8.7	4.4
Total	86.9	0.3	0.6	87.8	43.9

Belowground biomass and carbon in trees and bamboos

Tree belowground biomass in the mixed bamboo forest was high in the >40-50 cm diameter class. The higher diameter classes i.e. >60 cm contributed 45.1% to the total tree belowground biomass in the forest. The total tree belowground biomass and carbon were 42.0 Mg ha^{-1} and 19.7 Mg C ha^{-1}, respectively. The bamboo culm belowground biomass contributed 7.4% to the total belowground biomass in the mixed bamboo forest. The total culm belowground and carbon were 3.4 Mg ha^{-1} and 1.6 Mg C ha^{-1}, respectively (Table 5.39).

Table 5.39 Belowground biomass and carbon in different diameter classes of mixed bamboo forest in tropical landscape of Nongkhyllem wildlife. A. Tree species biomass and B. Bamboo species biomass

A. Tree species belowground biomass

	Belowground tree biomass		
DBH class (cm)	**(Mg ha^{-1})**	**(Mg C ha^{-1})**	**%**
>5-10	1.8	0.9	4.3
>10-20	4.6	2.2	11.1
>20-30	3.6	1.7	8.6
>30-40	5.8	2.7	13.7
>40-50	3.9	1.9	9.4
>50-60	3.3	1.5	7.8
>60-70	2.3	1.0	5.3
>70-80	4.7	2.2	11.3
>80-90	2.1	1.0	5.0
>90-100	2.3	1.1	5.5
> 100	7.5	3.5	18.0
Total	42.0	19.7	100.0

B. Bamboo species belowground biomass

DBH class (cm)	Belowground culm biomass		
	(Mg ha^{-1})	**(Mg C ha^{-1})**	**%**
<10	2.8	1.4	82.4
>10-15	0.5	0.2	14.7
Total	3.4	1.6	100.0

Herb and Shrub biomass and carbon

The herb and shrub biomass were the least (7.4 and 23.1 kg ha^{-1}) as compared to the total AGB of 367.7 Mg ha^{-1}. The corresponding figures for biomass carbon were 3.5 and 11.1 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass in the forest i.e. 0.002 and 0.006%, respectively (Table 5.27).

ESTIMATION OF CARBON FLUX IN TROPICAL FOREST ECOSYSTEMS

Soil respiration/CO_2 efflux

The highest soil CO_2 efflux of 2.3 mg CO_2 kg^{-1} day^{-1} was in mixed bamboo and sal plantation forest during summer season of 2005 and it was lowest in the old-growth broad-leaved forest (0.8 mg CO_2 kg^{-1} day^{-1}) during winter season of 2007 (Fig. 5.24). The CO_2 efflux was maximum during the summer season and minimum during winter season in all the forest types. The mean CO_2 efflux in the surface layer during three years period i.e. 2005-2008 was highest in the mixed bamboo forest (1.7 mg CO_2 kg^{-1} day^{-1}) and lowest in regenerating forest (1.1 mg CO_2 kg^{-1} day^{-1}). CO_2 efflux declined significantly ($p<0.01$) with the increase in the soil depth. Three-way ANOVA showed significant variation in CO_2 efflux due to forest types, season and depth (Table 5.40). CO_2 efflux in different forest types followed the order: TMBF> TSPF>TOBF> TTPF>TRBF.

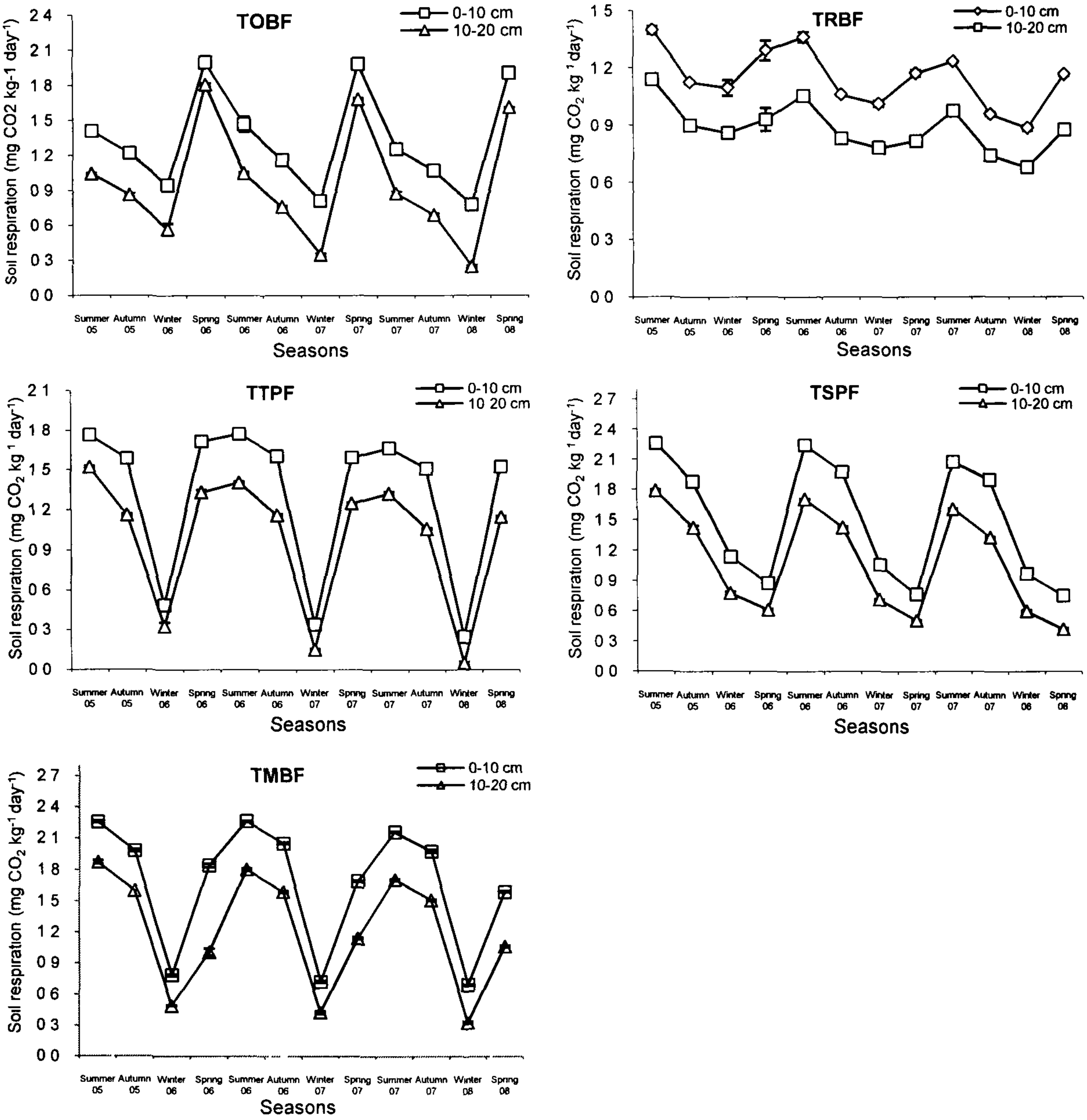


Fig. 5.24 Seasonal variation in the soil respiration (mg CO_2 kg^{-1} day^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary during 2005-2008. The values are mean (±SE) of five replicate samples

Table 5.40 Three-way ANOVA showing the effect of forest type, season and depth on soil respiration in different forest types in tropical landscape of Nongkhyllem wildlife sanctuary ($p<0.01$)**

Variation due to	df	SS	MS	F	p
Forest type	4	5.48	1.37	450.34**	0.000
Depth	1	2.66	2.66	874.74**	0.000
Season	11	13.17	1.20	393.47**	0.000
Forest type*Depth	4	0.17	0.04	14.11**	0.000
Forest type*Season	44	5.82	0.13	43.50**	0.000
Depth*Season	11	0.02	0.00	0.46	0.920
Forest type*Depth*Season	44	0.13	0.00	1.00	0.500

ESTIMATION OF ECOSYSTEM LEVEL BIOMASS AND NPP

Tropical old-growth broad-leaved forest (TOBF)

Total ecosystem biomass of the old-growth broad-leaved forest was 374.5 Mg ha^{-1}, of which 86.4% was in the aboveground compartment and 13.6% in the belowground compartment. Trees contributed 83.8%, herbs 0.003%, shrubs 0.01%, and litter 2.6% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 323.69 Mg ha^{-1}. The tree AGB and BGB were 313.8 and 50.8 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts accounted for 68.9, 14.1, 11.2 and 5.8%, respectively to the total litterfall. The total ecosystem NPP of the forest was 12.3 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed to 63.1, 6.3, 19.1 and 11.5% to the total litter production. The total ecosystem carbon content of the forest was 265.52 Mg C ha^{-1}. The soil organic carbon was 83.2 Mg C ha^{-1} contributing 31.3% to the total ecosystem carbon. The diameter class >40-50 and >60-70 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 47.7 Mg ha^{-1} and 47.9 Mg ha^{-1}, respectively. These two diameter classes accounted for 15.2% and 15.3 % of the total AGB of the forest. The aboveground NPP of the forest was 90.1%, while belowground NPP was 9.9% (Table 5.41).

Table 5.41 Total ecosystem, above and belowground biomass, carbon content and net primary production of tropical old-growth broad-leaved forest of Nongkhyllem wildlife sanctuary

Components	Biomass and carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Tree aboveground biomass	313.8	153.8	83.8	8.3	61.0
Herbs	0.01	0.01	0.003	0.002	0.01
Shrubs	0.04	0.02	0.01	0.004	0.03
Detrital biomass					
Leaves	6.8	3.2	1.8	2.5	18.3
Twigs	1.4	0.7	0.4	0.2	1.8
Branches	1.1	0.5	0.3	0.8	5.5
Misc.	0.6	0.3	0.2	0.5	3.3
Total detrital biomass	9.8	4.6	2.6	4.0	29.0
Total aboveground biomass	**323.7**	**158.4**	**86.4**	**12.3**	**90.1**
Tree belowground biomass	50.8	23.9	13.6	1.4	9.9
Total forest	374.5	182.3	100.0		
Total soil organic carbon		83.2			
Total Ecosystem	**374.5**	**265.5**	**100.0**	**13.6**	**100.0**
BNPP/NPP				0.1	9.9
ANPP/NPP				0.9	90.1

Regenerating broad-leaved forest

In the regenerating broad-leaved forest, total ecosystem biomass was 189.6 Mg ha^{-1}, of which 84% was in the aboveground compartment and 16% in the belowground compartment. Trees contributed 80.3%, herbs 0.01%, shrubs 0.02%, and litter 3.7% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 159.3 Mg ha^{-1}. The tree AGB and BGB were 152.3 and 30.3 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts accounted for 76.8, 19.3, 2.1 and 1.8%, respectively to the total litterfall. The total ecosystem NPP of the forest was 18.4 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed to 80.6, 12.4, 5.8 and 1.2% to the total litter production. The total ecosystem carbon content of the forest was 147.7 Mg C ha^{-1}. The soil organic carbon was 55.6 Mg C ha^{-1} contributing 37.6% to the total ecosystem carbon. The diameter class >5-10 cm and >10-20 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 81.5 Mg ha^{-1} that accounted for 53.5% of the total AGB of the forest (Table 5.31). The aboveground NPP of the forest was 87.5%, while belowground NPP was 12.5% (Table 5.42).

Table 5.42 Total ecosystem, above and belowground biomass, carbon content and net primary production of tropical regenerating broad-leaved forest of Nongkhyllem wildlife sanctuary

Components	Biomass and carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Tree aboveground biomass	152.4	74.7	80.3	12.5	68.1
Herbs	0.01	0.005	0.01	0.002	0.01
Shrubs	0.03	0.02	0.02	0.004	0.02
Detrital biomass					
Leaves	5.3	2.5	2.8	2.9	15.7
Twigs	1.3	0.6	0.7	0.4	2.4
Branches	0.1	0.1	0.1	0.2	1.1
Misc.	0.1	0.1	0.1	0.0	0.2
Total detrital biomass	6.9	3.3	3.7	3.6	19.4
Total aboveground biomass	**159.3**	**77.9**	**84.0**	**16.1**	**87.5**
Tree belowground biomass	30.3	14.2	16.0	2.3	12.5
Total forest	189.6	92.2	100.0		
Total soil organic carbon		55.6			
Total Ecosystem	**189.6**	**147.8**	**100.0**	**18.4**	**100.0**
BNPP/NPP				0.1	12.5
ANPP/NPP				0.9	87.5

Teak plantation forest

In the teak plantation forest, the total ecosystem biomass was 240.6 Mg ha^{-1}, of which 85.7% was in the aboveground compartment and 14.3% in the belowground compartment. Trees contributed 82.6%, herbs 0.004%, shrubs 0.02%, and litter 3.1% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 206.2 Mg ha^{-1}. The tree AGB and BGB were 198.8 and 34.4 Mg ha^{-1}, respectively. While *Tectona grandis* had 188.8 Mg ha^{-1} as AGB and 32.1 Mg ha^{-1} as BGB, the corresponding figures for other broad-leaved species were only 10 and 2.3 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts accounted for 82.2, 13.2, 3.6 and 1%, respectively to the total litterfall. The total ecosystem NPP of the forest was 9.6 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed to 40.9, 39.9, 15.2 and 3.9% to the total litter production. The total ecosystem carbon content of the forest was 159.2 Mg C ha^{-1}. The soil organic carbon was 41.7 Mg C ha^{-1} contributing 26.2% to the total ecosystem carbon. The diameter class >40-50 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 61.8 Mg ha^{-1} that accounted for 31.1% of the total AGB of the forest. The aboveground NPP of the forest was 87.5% as against the belowground NPP of 12.5% (Table 5.43)

Table 5.43 Total ecosystem, above and belowground biomass, carbon content and net primary production of tropical teak (*Tectona grandis*) plantation forest of Nongkhyllem wildlife sanctuary

Components	Biomass and carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Tree aboveground biomass of *Tectona grandis*	188.8	92.5	78.5	6.0	62.5
Tree aboveground biomass of other species	10.0	4.9	4.2	1.3	13.0
Herbs	0.01	0.005	0.004	0.002	0.02
Shrubs	0.04	0.02	0.02	0.004	0.04
Detrital biomass					
Leaves	6.1	2.8	2.5	0.5	4.9
Twigs	1.0	0.5	0.4	0.5	4.8
Branches	0.3	0.1	0.1	0.2	1.8
Misc.	0.1	0.0	0.0	0.0	0.5
Total detrital Biomass	7.4	3.5	3.1	1.2	12.0
Total aboveground biomass	**206.2**	**100.9**	**85.7**	**8.4**	**87.5**
Tree belowground biomass of *Tectona grandis*	32.1	15.4	13.4	0.9	9.8
Tree belowground biomass of other species	2.3	1.1	1.0	0.3	2.7
Total forest	240.6	117.4	100.0		
Total soil organic carbon		41.8			
Total Ecosystem	**240.6**	**159.2**	**100.0**	**9.6**	**100.0**
BNPP/NPP				0.1	12.5
ANPP/NPP				0.9	87.5

Sal plantation forest

The sal plantation forest had total ecosystem biomass of 545.2 Mg ha^{-1}, of which 86.6% was in the aboveground compartment and 13.44% in the belowground compartment. Trees contributed 84.6%, herbs 0.003%, shrubs 0.01%, and litter 1% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 472 Mg ha^{-1}. The tree AGB and BGB were 461.1 and 73.2 Mg ha^{-1}, respectively. While *Shorea robusta* had 320.8 Mg ha^{-1} as AGB and 49.9 Mg ha^{-1} as BGB, the corresponding figures for other broad-leaved species were 140.3 and 23.4 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts accounted for 80.6, 15.8, 2.3 and 1.2%, respectively to the total litterfall. The total ecosystem NPP of the forest was 14.4 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed to 72.6, 2.1, 18.8 and 6.5% to the total litter production. The total ecosystem carbon content of the forest was 314.4 Mg C ha^{-1}. The soil organic carbon was 48.2 Mg C ha^{-1} contributing 18.1% to the total ecosystem carbon. The diameter class >40-50 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 134 Mg ha^{-1} that accounted for 29.1% of the total AGB of the forest. The aboveground NPP of the forest was 88.4% as against the belowground NPP of 11.6% (Table 5.44)

Table 5.44 Total ecosystem, above and belowground biomass, carbon content and net primary production of tropical sal (*Shorea robusta*) plantation forest of Nongkhyllem wildlife sanctuary

Components	Biomass and carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Tree above ground biomass of *Shorea robusta*	320.8	157.2	58.8	6.2	43.2
Tree aboveground biomass of other species	140.3	68.8	25.7	4.6	32.2
Herbs	0.01	0.005	0.003	0.002	0.01
Shrubs	0.04	0.02	0.01	0.004	0.03
Detrital biomass					
Leaves	8.7	4.1	1.6	1.4	9.4
Twigs	1.7	0.8	0.3	0.0	0.3
Branches	0.3	0.1	0.1	0.4	2.4
Misc.	0.1	0.1	0.0	0.1	0.8
Total detrital biomass	10.8	5.1	2.0	1.9	13.0
Total aboveground biomass	**472.0**	**231.1**	**86.6**	**12.7**	**88.4**
Tree belowground biomass of *Shorea robusta*	49.9	23.9	9.2	0.9	6.1
Tree belowground biomass of other species	23.4	11.2	4.3	0.8	5.5
Total forest	545.2	266.2	100.0		
Total soil organic carbon		48.2			
Total Ecosystem	**545.2**	**314.4**	**100.0**	**14.4**	**100.0**
BNPP/NPP				0.1	11.6
ANPP/NPP				0.9	88.4

Mixed bamboo forest

The mixed bamboo forest had total ecosystem biomass of 413 Mg ha^{-1}, of which 89% was in the aboveground compartment and 11% in the belowground compartment. Trees contributed 66.2%, herbs 0.002%, shrubs 0.005%, and litter 1.6% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 367.7 Mg ha^{-1}. While bamboo culms had 87.8 Mg ha^{-1} as AGB and 3.4 Mg ha^{-1} as BGB, the corresponding figures for the broad-leaved species were 273.4 and 42 Mg ha^{-1}, respectively. The leaves, twigs and branches accounted for 63.2, 14.2 and 22.6%, respectively to the total litterfall. The total ecosystem NPP of the forest was 10 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed to 45.9, 4.1, 49.9 and 0.04% to the total litter production. The total ecosystem carbon content of the forest was 282.6 Mg C ha^{-1}. The soil organic carbon was 81.2 Mg C ha^{-1} contributing 28.7% to the total ecosystem carbon. The diameter class >100 cm had the highest tree biomass among all the diameter classes contributing maximum biomass of 63.5 Mg ha^{-1} that accounted for 23.2% of the total AGB of the forest. In the bamboo clumps the diameter class <10 cm class had AGB of 79.1 Mg ha^{-1}. The aboveground NPP of the forest was 89.9% as against the belowground NPP of 10.1% (Table 5.45).

Table 5.45 Total ecosystem, above and belowground biomass, carbon content and net primary production of tropical mixed bamboo forest of Nongkhyllem wildlife sanctuary

Components	Biomass and biomass carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Above ground biomass of bamboo culms					
Culm	86.9	42.6	21.1	0.2	2.1
Leaves	0.3	0.1	0.1	0.0	0.3
Twigs	0.6	0.3	0.1	0.0	0.2
Tree aboveground biomass	273.4	134.0	66.2	6.2	62.0
Herbs	0.01	0.005	0.002	0.001	0.01
Shrubs	0.02	0.01	0.006	0.002	0.02
Detrital biomass					
Leaves	4.1	2.0	1.0	1.2	11.6
Twigs	0.9	0.4	0.2	0.1	1.0
Branches/culm	1.5	0.7	0.4	1.3	12.6
Misc.	0.0	0.0	0.0	0.0	0.0
Total detrital biomass	6.5	3.1	1.6	2.5	25.2
Total aboveground biomass	**367.7**	**180.1**	**89.0**	**9.0**	**89.9**
Belowground biomass of bamboo culms	3.4	1.6	0.8	0.0	0.1
Tree belowground biomass	42.0	19.7	10.2	1.0	10.0
Total forest	413.0	201.4	100.0		
Total soil organic carbon		81.2			
Total Ecosystem	**413.0**	**282.6**	**100.0**	**10.0**	**100.0**
BNPP/NPP				0.1	10.1
ANPP/NPP				0.9	89.9

DISCUSSION

The estimation of forest aboveground biomass (AGB), is an important step in identifying the amount of carbon in terrestrial vegetation pools and is central to global carbon cycle studies (Drake *et al.* 2002). There are many uncertainties related to the global carbon budget and the role played by regenerating tropical forests in carbon sequestration and release is very important. Regenerating forests in the tropics are usually the result of agricultural land abandonment and increase in area of deforestation of mature forests (Vieira *et al.* 2003). The United Nations Framework Convention on Climate Change (UNFCCC) has recognized the importance of plantation forestry as a greenhouse gas mitigation option. The need to monitor, preserve and enhance terrestrial carbon stocks has been explained in the UNFCC (Updegraff *et al.* 2004). In addition, production from plantation forests may relieve pressure on timber extraction from natural forests, and thus contribute to forest conservation and management of carbon sinks.

Vegetation characteristics of the tropical forest ecosystem

The species richness was high in the old-growth broad-leaved forest (94 species), followed by mixed bamboo forest (73 species), sal plantation forest (67 species), regenerating broad-leaved forest (34 species) and the teak plantation forest (33 species). The tree density in the tropical landscape was high in the regenerating broad-leaved forest (1728 individuals ha^{-1}), followed by sal plantation forest (1104 individuals ha^{-1}), old-growth broad-leaved forest (1090 individuals ha^{-1}), teak plantation forest (864 individuals ha^{-1}) and mixed bamboo forest (794 individuals ha^{-1}). The species diversity in the old-growth broad-leaved forest was high as it was an undisturbed primary forest in the Nongkhyllem wildlife sanctuary and is well protected.

All the forest types showed clumped distribution pattern (91.0 – 94.5) due to site heterogeneity and 5.5 – 9.0% showed random distribution pattern. The clumped and random distribution pattern in the present study is in accordance to the findings of Richard *et al.* (1980) and Barik *et al.* (1992). Both the bamboo species exhibited 100% clumped distribution pattern.

Schima wallichii (IVI 25.9), *Actinodaphne obovata* and *Artocarpus chaplasha* (IVI 54.4 & 52.4), *Tectona grandis* (IVI 179.5), *Shorea robusta* (IVI 121.2) and *Shorea robusta* (IVI 37.4) were the dominant tree species in the old-growth broad-leaved, regenerating broad-leaved, teak plantation, sal plantation and mixed bamboo forests, respectively. The two species of bamboo *viz*., *Dendrocalamus hamiltonii* and *Teinostachyum dullooa* were the only bamboo species present in the mixed bamboo forest. The basal area of the forest was high in the sal plantation forest (93 m^2 ha^{-1}), followed by old-growth broad-leaved forest (65.1 m^2 ha^{-1}), mixed bamboo forest (61.8 m^2 ha^{-1}), teak plantation forest (38.3 m^2 ha^{-1}) and regenerating broad-leaved forest (9.9 m^2 ha^{-1}). The relatively high total basal area and low density is the characteristic feature of old growth forest. The Shannon's diversity and Simpson's dominance indices in the tropical landscape were high in the old-growth broad-leaved forest and followed the order: TOBF>TMBF>TSPF>TRBF>TTPF. The Species evenness index was highest in the regenerating broad-leaved forest and followed the order: TRBF>TMBF> TOBF>TSPF>TTPF. Alpha diversity was highest in the old-growth broad-leaved forest and followed the order: TOBF>TMBF>TSPF>TTPS>TRBF.

Soil properties as a function of forest ecosystem carbon

In the present study, the soil texture was primarily sandy in all the tropical forest types of Nongkhyllem wildlife sanctuary. The clay content was higher in the sub-surface soil layer whereas water holding capacity (WHC) was higher in the surface soil layer, which had greater accumulation of organic matter thereby indicating a stronger influence of SOC on WHC than the clay particles. The water holding capacity was higher than the reported value of 52 – 67% in the Kumaun Hilamalaya (Rikhari *et al.* 1991). The clay percentage ranged between 9.2 – 13.3% in the surface layer and 11.2 – 17.3% in the sub-surface layer. The silt percentage varied between 0.9 – 6.1% and 2 - 8.1% respectively, in the corresponding layers. The corresponding figures for sand ranged between 84.8 – 90.8 and 78.6 – 86.8%, respectively. These values were higher than the reported values by Sakin *et al.* (2011) who reported clay, silt and sand percentage to range between 3 - 31%, 21 - 41% and 36 - 66% from the tropical Harran plain soils. In all the tropical forest types the amounts of clay and silt increased with depth. The less clay and silt values in the tropical landscape were due to leaching and run off of the fine particles due to high rainfall.

Soil bulk density (BD) is an important indicator of soil compaction and the ability to function for structural support, water and solute movement, and soil aeration (Acosta-Martinez *et al.* 1999). The bulk density in the tropical forest types followed the order: TOBF>TMBF>TSPF>TTPF>TRBF. The increased bulk density has been attributed to the increase in sand and clay and soil organic matter percentage (Guerrero *et al.* 2000). To predict the change, flow and concentration of nutrients in the soils, bulk density (BD) is an important indicator (Bernoux *et al.* 1998). To estimate these pools correctly, the organic carbon contents and bulk density of the soils are pre-requisites (Sakin 2011). A strong negative correlation between BD and SOC was observed in the tropical landscape ($R^2 = 0.84$; $p = 0.002$) (Fig. 5.25). The SOC declined significantly with increase in BD ($p<0.01$). Li *et al.* (2007) also obtained a strong negative correlation between bulk density and organic carbon ($R^2 = -0.78$, $p<0.01$). Curtis and Post (1964) also stated that there was a reverse correlation between organic matter and bulk density.

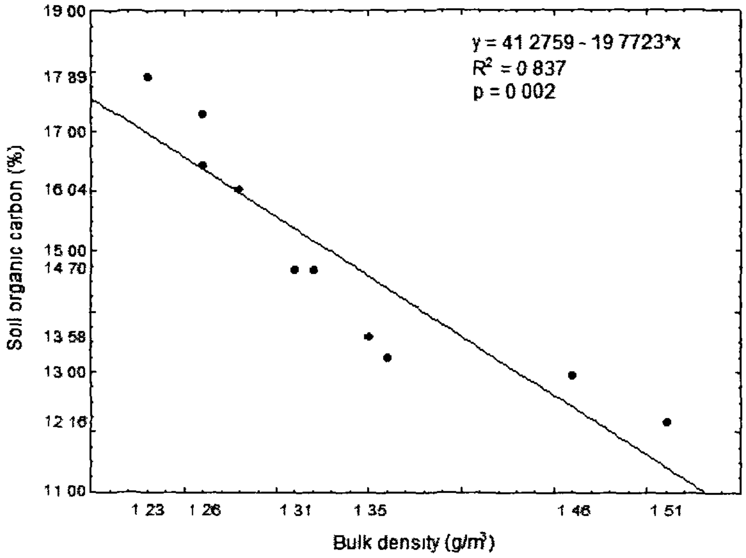


Fig. 5.25 Relationship between BD and SOC in tropical landscape of Nongkhyllem wildlife sanctuary

The SMC was low in the surface layer in all the tropical forest types. The lower SMC in the surface soil layer during the dry winter season could be the result of higher evapotranspiration from the soil and plant surfaces and percolation and infiltration of water to the lower depths (Tiwari *et al.* 1992).

The soils of the tropical forest type was acidic (pH = 4.2 - 6.3) in nature. Singh *et al.* (1995) recorded higher soil pH in jhum fallows than mixed bamboo forest and natural forests of north-eastern India. The acidity of the soil is due to the presence of exchangeable Al^{+3} and intensive leaching of bases (Fitzpatrick 2003). A drop in the pH during the rainy season could be due to the result of excessive leaching of basic cations by rainwater (Wild 1996).

The availability of soil nitrogen to plant is largely determined by the mineralization process during decomposition of organic matter (Jordan 1985; Swift *et al.* 1979) and thereby affecting tree growth (Yamakura and Sahunalu, 1990). Higher concentration of TKN in the surface and sub-surface layer of all the tropical forest types could be due to higher organic matter concentration. A sharp decline in TKN value from 0.22% in regenerating broad-leaved forest to 0.18% in teak plantation in the tropical landscape may be attributed to runoff losses caused by heavy rainfall, besides low SOM content. The highest concentration of TKN during winter season as observed in the present study for all the tropical forest types were in accordance with the findings of Kamei *et al.* (2007).

Available phosphorus in the soils of north-eastern region of the country is often present at very low concentration. Available P was low and showed a significant ($p < 0.01$) variation due to forest type, season and depth. The SOC did not show a positive correlation with available P in the tropical landscape. Therefore, no significant role of organic matter in P availability was observed. Several workers (Saikh *et al.* 1998; Singh 2002) did observe significant seasonal variation in available Phosphorus contents in the regenerating jhum fallows, cultivated fields, grasslands and natural forests.

In the tropical soils, the total K content may be quite low because of the ontogeny of the soils, high rainfall and high temperature (Yawson *et al.* 2011). High exchangeable K (454.2 $\mu g\ g^{-1}$) was observed in the old-growth broad-leaved forest during winter season and low in the regenerating broad-leaved forest (117.5 $\mu g\ g^{-1}$) during summer season. Potassium concentration declined further in the next rainy season in all the tropical forest types due to excessive leaching and runoff losses. A greater fluctuation in the concentration of K than the other nutrients was observed. This is due to the fact that K cycles through vegetation and soil, solely acts as unbound ion, and is easily leached from living and decomposing plant tissues compared to other nutrients. The exchangeable K concentration in the surface soil layer of the old-growth broad-leaved forest and regenerating broad-leaved forest of tropical landscape was higher because K was retained and cycled dynamically in the forest ecosystem (Bradley *et al.* 2001). Exchangeable Potassium showed a positive correlation with SOC ($R^2 = 0.415$; $p < 0.05$) (Fig. 5.26).

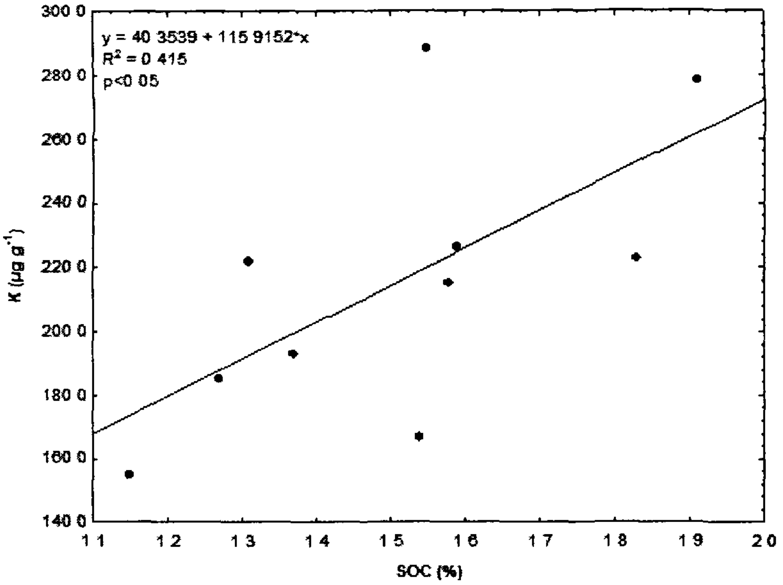


Fig. 5.26 Relationship between SOC and K in tropical landscape of Nongkhyllem wildlife sanctuary

Soil carbon pools are not only important to governing soil properties and nutrient cycling in forest ecosystems, but also play a critical role on forest productivity. Additionally, soil C pools form a large and dynamic reservoir of C, which is an important part of the global C cycle and a potential sink for atmospheric CO_2. Due to the important role of soil C pools in nutrient cycling of forest ecosystems and global C balance, there has long been an interest in understanding the effect of forest soil management on soil C pools (Ussiri and Johnson, 2007). In strongly seasonal climate where decomposition rate is fast, litter protects the soil surface throughout the year and promotes organic matter accumulation (Brown *et al.* 1994). Significant reduction in soil organic matter and organic carbon has been reported by Brown *et al.* (1994), Henrot and Robertson (1994) and Guo and Gifford (2002) following conversion of tropical forest into pastures, agricultural fields, and shifting cultivation plots. Loss could even reach upto 50% in organic matter and total nitrogen content in comparison to the undisturbed natural forest sites (Hajabbasi *et al.* 1997; Saikh *et al.* 1998).

The favourable effects of SOM on the physico-chemical properties of soil on biological activity in sustaining soil productivity are well known. Soil has a stabilizing effect on soil structure, improves the moisture retention and release characteristics of soil, and protects soil against erosion. Decomposing organic matter releases nutrients such as N, P, K and S, essential for plant and microbial growth. Soil organic matter is further an important determinant of the cation exchange capacity, particularly in coarse textured soils and in 'low activity clay' soils.

The amount of soil organic carbon in the old-growth broad-leaved forest was 83.2 Mg C ha^{-1}, followed by mixed bamboo forest (81.2 Mg C ha^{-1}), regenerating broad-leaved forest (55.9 Mg C ha^{-1}), sal plantation forest (48.2 Mg C ha^{-1}) and teak plantation forest (41.7 Mg C ha^{-1}). The values for old-growth broad-leaved forests (83.2 Mg C ha^{-1}) and mixed bamboo forests (81.2 Mg C ha^{-1}) are comparable with the findings (72 – 149 Mg C ha^{-1}) of Glaser *et al.* (2003) for the Amazonian rain forest near Belterra and of Lu *et al.* (2010) in the tropical seasonal forest of China (84 to 102 Mg C ha^{-1}).

The values were higher than the reported values (34 – 56 Mg C ha^{-1}) by Sombroek *et al.* (1993) for sandy to clay soil of Amazonian forests. The present SOC values are less than the values obtained by earlier workers for the tropical forests of Asia. For example, Dixon *et al.* (1994) reported a value of 139 Mg C ha^{-1} in the tropical forests of Asia and Glaser *et al.* (2003) reported the value of 147 - 506 Mg C ha^{-1} in Amazonian rain forests near Manaus. IPCC (2000) reported average SOC value of 86 Mg C ha^{-1} for the tropical forests of the world which coincides with the present findings. The SOC in sal plantation forest with 48.2 Mg C

ha^{-1} and that in teak plantation forest with 41.7 Mg C ha^{-1} were less than the earlier reported values of 119.4 Mg C ha^{-1} and 123.3 Mg C ha^{-1} for the sal and teak forests, respectively in India (Jha *et al.* 2003). The SOC values for sal, teak plantation and regenerating broad-leaved forests are comparable with the findings of Sombroek *et al.* (1993) (34 – 56 Mg C ha^{-1}) for sandy to clay soil of Amazonian forests. Changing patterns of land-use and land-use management practices can have significant direct and indirect effects on soil organic pools, due to changes in plant species, primary productivity, litter quantity and quality and soil structure (Schwendenmann *et al.* 2007).

The soil organic matter upto 1 m depth in the old-growth broad-leaved forest was highest (143.5 Mg ha^{-1}), followed by mixed bamboo forest (140.1 Mg ha^{-1}), regenerating broad-leaved forest (95.8 Mg ha^{-1}), sal plantation forest (83.1 Mg ha^{-1}) and teak plantation forest (72 Mg ha^{-1}). The soil organic matter obtained in the present study was within the reported value of 162 Mg ha^{-1} for tropical soils (Malhi *et al.* 1999). The Soil organic matter is a major factor in ecosystem functioning and determines whether soils act as sinks or sources of carbon in the global carbon cycle (Brown and Lugo, 1982; Smithson *et al.* 2002).

The C/N ratio and N concentration of the forest floor as well as in situ N mineralization and nitrification rate were identified as key indicators of the ecosystem's nitrogen status (Andersson *et al.* 2002; Chen and Mulder, 2007; Swift *et al.* 1979). Soil C/N ratio is an index of N mineralization, decomposition and quality of the organic matter in soil (Bengtsson *et al.* 2003; Springob and Kirchmann, 2003). Large C/N ratio resulted from low mineralization rates and consequently their levels of total N were low. The soil C/N ratio varied among the forest types and increased with depth in all tropical forest types, except for the sal plantation forest, where it showed a reverse trend due to less SOC in the sub-surface layer. The high

C/N ratio in the mixed bamboo forest promotes faster rate of decomposition (k = 0.06), as is evident from litter decomposition studies. The higher C/N ratios in deeper soil layers are mainly due to the decreasing organic C concentration along the soil profile. The C/N ratios in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers of tropical landscape ranged between 7.1 – 9.6 and 7.3 – 10.3, respectively. These values are comparable with the reported global values of C/N in the tropical forests (9.2 – 30.2). Relatively large amounts of N in surface soil in tropical landscape are associated with recalcitrant humic material in advanced state of decay with low C/N (Post *et al.* 1985).

Seasonal and spatial dynamics of microbial biomass carbon (MBC)

The concentration of MBC obtained in the present study is within the reported range (61-2000 $\mu g\ g^{-1}$) for various temperate and tropical forest soils (Vance *et al.* 1987; Henrot and Robertson, 1994; Diaz-Ravina *et al.* 1995). The low value during winter is in accordance with the findings of Lynch and Panting (1982) and Sarathchandra *et al.* (1989). Higher values of microbial biomass during summer (rainy) season have also been reported from pasture and dry tropical forest soils (Sarathchandra *et al.* 1984; Singh *et al.* 1989). Piao *et al.* (2000) reported greater accumulation of soil MBC in the winter with drying-rewetting cycles not frequently occurring. In our case, minima and maxima of SMC and MBC were observed during winter and rainy season respectively. These results are in conformity with those of Singh (2002) who reported higher MBC during warm-wet months and lower during dry-cold period in the undegraded forest of Arunachal Pradesh. The MBC showed a strong positive correlation with SMC (R^2=0.9, p<0.001) (Fig. 5.27). Our results are consistent with previously reported studies (Arunachalam and Arunachalam, 2000; Wright *et al.* 2005).

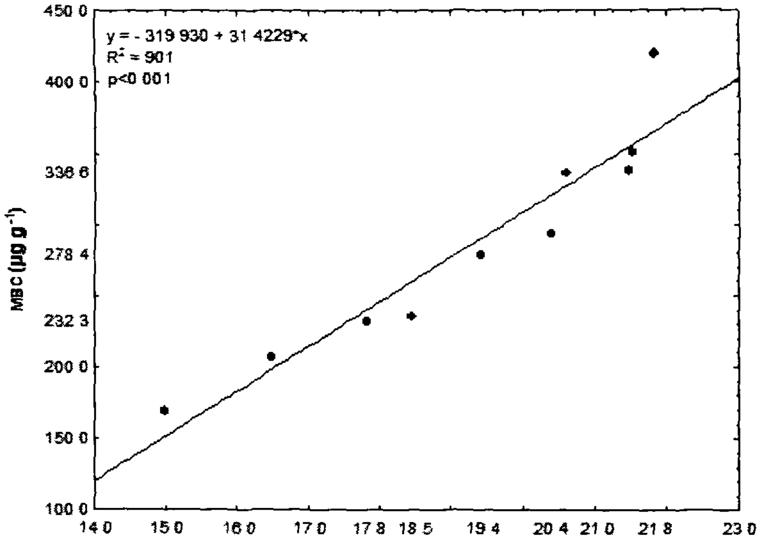


Fig. 5.27 Relationship between SMC and MBC in tropical landscape of Nongkhyllem wildlife sanctuary

Contribution of microbial biomass carbon to SOC

The MBC expressed as percentages of SOC gives estimates of the quantities of carbon present in the microbial biomass, substrate availability and organic matter dynamics in soils (Sparling 1992). In the present study, the contribution of MBC to soil organic carbon in the surface soil layer during summer season (2% to 4.9%) in tropical landscape was within the reported range for tropical forests. For example, Theng *et al.* (1989) and LuiZao *et al.* (1992) reported a range of 1.5 - 5.3% for the tropical forests of central Amazon, Ralte (2004) reported 0.51 – 8.69% for tropical forest of Nokrek Biosphere reserve in Meghalaya and Vance *et al.* (1987) 1.8 – 2.9 % for temperate forest soils. However, in the present study, the percentage contribution of MBC to SOC in the tropical landscape suggests that microbial communities in these forest types have evolved a more complex system of substrate-use. The relatively dense structure of plants had a greater accumulation of litter and fine roots in the understory of forest and may favor the growth of microbial populations and the accumulation of C in microbial biomass. Our results are consistent with previously reported studies

(Arunachalam and Arunachalam, 2000; Sharma *et al.* 2004; Wright *et al.* 2005). The SOC showed a strong positive correlation with MBC (R^2=0.9, p<0.001) (Fig. 5.28) in the tropical landscape.

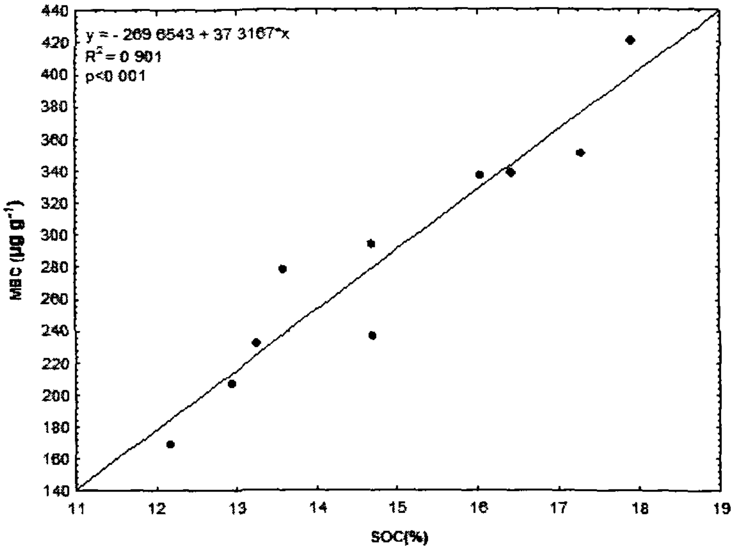


Fig. 5.28 Relationship between SOC and MBC in tropical landscape of Nongkhyllem wildlife sanctuary

Litter production and accumulation

The total annual litterfall obtained in the different forest types (6.5 – 10.8 Mg ha^{-1}) was within the reported range for various tropical forests (2.2 - 22.6 Mg ha^{-1}) (Vogt *et al.* 1986), Cuveas and Medina (1986) (2.4 - 10.3 Mg ha^{-1}) for the Amazonian tropical forest, and 5.5 Mg ha^{-1} for the tropical forest of Meghalaya (Singh 1980). The values were however lower (13 – 15 Mg ha^{-1}) than the moist deciduous forest of Indian Western Ghats (Swamy and Procter, 1994). The litterfall values obtained in the present study was within the reported range of 6.8 to 10 Mg ha^{-1} yr^{-1} for the humid tropical forest of West Africa (Cornforth 1970) and three Colombian forests (Folster and Salas, 1976). The high litter production in the sal plantation forest may be because of high density, basal area, leaf size and leaf dry weight. The study by Singh (1980) in the mature forest of Nongkhyllem reserve forest also showed a similar result, where sal trees contributed maximum litter. The life span, size and weight of leaf play an important role in the litter production (Singh 1980). In the present study, the proportion of leaf (63 - 82%) in the litterfall of tropical forest types was similar to the values reported by Meentemeyer *et al.* (1982) (70%) and for the tropical rainforest in Malaysia (71%) and Venezuela (74%) by Haase (1999).

The litter accumulation values obtained in the present study (2,172 – 5,865 kg ha^{-1}) are close to the reports from tropical dry evergreen forest (4,100 - 4,900 kg ha^{-1}) by Pragasan and Parthasarathy (2005), tropical forest (3,800 - 5,500 kg ha^{-1}) by Sundarapandian and Swamy (1999) and semi deciduous forest of Brazil (5,500 kg ha^{-1}) by Morellato (1992). However, the values were lower than the tropical wet evergreen forest and semi-evergreen forest of Western Ghats, India (11,700 and 10,400 kg ha^{-1}) (Parthasarathy 1992). The findings were in consistence with the study of Vogt *et al.* (1986), where rapid decomposition of litter has been attributed as an important cause of lower litter accumulation on the forest of wet tropical forest. Pargasan and Parthasarathy (2005) also reported that the significant differences in total litter accumulation between two tropical dry evergreen forest sites on the Coromandel coast of south India, was related to differences in temperature, soil moisture and decomposition rates besides vegetation variables.

Litter decomposition in the tropical landscape

Litter decomposition plays a crucial role in the nutrient budget of the tropical forest ecosystems where vegetation depends mainly on the recycling of nutrients present in the plant detrital matter (Vogt *et al.* 1986). The decomposition rate is strongly influenced by climatic conditions and initial chemical composition of the litter (Couteaux *et al.* 1995). Among the climatic variables, rainfall and air temperature determine the rate of decomposition in areas subjected to unfavourable weather conditions (Arunachalam *et al.* 1998b)

Litter decomposition in forests of the humid tropics is generally perceived as a rapid process, because of favorable climatic conditions for microbial activity. The tropical forest types showed a rapid phase of decomposition followed by stabilization. The faster initial phase lasted for about 3 months (June – August) where environmental conditions were most favourable for decomposition. The faster initial phase of decomposition reflects the rapid process of decomposition and leaching of soluble compounds (Loranger *et al.* 2002; Swift *et al.* 1979). After the initial rapid phase, decomposition rate rather appears to be slow and constant. The faster rate of decomposition in the mixed bamboo forest may be attributed to high C/N ratio, soil moisture, temperature and microbial biomass which aids in faster decomposition of litter on the forest floor. The decomposition rates (k=0.06 – 0.12) of leaf litter fall below the reported range for semi-evergreen tropical forest of Grande-Terre, Guadeloupe, French West Indies (k= 0.41-2.39; Loranger *et al.* 2002), tropical forest of Nokrek Biosphere Reserve in Meghalaya (k=0.88-1.76; Ralte 2004). The species richness can influence decomposition by impacting the quality of litter and microclimate of the decomposing litter (Kamei *et al.* 2009; Knops *et al.* 2001).

Carbon flux in the tropical forest ecosystem

Soil respiration has gained much attention as it is recognized as a major soil carbon efflux and one of the key components of the carbon cycle in terrestrial ecosystems (Raich and Schlesinger, 1992). Soil respiration represents about 40–90% of the forest ecosystem respiration (Schlesinger and Andrews, 2000). Soil respiration is a critical component in the carbon cycle processes as well as the biogeochemical cycles. Inputs and removal of aboveground (leaves, twigs, seeds, other fine litters) and belowground (mostly fine roots) source of soil organic matter influence decomposition and soil respiration rates. Soil CO_2 efflux value was highest in the mixed bamboo forest (1.7 mg CO_2 kg^{-1} day^{-1}), followed by sal plantation forest (1.5 mg CO_2 kg^{-1} day^{-1}), old-growth broad-leaved forest (1.3 mg CO_2 kg^{-1} day^{-1}), teak plantation forest (1.3 mg CO_2 kg^{-1} day^{-1}) and regenerating broad-leaved forest (1.1 mg CO_2 kg^{-1} day^{-1}). These values are higher than the values reported by Aghasi *et al.* (2011) (0.02 - 0.1 mg CO_2 $kg^{-1}day^{-1}$).

The microbial activity in the soil was at its maximum during summer season when favourable conditions were at its best and lowest during the winter season. The decrease in soil respiration values during winter season is in accordance with the report of Mo *et al.* (2005). The peaks in microbial biomass values during June - August can be attributed to favourable soil moisture and temperature that supported microbial activities and proliferation.

Higher CO_2 efflux rate in both the plantation forest as compared to the old-growth broad-leaved forests could be due to the organic and inorganic amendments which might have supported more microbial growth and utilization of the available substrates by altering the respiratory quotient of microbes (Anon *et al.* 2001). Higher soil moisture and humidity in both the plantation forests could also have account for the high CO_2 efflux rate. Recently, Tian *et al.* (2009) reported strong effects of soil temperature and soil moisture content on soil CO_2 efflux in Chinese fir plantations. The lower soil CO_2 efflux rate in both the old-growth and regenerating broad-leaved forests was because of increased runoff loss due to heavy rainfall during the study period that may have carried the microbial propagules leading to decreased microbial biomass. The soil CO_2 efflux rate is mainly influenced by the microclimatic variables which are affected by vegetation cover and soil management practices (Piao *et al.* 2000). The CO_2 efflux in the different forest types was in the order: TMBF>TTPF>TSPF>TOBF>TRBF.

Soil CO_2 efflux showed a positive correlation with microbial biomass carbon in the tropical ($R^2 = 0.65$; $P<0.01$) (Fig. 5.29). The findings are in support of positive correlation found between CO_2 efflux and microbial biomass in the forests of Puerto Rico by Li *et al.* (2005).

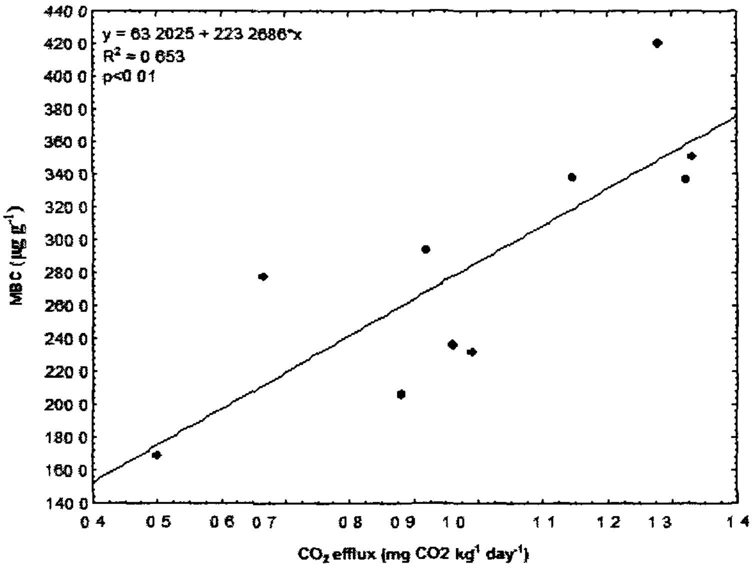


Fig. 5.29 Relationship between CO_2 efflux and microbial biomass carbon in the tropical landscape of Nongkhyllem wildlife sanctuary

Biomass and carbon sequestration potential of the tropical forest ecosystems

The potential of forests to sequester carbon depends on the forest type, age of forest and size class of trees (Terakunpisut *et al.* 2007). The observed AGB value of 323.7 Mg ha^{-1} in the natural old-growth broad-leaved forest and 368 Mg ha^{-1} in the mixed bamboo forests are comparable with the findings of Ramachandran *et al.* (2007) who reported a value of 307 Mg ha^{-1} for the tropical evergreen forests of eastern coast of Tamil Nadu, India and Hase and Foelster (1983) for the mature forests of West Venezuela (398 Mg ha^{-1}) and 362 Mg ha^{-1} for the tropical forests of Asia (Houghton 2005). The present AGB values for the old-growth broad-leaved and mixed bamboo forests are within the reported range for the primary rainforests of Southeast Asia, which ranged from 300 - 500 Mg ha^{-1} (Kato *et al.* 1978; Kira and Shidei, 1967; Laumonier *et al.* 2010; Yamakura *et al.* 1986). The present value is however less than the AGB values of 607.7 Mg ha^{-1} and 468 Mg ha^{-1} reported for tropical wet evergreen forest and tropical semi-evergreen forest of Western Ghats of India by Rai (1981) and Swamy (1989), respectively. The AGB value was also close to those reported by Muller (1982) for the tropical forests of eastern hardwood region of USA (330 Mg ha^{-1}) and by Brown and Lugo (1982) for the tropical rain forests in Malaysia (225 - 446 Mg ha^{-1}) and Cameroon (238 - 341 Mg ha^{-1}). The value range of 153 - 221 Mg ha^{-1} reported from Sri Lankan tropical rain forests (Brown and Lugo, 1982) was lower than the values found in the present study. Terakunpisut *et al.* (2007) reported an AGB value of 96 - 276 Mg ha^{-1} for the tropical rain forests of Thailand which was also less than the present study. The lower AGB value of 170 Mg ha^{-1} for the broad-leaved forests of tropical America, 260 Mg ha^{-1} for tropical Africa, 215 Mg ha^{-1} for tropical Asia and 150 Mg ha^{-1} for total tropics were reported by Brown and Lugo (1984). However, the AGB value obtained for this northeastern Indian tropical forest was much lower than the highest AGB value reported so far i.e. 500 - 600 Mg ha^{-1} for the undisturbed deciduous forest in the southern Appalachain Mountains (Whittaker 1996).

The AGB of 472 Mg ha^{-1} obtained for the sal plantation forest in the present study was within the earlier reported ranges of 337 - 698 Mg ha^{-1} estimated for sal-dominated forest in Central Nepal (Shrestha *et al.* 2000). However, it was much higher than those estimates of 304 Mg ha^{-1} for the forests of Uttar Pradesh (Singh *et al.* 1992) and 261 Mg ha^{-1} for a 10 year old recovering tropical sal forest of Eastern Ghats, India (Behera and Misra, 2006). The total AGB of 206.2 Mg ha^{-1} obtained in the teak plantation forest is within the reported range (84 - 284 Mg ha^{-1}) by Cordero and Kanninen (2003) and (85.7 Mg ha^{-1}) for teak forests of Colombia by Usuga *et al.* (2010). The values were higher than the 20 years old teak plantation in Tripura (138 Mg ha^{-1}) by Negi *et al.* (1990), 28 - 85.3 Mg ha^{-1} in the dry deciduous forest of Madhya Pradesh by Pande (2005), but lower than the 18 years old teak plantation forest (378 Mg ha^{-1}) reported by Ola-Adams (1990) from South-Western Nigeria. The AGB of 159.3 Mg ha^{-1} obtained in the regenerating broad-leaved forest was higher than the 20 years (118 Mg ha^{-1}) and lower than 30 years old (177 Mg ha^{-1}) tropical secondary forest of Ecuador (Fehse *et al.* 2002). The aboveground biomass of regenerating broad-leaved forest (20 – 22 years) in the present study was within the range for various tropical secondary forests (14 – 272 Mgha^{-1}) of 15 – 20 years old (Hashimoto *et al*, 2000; Marin-Spiotta *et al.* 2008).

Previous studies estimated high potential of carbon storage in Indian forests, especially through raising long rotation plantations (Baishya *et al.* 2009; Bhadwal and Singh, 2002; Hooda *et al.* 2007; Lal and Singh, 2000; Manhas *et al.* 2006). Baishya *et al.* (2009) compared the carbon storage potential of natural semi-evergreen forest and sal plantation forest in the humid tropical region of northeast India. Their results suggest that the old-growth broad-leaved forest had lower aboveground biomass (323.7 Mg ha^{-1}) than the sal plantation forest (472 Mg ha^{-1}). Although both the forests had the potential for greater carbon sequestration but the sal plantation forestry had an edge over the natural old-growth broad-leaved forest because of better silvicultural practices (Baishya *et al.* 2009).

The amount of AGB carbon stored in the old-growth broad-leaved forest (158.4 Mg C ha^{-1}) and in the mixed bamboo forest (180.1 Mg C ha^{-1}) in the present study were greater than the tropical forests of Sri Lanka (77 Mg C ha^{-1}) by Brown and Lugo (1982), the primary Amazon forest (110 Mg C ha^{-1}) by Phillips *et al.* (1998) and tropical forests of Asia (127 Mg C ha^{-1}) by Houghton (2005), but lower than the relatively undisturbed matured tropical rain forest of Malaysia (223 Mg C ha^{-1}) reported by Brown and Lugo (1982). The present value for old-growth broad-leaved forest and mixed bamboo forest are in conformity with the findings of Clark *et al.* (2001b) for the tropical old-growth forest of Ivory Coast (151.5 - 256.5 Mg C ha^{-1}) and 151 - 203 Mg C ha^{-1} for Brazil. The values were however lower than the reported values 324.5 Mg C ha^{-1} for India by Clark *et al.* (2001b), 123.5 Mg C ha^{-1} for the Amazonian forest by Girardin *et al.* (2010), and 48 - 138 Mg ha^{-1} for the tropical rain forests of Thailand by Terakunpisut *et al.* (2007). Ogawa *et al.* (1965) reported a carbon stock of 60 to 179 Mg C ha^{-1} in different tropical forest types of Thailand. Flint and Richards (1996) estimated carbon sequestration in Southeast Asia including India, Thailand, Cambodia, Malaysia and Indonesia, and reported the value ranged from 17 Mg C ha^{-1} in severely degraded tropical dry forest to 350 Mg C ha^{-1} in the undisturbed matured tropical rain forests. The AGB carbon in the regenerating broad-leaved forest (77.9 Mg C ha^{-1}) is comparable with the findings of Brown and Lugo (1982) in the disturbed tropical forests of Sri Lanka (77 Mg C ha^{-1}). The aboveground biomass carbon of 100.9 Mg C ha^{-1} obtained in the teak plantation forest was within the reported values of 86.8 - 122.2 Mg C ha^{-1} by Kranzel *et al.* (2003) for the matured teak plantation forests of Panama and more than the values 2.9 - 40.7 Mg C ha^{-1} reported by Derwisch *et al.* (2009) for 1-10 year old teak plantation of Western Panama and teak forest of dry tropical forest of Madhya Pradesh, India

(28 – 85 Mg C ha^{-1}) by Pande (2005). The aboveground biomass carbon obtained in the sal plantation forest (231.1 Mg C ha^{-1}) was more than the sal forest of India (82 Mg C ha^{-1}) reported by Kaul *et al.* (2010). The ecosystem level biomass carbon of different forest types ranging between 189.6 – 545.2 Mg ha^{-1} is higher than the tropical forests of Asia with values of 25 - 300 Mg C ha^{-1} (Brown *et al.* 1993; Lasco 2002; Lu *et al.* 2010).

The ecosystem level carbon stock in the tropical landscape was highest in the sal plantation forest (314.4 Mg C ha^{-1}), followed by mixed bamboo forest (282.6 Mg C ha^{-1}), old-growth broad-leaved forest (265.5 Mg C ha^{-1}), teak plantation forest (159.2 Mg C ha^{-1}) and regenerating broad-leaved forest (147.8 Mg C ha^{-1}).

The large trees (>60 cm DBH) in the old-growth broad-leaved, teak plantation, sal plantation and mixed broad-leaved forests contributed 43, 5, 32 and 52% to the total AGB in forests. In contrast, the contribution of the smaller trees to total AGB in the regenerating broad-leaved forest was significantly higher (100%) followed by teak plantation forest (95%) and sal plantation forest (68%) than the larger trees. The greater contribution of large trees to AGB in old-growth broad-leaved forest and the mixed bamboo forest was in conformity with the findings of earlier workers (Brown 1996; Brown *et al.* 1995; Brown and Lugo, 1992; Clark and Clark, 1996) who reported up to 50% contribution to AGB by the large trees (>70 cm dbh). On the other hand, Brown *et al.* (1997) reported that smaller trees contribute to most AGB in forests with <300 Mg ha^{-1} aboveground biomass. The distribution of biomass in large trees, therefore, could be an indicator of the presence or absence of past anthropogenic disturbance (Brown 1996).

A higher proportion of AGB in the higher diameter classes in old-growth broad-leaved and mixed bamboo forest does indicate the important role of large trees in carbon storage, but does not undermine the role of small trees (<60 cm dbh) which would enhance the future carbon stock because of their high carbon sequestration potential. It is well established that forest plantations sequester carbon till maturity that varies from 25 to 75 years depending upon the forest type. Beyond maturity, the trees generally have marginal carbon sequestration capability (Lal and Singh, 2000). The higher AGB in the sal plantation forest than old-growth broad-leaved forest may be attributed to the more or less uniform stand structure that results from a combination of site factors and adopted management practices (Baishya *et al.* 2009). On the other hand, wide variation in stand structure and tree growth in the natural forest resulted in lower aboveground biomass. Other factors responsible for such low total AGB are, different stages of forest growth cycle, habitat and species variabilities, and varying tree density (Terakunpisut *et al.* 2007).

Many workers have reported that in old-growth broad-leaved forests, there is a net addition to standing biomass leading to carbon storage if most trees are yet to be matured. Such scenarios are applicable to the forests where disturbance events are sporadic and concurrent. On the other hand, the matured forests do not add up any further biomass because most part of the gross primary productivity is either used up in respiration or returned to soil as litter with no net addition to the aboveground biomass density. Such old-growth broad-leaved forests thus do not significantly contribute towards carbon uptake, though they are important for regeneration and sustaining biodiversity. However, plantation forests with higher annual productivity were reported to be ideal for carbon storage and sequestration (Lal and Singh, 2000). Thus, creation of new plantation on degraded lands is a better option for carbon storage when these are planted and harvested periodically and used as a long-term source of timber.

In agreement with the findings of earlier workers, the sal and teak plantation forests, and the regenerating broad-leaved forest with less AGB in higher diameter class had the greater potential to accumulate significant quantities of biomass, and thus sequestering more atmospheric C than the old-growth broad-leaved and mixed bamboo forest with more AGB in higher diameter classes. However, given the fact that all the forest types had large tree populations under smaller dbh classes <60 cm, it can be argued that these forests have high potential for carbon sequestration.

Net primary productivity of the tropical forest ecosystems

The total aboveground NPP in the different forest types was 8.4 - 16.1 Mg ha^{-1} yr^{-1} with highest aboveground NPP in regenerating broad-leaved forest (16.1 Mg ha^{-1} yr^{-1}) and lowest in teak plantation forest (8.4 Mg ha^{-1} yr^{-1}). The aboveground NPP of the old-growth broadleaved forest and mixed bamboo forest (12.3 and 9.0 Mg ha^{-1}yr^{-1}) are in the range of aboveground NPP reported by Clark *et al.* (2001b) for Ivory Coast (9.9 - 14.3 Mg ha^{-1}yr^{-1}). The aboveground NPP in the regenerating broad-leaved forest (16.1 Mg ha^{-1} yr^{-1}) was greater than the 18-year young mixed forest (5.2 Mg ha^{-1} yr^{-1}) in Japan (Ohtsuka *et al.* 2010). The values were however more than the reported values for India (3.3 Mg ha^{-1}yr^{-1}) and (6.4 - 7.6 Mg ha^{-1}yr^{-1}) for Brazil by Clark *et al.* (2001b). The total ecosystem NPP in the teak plantation (9.6 Mg ha^{-1}yr^{-1}) was within the reported range (7.2 - 9.2 Mg ha^{-1}yr^{-1}) of tropical dry deciduous teak forest of Madhya Pradesh, India by Pande (2005). The aboveground NPP in the sal plantation forest with (12.7 Mg ha^{-1}yr^{-1}) was greater than the range (2.1 – 9.2 Mg ha^{-1}yr^{-1}) reported by Gautam *et al.* (2011) for the tropical sal forests of Doon Valley Himalayas, India. The average NPP for the tropical forest types (9.6 – 18.4 Mg ha^{-1} yr^{-1}) is significantly greater than the value reported by Luyssaert *et al.* (2007) (8.64 Mg ha^{-1} yr^{-1}) for 29 tropical humid evergreen forests of the world.

The total NPP of the forest was high, which may be attributed to relatively higher soil NPK content and moisture regime in the various forest types. Gower *et al.* (1994) reported that N fertilization increases NPP of forest, and carbon allocation to belowground components decreases with increase in soil N availability (Haynes and Gower, 1995). The extremely low BGB of 30.3, 34.4, 45.3, 50.8 and 73.2 Mg ha^{-1} in the regenerating broad-leaved, teak plantation, mixed bamboo, old-growth broad-leaved and sal plantation forests as obtained in this study may be attributed to this reason. Vogt *et al.* (1996) reported that root NPP does not depend only on nutrient availability in many species. Relatively low root NPP as observed in this study for various forest types (1 – 2.3 Mg ha^{-1} yr^{-1}) in comparison to the other works (Karizumi 1974) supports this argument.

The clay content in the soil appears to be more important than soil fertility in determining the proportion of NPP allocated below-ground. The low belowground NPP in the forest types may be due to high clay content. It has been reported that forests with higher clay content tend to allocate a lower proportion of their NPP to below-ground (Aragao *et al.* 2009).

The net ecosystem NPP of the forest was high in the regenerating broad-leaved forest (18.4 Mg ha^{-1} yr^{-1}) and low in the teak plantation (9.6 Mg ha^{-1} yr^{-1}). The sal plantation forest had 14.4 Mg ha^{-1} yr^{-1} greater than old-growth broadleaved forest (13.6 Mg ha^{-1} yr^{-1}) and mixed bamboo forest (10.1 Mg ha^{-1} yr^{-1}). The net ecosystem NPP values of different forest types are comparable with the findings of Clark *et al.* (2001b) with 3.1 to 21.7 Mg ha^{-1} yr^{-1} for the tropical forests of the world. The ecosystem level carbon sequestration values in the old-growth broad-leaved, regenerating broad-leaved, teak plantation, sal plantation and mixed bamboo forests in the tropical landscape had 265.5, 147.7, 159.2, 314.4 and 282.6 Mg C ha^{-1}, respectively.

Tropical forests tend to carry their biomass in the standing crop relatively more than temperate forests. The carbon sequestration potential of the tropical forests are more effective in carbon sequestration than temperate forest due to net productivity differences (Brown *et al.* 1989).

The present study concludes that the carbon sequestration potential of old-growth broad-leaved forest and the mixed bamboo forests of northeast India are high, even with a past disturbance history. The carbon values are comparable to most other tropical forests of the region. Considering the DBH-AGB distribution pattern, the sal plantation forest had greater carbon stock as well as higher sequestration potential than other tropical forest types because of ongoing scientific management practices, uniform age and stand structure (Baishya *et al.* 2009).

CHAPTER VI

CARBON SEQUESTRATION IN MONTANE SUB-TROPICAL LANDSCAPE

INTRODUCTION

The amount of C stored in plant biomass globally exceeds that of atmospheric CO_2, and nearly 90% of the plant biomass C is stored in tree biomass (Korner 2006). This emphasizes the importance of forest ecosystems in the global carbon cycle and the necessity to accurately evaluate the amount of C stored in forest ecosystems (Korner 2006). The carbon pool of a forest ecosystem varies with age (Clark *et al.* 2004; Kurz and Apps, 1995). While young and middle-aged forest stands act as active carbon sinks (Valentini *et al.* 2000), old stands are moderate to small C sinks or even C sources depending on the forest type and species composition (Desai *et al.* 2005; Knohl *et al.* 2003; Law *et al.* 2003; Malhi *et al.* 1999). The estimation of carbon in a forest landscape includes C in soil, microbial biomass, litter, aboveground and belowground biomass compartments. In view of the global data gap on the carbon pools and flux size in montane sub-tropical landscape, the present study was undertaken.

Soils of the world are potentially viable sinks for atmospheric carbon (C) and may significantly contribute to mitigation of global climate change (Lal *et al.* 1998). The soil organic carbon, which is 30% of the total global carbon, is stored in sub-tropical and tropical ecosystems. However, it is being rapidly lost due to deforestation. Tree plantations have been advocated as C sink. However, little is known about the rates of C turnover and sequestration into soil organic matter under sub-tropical and tropical tree plantations (Richards *et al.* 2007). The assessment of potential C sequestration in soil requires the estimation of C pools under existing land uses, and the distribution of C in the soil profile.

Removal of trees from the forest displaces a large amount of sequestered carbon (IPCC 2000) and consequently reduces the SOC held in soil profiles (Glaser *et al.* 2000). The impact of deforestation on SOC decrease is more pronounced in the upper soil layer (Sombroek *et al.* 1993) than the deeper layer. Gradual conversion of forest and grassland to cropland has resulted in significant losses of soil carbon worldwide (Lal 2002). The soil carbon pools in different forest types in montane sub-tropical landscape have not been studied.

The soil microbial biomass forms a transformation matrix of organic matter and the associated mineralization of important nutrients that regulate plant productivity (Cleveland *et al.* 2004). Soil microbial biomass constitutes a significant part of the potentially mineralizable N and serves both as the transformation agent and source-sink of N (Bonde *et al.* 1998). It plays an important role in N cycling due to rapid turnover rate. Plant community composition influences soil microbial communities in various ways. The microbial biomass is the driving force for organic matter transformation and nutrient cycling in soil systems (Grayston *et al.* 2001; Zeller *et al.* 2001). Different soil microbes associated with different tree species often have variable amounts of microbial biomass (Bauhus *et al.* 1998; Templer *et al.* 2003). Soil microbial biomass is also related to climate (Dyer *et al.* 1990), soil moisture (Taylor *et al.* 1999), soil texture (Bauhus *et al.* 1998; Hassink 1994), plant productivity (Zak *et al.* 1990, 1994) and soil organic matter quality (Garcia-Gill *et al.* 2000; Taylor *et al.* 1989; Zak *et al.* 1990). Nutrients derived from accumulated litter on the soil surface during dry season are immobilized by microorganisms as water availability during rainy season allows microbial growth (Mao *et al.* 1992). Apart from the soil moisture, detrital material on the forest floor and relatively high temperature are important factors that favour growth of microbial population in soil during rainy season. Contrary to these conditions, soil moisture and temperature decline during winter and conditions become unfavourable for microbial growth, though some amount of partially decomposed detrital material is still available on the forest floor. This indicates the role of soil moisture and ambient temperature in seasonal variation of MBC on the forest floor.

Litter fluxes and the associated nutrient release through litter decay play a fundamental role in maintaining the sustainability of natural forests (Arunachalam *et al.* 1998b; Morrison 1991). Litter decomposition is controlled by three factors namely climate, litter quality and the nature and abundance of the decomposing organisms (Bottner *et al.* 1998). Within an ecosystem, plant litter quality is the most important factor in determining the rate of decomposition (Aerts 1997; Cadisch and Giller, 1996). Therefore, the litter characteristics of the dominant plant species in an ecosystem strongly influence decomposition processes (Hoorens *et al.* 2003). During decomposition, some of the C and N are assimilated into microbial tissue and a part is converted into resistant humic substances through microbial action. The decomposition processes start both through leaching and through maintenance of optimal residue moisture content for microbial catabolism (Vanlauwe *et al.* 1995). Therefore, the size of populations of decomposing fungi, bacteria and invertebrates (Dighton 1997), soil temperature and soil moisture content (Kochy and Wilson, 1997) and the quality of litter in terms of its susceptibility to the attack of by decomposers influence decomposition rate. Swift *et al.* (1979) concluded that leaching was the main factor influencing the initial weight loss of leaf litter. Litter production is of great significance in ecosystem function and a few studies on this aspect have been conducted in north-eastern India as well (Arunachalam *et al.* 1998a; Kamei 2007; Singh 1980).

Soil respiration, a key component of biogeochemical cycle, is the second largest carbon flux in terrestrial ecosystems and plays a critical role in global carbon cycling (Law *et al.* 1999). Thus, a great number of studies on soil respiration have been conducted (Davidson *et al.* 2006; Raich and Schlesinger, 1992; Schlesinger 1997). Because different vegetation modifies microclimate and substrate availability differentially (Hibbard *et al.* 2005). In spite of the importance of montane sub-tropical forest in the carbon budget and climatic system, soil respiration in these forests is poorly understood and little information on this aspect is available.

Net primary productivity (NPP) represents the net carbon input from the atmosphere to terrestrial vegetation (Melillo *et al.* 1993). It is the net organic matter produced by live plants at the end of a specific time interval (Clark *et al.* 2001). It is an important index for estimating carbon budget and evaluating the patterns, processes and dynamics of carbon cycling in forest ecosystems at local, regional and global scales (Luo *et al.* 2002). Most NPP studies emphasize on the aboveground and only a few on the belowground compartment, but very little is known about the ecosystem level NPP in different forest types of the world. The data for all the vegetation components other than tree are also not available. Therefore, works on total ecosystem level NPP estimation are extremely limited.

In order to bridge the carbon data gap in the montane sub-tropical landscape, the present study was undertaken to analyze the seasonal and spatial distribution pattern of soil organic carbon, microbial biomass carbon, litterfall, aboveground biomass, belowground biomass, ecosystem level biomass and carbon, and ecosystem level NPP in different montane sub-tropical forest types. The pools and fluxes of C in four dominant montane sub-tropical forest types of Meghalaya *viz.*, old-growth broad-leaved, old-growth pine, regenerating pine and regenerating broad-leaved forests were studied. Some selected soil physical and chemical parameters were also studied to assess their impact on carbon pools and fluxes and vice-versa.

RESULTS

The montane sub-tropical landscape had four forest types. These are (1) old-growth broad-leaved forest, (2) old-growth pine forest, (3) regenerating pine forest, and (4) regenerating broad-leaved forest.

Tree species composition and community characteristics

Old-growth broadleaved forest

Total number of tree species was 21 and *Lindera latifolia* was the dominant species (IVI 60.5) and *Lithocarpus dealbatus* (IVI 55.1) was the co-dominant species. The basal area of trees in this forest was 55.44 m^2 ha^{-1}. *Lindera latifolia* had the highest basal area of 15.1 m^2 ha^{-1} and highest density of 272 individuals ha^{-1}. The total density of the forest was 980 individuals ha^{-1}. Based on A/F ratio, 76.2% of total tree species exhibited clumped distribution, 14.3% exhibited random distribution and 9.5% exhibited uniform distribution pattern (Table 6.1).

Table 6.1 Tree species composition, density, frequency, basal area, IVI and A/F ratio in montane sub-tropical old-growth broad-leaved forest of Upper Shillong

Sl.No.	Species	Density	Frequency	Basal Area $m^2\ ha^{-1}$	IVI	A/F
1	*Albizia mollis* Boiv	10	8	0.44	3.81	0.16
2	*Alnus nepalensis* D Don	26	16	0.11	6.83	0.10
3	*Betula alnoides* Buch -Ham	16	12	0.13	4.86	0.11
4	*Castanopsis purpurella* (Miq) N P Balak	98	70	11.43	48.03	0.02
5	*Coffea khasiana* Hook f	8	8	0.38	3.49	0.13
6	*Daphniphyllum himalayense* (Benth) Mull Arg	30	26	0.08	9.66	0.04
7	*Elaeocarpus braceanus* Watt Ex C B Clarke	34	12	4.81	15.12	0.24
8	*Engelhardtia spicata* Blume	6	6	0.12	2.33	0.17
9	*Euonymus lawsonii* C B Clarke ex Prain	16	14	0.17	5.43	0.08
10	*Leucosceptrum canum* Sm	44	34	0.31	13.51	0.04
11	*Lindera latifolia* Hook f	272	22	15.11	60.47	0.56
12	*Lithocarpus dealbata* Rehedr	214	56	10.70	55.07	0.07
13	*Litsea elongata* Benth & Hook f	8	4	0.06	1.91	0.50
14	*Litsea khasyana* Meissn	4	6	0.03	1.95	0.11
15	*Lyonia ovalifolia* Hort	12	2	0.33	2.32	3.00
16	*Myrica esculenta* Buch -Ham ex D Don	64	4	5.86	18.09	4.00
17	*Persea kingii* (Hook f) Kosterm	56	50	2.54	22.73	0.02
18	*Persea odoratissima* (Nees) Kosterm	28	24	1.11	10.83	0.05
19	*Quercus griffithii* Hook f & Thomson ex Miq	12	10	1.44	6.31	0.12
20	*Rhododendron arboreum* Sm	14	10	0.27	4.40	0.14
21	*Schefflera hypoleuca* Harms	8	8	0.03	2.85	0.13
	Total	980	402	55.44	300.00	

Old-growth pine forest

In the old-growth pine forest, the total number of tree species was 9 and being a pine forest, *Pinus kesiya* was the dominant species with high IVI of 177.4. *Lyonia ovalifolia* was the second important tree species (broad-leaved) in the forest with IVI of 30.8. The basal area of trees in this forest was 126 $m^2\ ha^{-1}$. *Pinus kesiya* had the highest basal area of 97.4 $m^2\ ha^{-1}$ and tree density of 454 individuals ha^{-1}. The total tree density of the forest was 628 individuals ha^{-1}. Based on A/F ratio, 55.5 % of total tree species exhibited clumped distribution, 11.1% exhibited random distribution and 33.3% exhibited uniform distribution pattern (Table 6.2).

Table 6.2 Tree species composition, density, frequency, basal area, IVI and A/F ratio in montane sub-tropical old-growth pine (*Pinus kesiya*) forest of Upper Shillong

Sl. No.	Species	Density	Frequency	Basal Area m^2 ha^{-1}	IVI	A/F
1	*Alnus nepalensis* D.Don	8	30	0.36	9.89	0.02
2	*Lindera latifolia* Hook f.	36	74	2.05	27.91	0.01
3	*Lyonia ovalifolia* (Wall.)	36	90	0.10	30.81	0.01
4	*Magnolia hodgsonii* (Hook.f. Thomson) H.Keng	2	4	0.03	1.46	0.25
5	*Myrica esculanta* Buch.-Ham. D.Don	23	30	1.07	12.84	0.05
6	*Pinus kesiya* Royle ex Gordon	454	100	97.40	177.36	0.09
7	*Rhododendron arboreum* Sm Sm	5	16	0 15	5 36	0.04
8	*Rhus javanica* L.	8	16	2.03	7.33	0.06
9	*Schima khasiana* Dyer	56	10	22.84	29.82	1.12
	Total	628	360	126.03	300.00	

Regenerating pine forest

Total number of tree species in the regenerating pine forest was 6 and being a pine forest, *Pinus kesiya* was the dominant species (IVI 213.1) and *Lyonia ovalifolia* was the next important species in the forest (IVI 46.4). Total tree basal area of this forest was 17.9 m^2 ha^{-1}. *Pinus kesiya* had the highest basal area of 16 m^2 ha^{-1} and high tree density of 1460 individuals ha^{-1}. The total density of the forest was 1768 individuals ha^{-1}. Based on A/F ratio, 83.3% of total tree species exhibited clumped distribution and 16.7% exhibited random distribution pattern (Table 6.3).

Table 6.3 Tree species composition, density, frequency, basal area, IVI and A/F ratio in montane sub-tropical regenerating pine (*Pinus kesiya*) forest of Upper Shillong

Sl.No.	Species	Density	Frequency	Basal Area ha^{-1}	IVI	A/F
1	*Alnus nepalensis* D.Don	84	30	0.27	18.66	0.09
2	*Lyonia ovalifolia* Hort.	180	74	1.01	46.40	0.03
3	*Myrica esculanta* Buch.-Ham. ex D.Dc	8	6	0.42	5.28	0.22
4	*Pinus kesiya* Royle ex Gordon	1460	100	15.97	213.07	0.15
5	*Rhododendron arboreum* Sm.	16	16	0.17	8.47	0.06
6	*Rhus javanica* L.	20	16	0.07	8.13	0.08
	Total	1768	242	17.91	300.00	

Regenerating broad-leaved forest

In the regenerating broad-leaved forest, the total number of tree species was 35, highest among all the forest types. *Rhododendron arboreum* was the dominant species (IVI 37.6) and *Lithocarpus dealbatus* (IVI 32.25) was the co-dominant species. The total tree basal area of the forest was 10.5 m^2 ha^{-1}. *Rhododendron arboreum* had the highest basal area of 1.8 m^2 ha^{-1}. *Lithocarpus dealbatus* had the highest density of 264 individuals ha^{-1}. The total density of the forest was 2106 individuals ha^{-1}. Based on A/F ratio, 48.6 % of total tree species exhibited clumped distribution, 45.7% exhibited random distribution and 5.7% exhibited uniform distribution pattern (Table 6.4).

Table 6.4 Tree species composition, density, frequency, basal area, IVI and A/F ratio in montane sub-tropical regenerating broad-leaved forest of Upper Shillong

Sl.No.	Species	Density	Frequency	Basal Area m^2 ha^1	IVI	A/F
1	*Acer laevigatum* Wall	36	36	0 28	6 87	0 03
2	*Albizia mollis* Boiv	44	32	0 33	7 57	0 06
3	*Alnus nepalensis* D Don	20	10	0 09	2 32	0 20
4	*Betula alnoides* Buch -Ham	38	24	0 14	4 44	0 07
5	*Cinnamomum bejolghota* Sweet	76	58	0 7	16 31	0 05
6	*Cinnamomum tamala* Nees	52	34	0 2	6 24	0 04
7	*Cinnamomum zeylanicum* Blume	10	6	0 02	0 99	0 28
8	*Daphniphyllum himalayense* (Benth) Mull Arg	92	64	0 19	9 79	0 02
9	*Eleagnus pyriformis* Hook f	92	78	0 4	16 25	0 03
10	*Eurya acuminata* DC	98	42	0 37	10 05	0 06
11	*Ficus nemoralis* Wall	22	24	0 17	4 71	0 07
12	*Ficus neriifolia* J E Sm	14	8	0 07	1 75	0 22
13	*Leucosceptrum canum* Sm	18	10	0 08	2 08	0 18
14	*Ligustrum robustrum* Bedd	108	46	0 24	9 38	0 05
15	*Lithocarpus dealbatus* Rehedr	264	90	1 13	32 25	0 05
16	*Lithocarpus elegans* (Blume) Hatusima ex Soepadmo	60	40	0 18	6 76	0 04
17	*Lithocarpus fenestrata* Rehder	26	16	0 08	2 85	0 10
18	*Litsea khasyana* Meissn	84	64	0 37	11 2	0 02
19	*Lyonia ovalifolia* Hort	90	76	0 79	19 01	0 03
20	*Myrica esculenta* Buch -Ham ex D Don	76	64	0 75	17 16	0 04
21	*Neolitsea cassia* (L) Kosterm	32	16	0 15	3 75	0 13
22	*Persea kingii* (Hook f) Kosterm	56	34	0 23	6 63	0 05
23	*Photinia notoniana* Wall	20	18	0 16	4 03	0 10
24	*Rhododendron arboreum* Sm	236	96	1 82	37 57	0 04
25	*Rhus acuminata* DC	72	48	0 17	7 7	0 03
26	*Rhus javanica* L	124	68	0 46	18 1	0 05
27	*Schefflera hypoleuca* Harms	16	12	0 08	2 24	0 11
28	*Schima khasiana* Dyer	24	16	0 12	3 13	0 09
29	*Symplocus crateogoides* Buch ex D Don	44	32	0 02	4 25	0 05
30	*Symplocus javanica* Kurz	52	36	0 25	6 9	0 04
31	*Symplocus spicata* Roxb	10	10	0 06	1 66	0 10
32	*Toddalia asiatica* Baill	10	10	0 05	1 56	0 10
33	*Viburnum simonsii* Hook f & Thomso	42	52	0 07	7 66	0 03
34	*Wendlandia wallichi* Wight & Arn	26	18	0 13	3 47	0 08
35	*Zanthoxylum acanthopodium* DC	22	18	0 13	3 35	0 07
	Total	2106	1306	10 47	300 00	

Among all the forest types, the regenerating broad-leaved forest had the greatest tree density (2996 individuals ha^{-1}) and species richness (35 species). The Shannon diversity, Simpson dominance and Species evenness indices were high in the regenerating broad-leaved forest and followed the order: SRBF>SOBF>SOPF>SRPF. Among the montane sub-tropical landscape elements, α diversity was highest in the regenerating broad-leaved forest and followed the order: SRBF>SOBF>SOPF>SRPF (Table 6.5).

Table 6.5 Community indices for different montane sub-tropical forest types of Upper Shillong (SOBF= sub-tropical old-growth broad-leaved forest; SOPF= sub-tropical old-growth pine forest; SRPF= sub-tropical regenerating pine forest; SRBF= sub-tropical regenerating pine forest)

Parameters	SOBF	SOPF	SRPF	SRBF
Species richness	21	9	6	35
Dominance	0.2	0.5	0.7	0.1
Shannon -Weiner diversity index	2.3	1.0	0.7	3.2
Simpson dominance index	0.9	0.5	0.3	1.0
Evenness	0.5	0.4	0.3	0.7
α diversity	3.8	0.9	0.8	6.0

PHYSICO-CHEMICAL PROPERTIES OF SOIL

Soil texture

The soil texture at both surface (0-10 cm) and sub-surface (10-20 cm) soil layers was sandy in all the forest types (Table 6.6).

Table 6.6 Proportion of soil particles and soil texture class in different montane sub-tropical forest types of Upper Shillong

Forest types	Depth (cm)	Proportion of soil particle			Soil textural class
		Clay (%)	Silt (%)	Total Sand (%)	
SOBF	0-10	9.8	0.8	89.5	Sandy
	10-20	10.9	0.9	88.2	Sandy
SOPF	0-10	10.3	1.5	88.2	Sandy
	10-20	11.4	2.1	86.6	Sandy
SRPF	0-10	8.4	0.4	91.3	Sandy
	10-20	9.3	2.1	88.6	Sandy
SRBF	0-10	7.2	0.4	92.5	Sandy
	10-20	8.2	2.1	89.7	Sandy

Water holding capacity

The water holding capacity was high in the old-growth broad-leaved forest and old-growth pine forest and low in the regenerating pine forest (Fig. 6.1). Water holding capacity decreased significantly ($p<0.01$) with depth in all the forest types. Two-way ANOVA showed significant ($P< 0.01$) variation in the water holding capacity of the soil due to forest type (F = 170.3) and soil depth (F = 304.3).

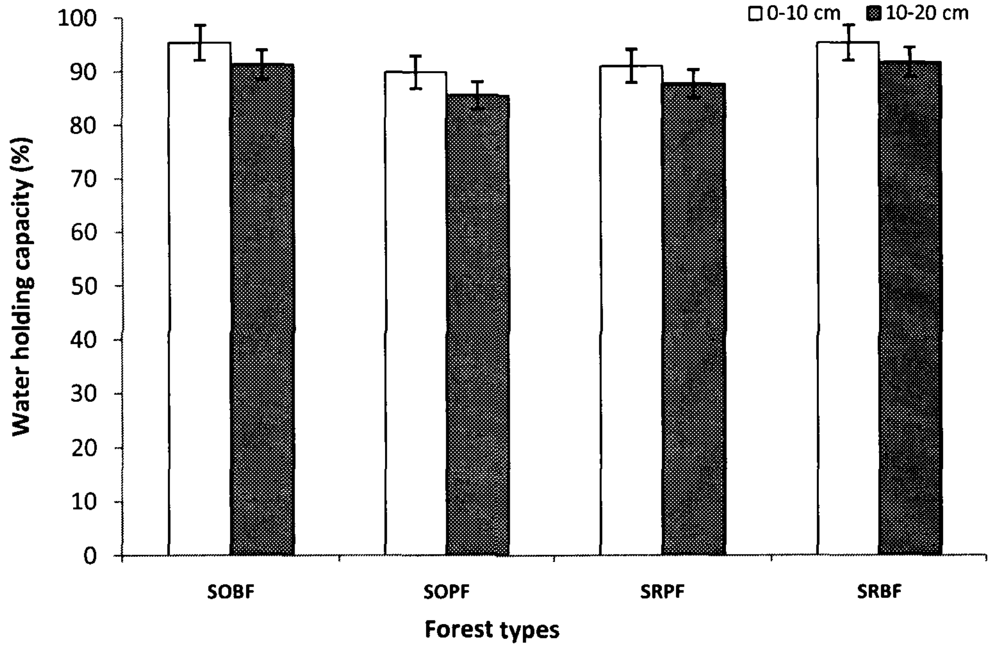


Fig. 6.1 Water holding capacity (%) in surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical landscape of Upper Shillong. The values are mean (±SE) of five replicate samples

Bulk density and Porosity

Bulk density was highest in the old-growth broad-leaved forest (1.5 g cm^{-3}) and lowest in the old-growth pine forest (1.2 g cm^{-3}) (Table 6.7). The bulk density increased with depth in all the forest types. Two-way ANOVA showed significant ($P< 0.01$) variation in the bulk density of the soil due to forest type (F = 307.9) and soil depth (F = 19.1). Conversely, soil porosity was highest in the regenerating pine forest (53.6%) and lowest in the old-growth broad-leaved forest (44.1%) (Table 6.7). Soil porosity declined with increase in depth in all the forest types. Two-way ANOVA showed a significant (*P*< 0.01) variation in the porosity of the soil due to forest type (F = 314.1).

Table 6.7 Bulk density (g cm^{-3}) and porosity (%) in surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical landscape of Upper Shillong. The values are mean (±SE) of five replicate samples

Forest types	Depth (cm)	Bulk density (g m^3)	Porosity (%)
SOBF	0-10	1.5	44.2
	10-20	1.6	43.1
SOPF	0-10	1.2	47.2
	10-20	1.3	46.8
SRPF	0-10	1.3	53.6
	10-20	1.3	52.1
SRBF	0-10	1.4	51.7
	10-20	1.4	50.6

Soil moisture content (SMC)

The SMC was highest during summer season and lowest during winter season (Fig. 6.2). When compared among the forest types, both the pine forests had lower soil moisture content than the broad-leaved forests. Three-way ANOVA revealed significant (*P*< 0.01) variation in the SMC due to forest type and season (Table 6.8).

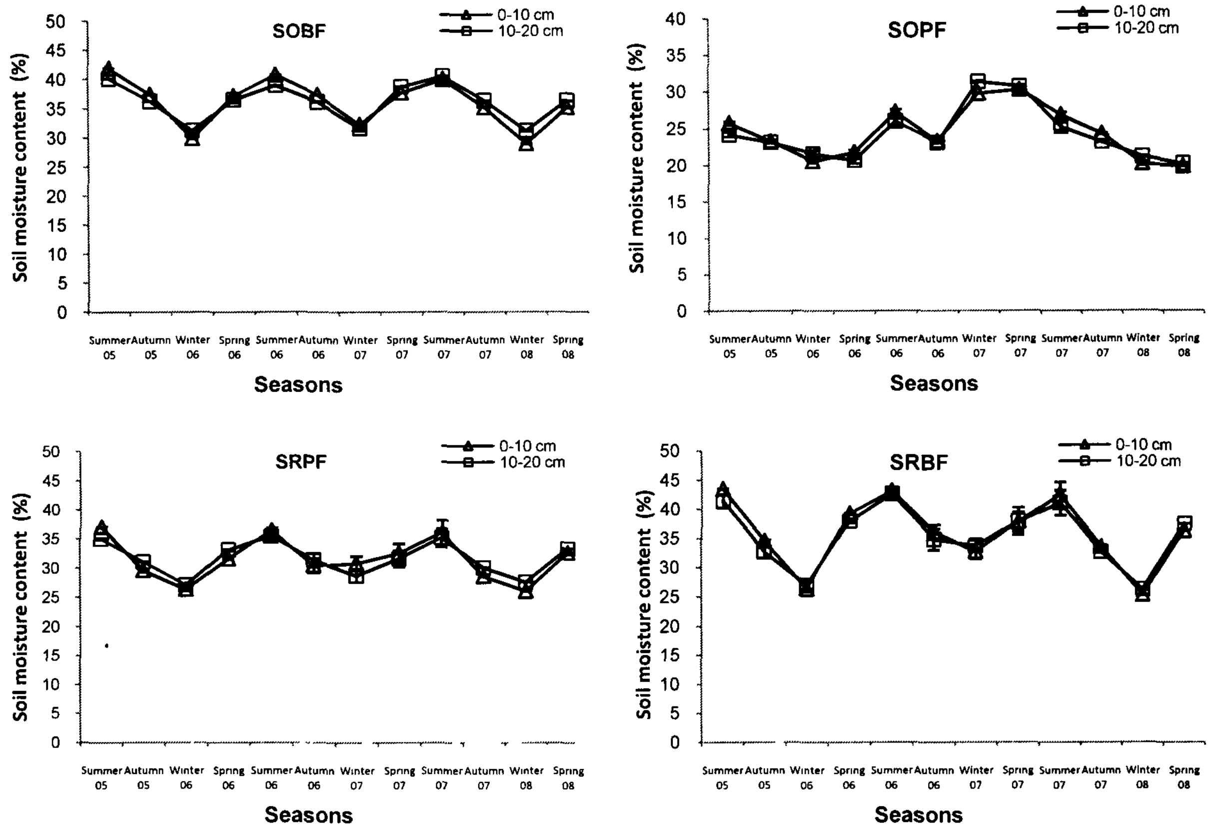


Fig. 6.2 Seasonal variation in soil moisture content (%) in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.8 Three-way ANOVA showing effect of forest type, season and soil depth on soil moisture content (%) in different forest types in montane sub-tropical landscape of Upper Shillong (*$p<0.01$, **$p<0.01$)

Variation due to	df	SS	MS	F	*p*
Forest type	3	2154.96	718.32	1405.40**	0.000
Depth	1	0.34	0.34	0.67	0.419
Season	11	1064.89	96.81	189.41**	0.000
Forest type*Depth	3	1.21	0.40	0.79	0.507
Forest type*Season	33	426.21	12.92	25.27**	0.000
Depth*Season	11	18.85	1.71	3.35*	0.003
Forest type*Depth*Season	33	16.87	0.51	1.00	0.500

Soil pH

The soil pH was acidic ranging from pH value of 4.2 to 5.9. The regenerating broad-leaved forest had the lowest pH value than the other forest types (Fig.6.3). The difference in pH between surface (0-10 cm) and sub-surface (10-20 cm) soil layers was significant ($p<0.01$) and the soil was more acidic in the surface layer than the sub-surface layer. The soil was more acidic in summer season than the other seasons ($p<0.01$).

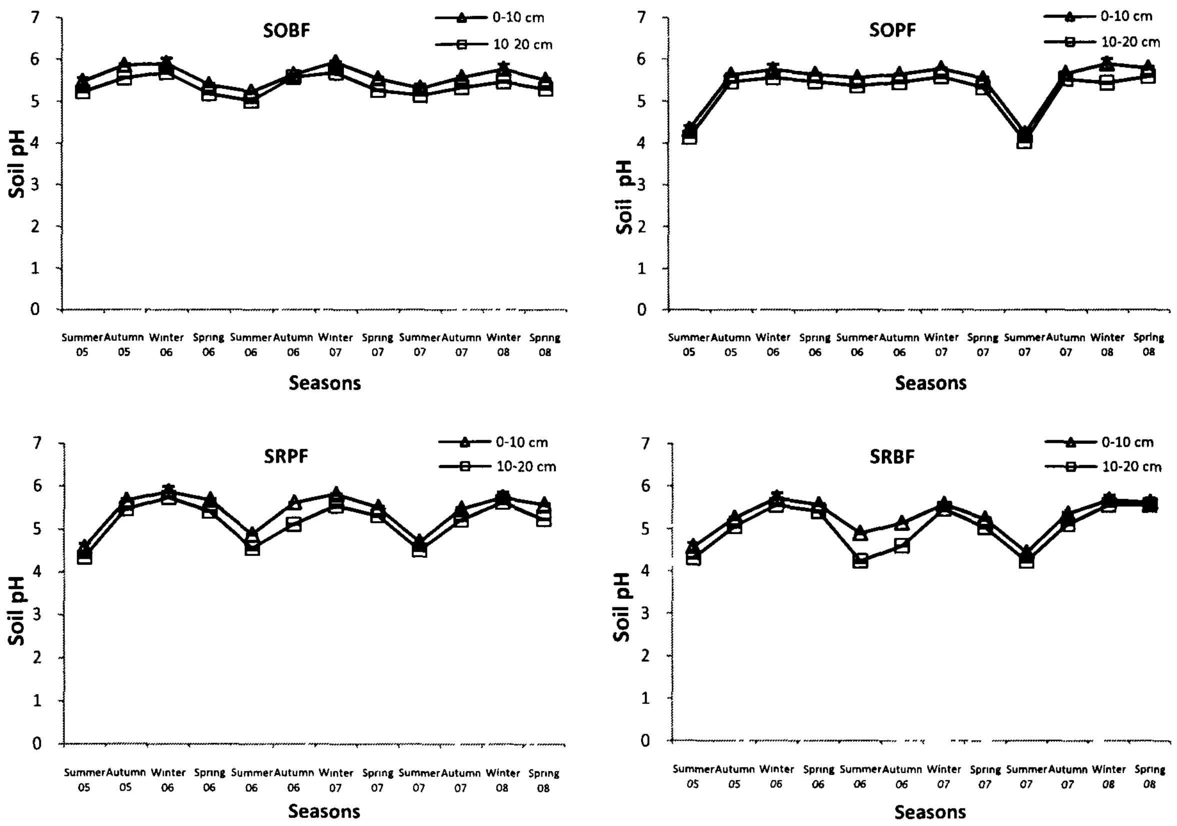


Fig. 6.3 Seasonal variation in soil pH of the different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Total Kjeldahl Nitrogen (TKN)

The highest TKN value was recorded during winter season and lowest during summer season in all the forest types. The highest TKN value was recorded for the old-growth pine forest (0.70%) and it was lowest in the regenerating broad-leaved forest (0.20%) (Fig.6.4). TKN declined significantly (*p*<0.01) with the increase in the soil depth. Three-way ANOVA showed significant variation in TKN due to forest type, season and depth (p<0.01) (Table 6.9).

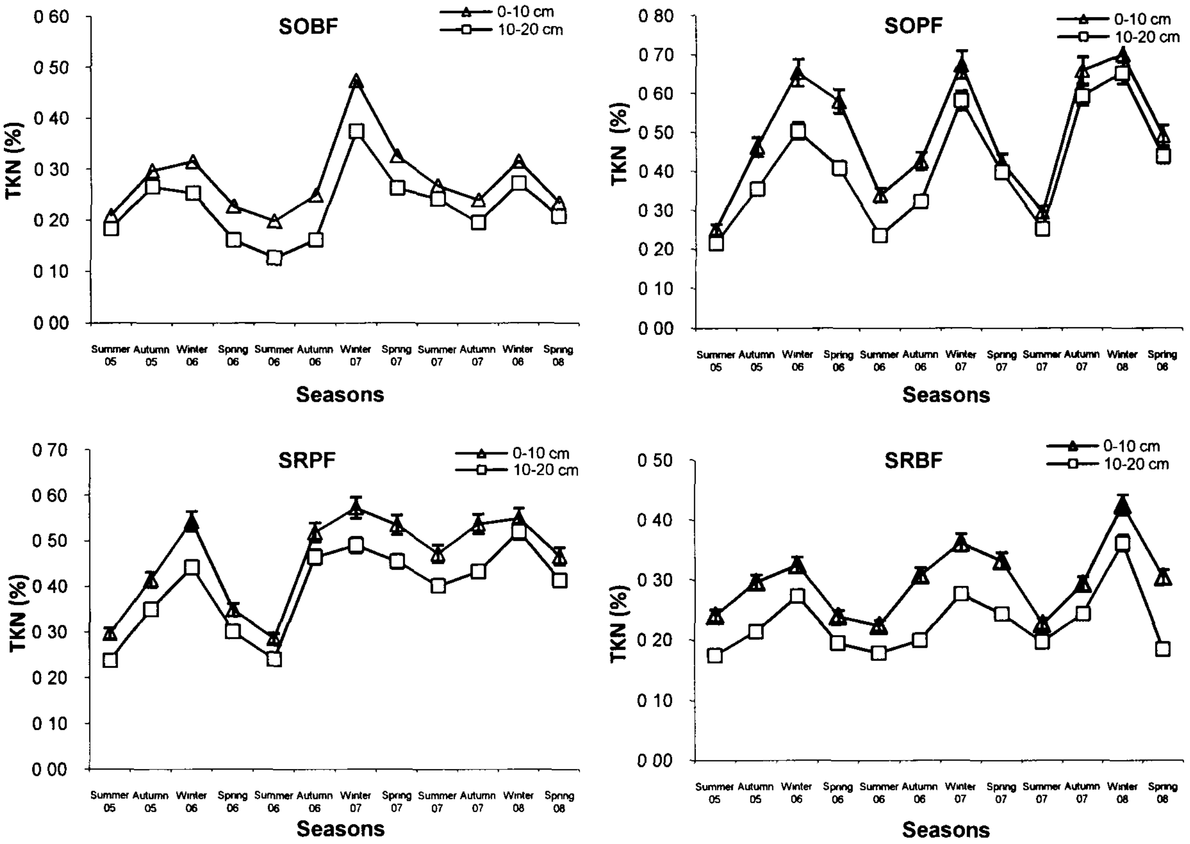


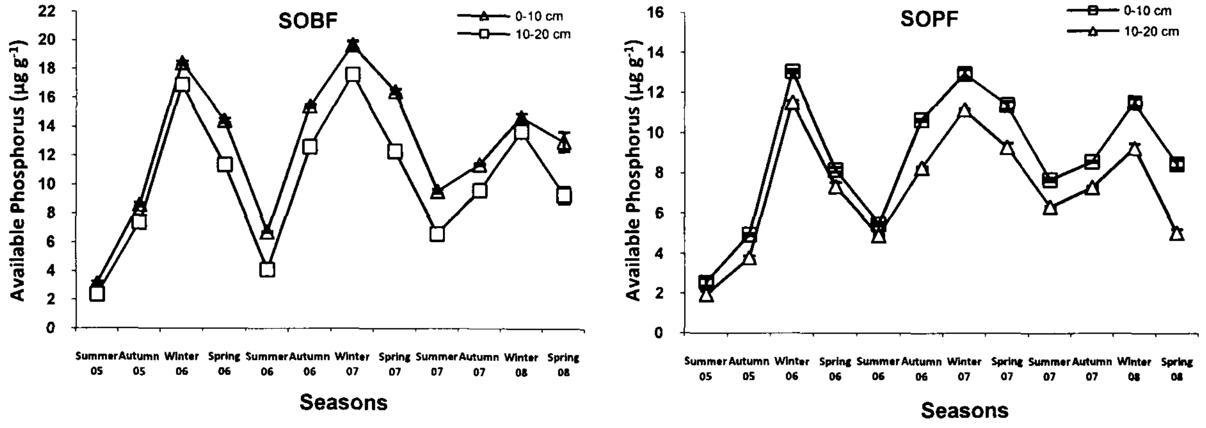
Fig. 6.4 Seasonal variation of total kjeldahl nitrogen (TKN, %, ±SE) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.9 Three-way ANOVA showing the effect of forest type, season and depth on total kjeldahl nitrogen in different forest types in montane sub-tropical landscape of Upper Shillong ($p<0.01$)**

Variation due to	df	SS	MS	F	*p*
Forest type	3	0.82	0.27	579.34**	0.000
Depth	1	0.11	0.11	235.61**	0.000
Season	11	0.59	0.05	113.69**	0.000
Forest type*Depth	3	0.00	0.00	1.80	0.167
Forest type*Season	33	0.28	0.01	18.21**	0.000
Depth*Season	11	0.01	0.00	1.26	0.290
Forest type*Depth*Season	33	0.02	0.00	1.00	0.500

Available Phosphorus (P)

The highest available Phosphorus value was obtained in the regenerating broad-leaved forest (45.5 μg g^{-1}) and lowest in the old-growth pine forest (2.6 μg g^{-1}). The values were greatest during winter and lowest in summer season (Fig. 6.5). The available Phosphorus declined significantly ($p<0.01$) with the increase in the soil depth. Three-way ANOVA showed significant variation ($p<0.01$) due to forest type, depth and season (Table 6.10).



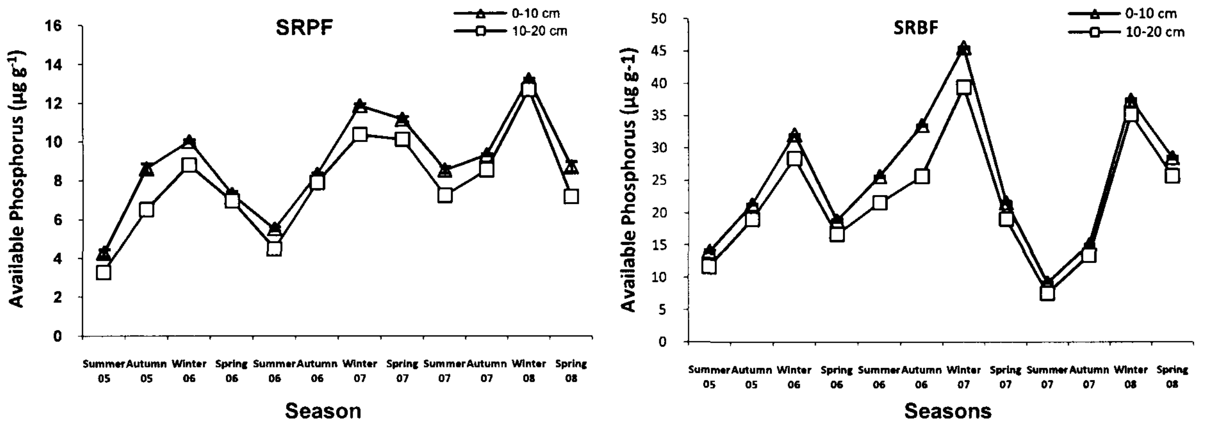


Fig. 6.5 Seasonal variation in available phosphorus (µg g^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.10 Three-way ANOVA showing the effect of forest type, season and depth on available Phosphorus in different forest types in montane sub-tropical landscape of Upper Shillong (p<0.01)**

Variation due to	df	SS	MS	F	*p*
Forest type	3	3851.71	1283.90	1970.03**	0.000
Depth	1	102.60	102.60	157.43**	0.000
Season	11	1860.62	169.15	259.54**	0.000
Forest type*Depth	3	16.72	5.57	8.55**	0.000
Forest type*Season	33	1190.67	36.08	55.36**	0.000
Depth*Season	11	10.38	0.94	1.45	0.199
Forest type*Depth*Season	33	21.51	0.65	1.00	0.500

Exchangeable Potassium (K)

The exchangeable K value was highest in the old-growth broad-leaved forest (239.5 µg g^{-1}) and lowest in the old-growth pine forest (74 µg g^{-1}). During winter season, exchangeable K value was the highest and during summer it was the lowest in all the three years of study (Fig. 6.6). The exchangeable K declined significantly (p<0.01) with increase in the soil depth. Three-way ANOVA showed significant variation due to forest type, depth and season (p<0.01) (Table 6.11).

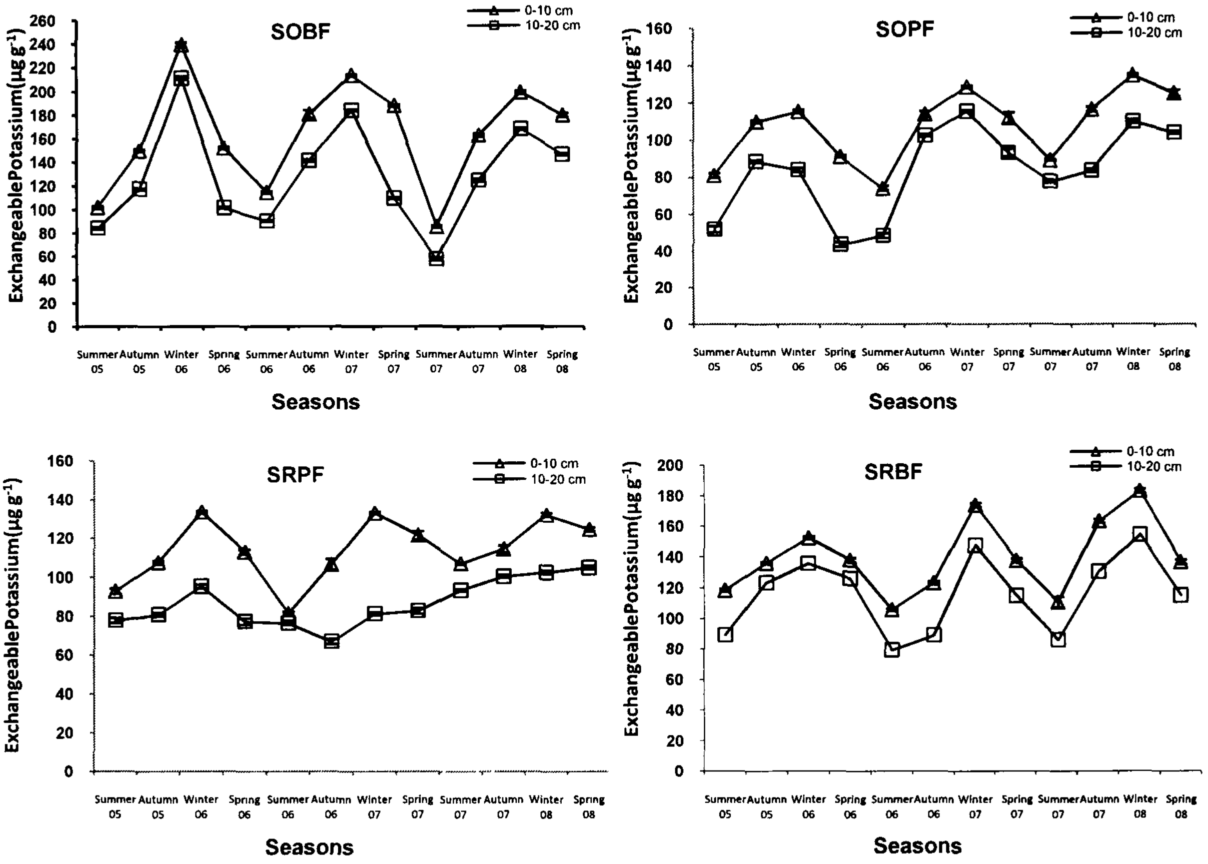


Fig. 6.6 Seasonal variation in exchangeable potassium (µg g^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different montane sub-tropical forest types of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.11 Three-way ANOVA showing the effect of forest type, season and depth on exchangeable Potassium in different forest types in montane sub-tropical landscape of Upper Shillong (*p<0.01, **p<0.01)

Variation due to	df	SS	MS	F	p
Forest type	3	33116.80	11038.93	66.31**	0.000
Depth	1	30171.14	30171.14	181.24**	0.000
Season	11	42050.49	3822.77	22.96**	0.000
Forest type*Depth	3	2373.36	791.12	4.75*	0.007
Forest type*Season	33	25094.47	760.44	4.57*	0.000
Depth*Season	11	1986.52	180.59	1.08	0.402
Forest type*Depth*Season	33	5493.48	166.47	1.00	0.500

SOIL ORGANIC CARBON (SOC) POOL

The SOC pool in the surface and sub-surface soil layer was largest in the old-growth broad-leaved forest (27.5 & 25.9 Mg ha^{-1}, respectively) and lowest in the regenerating pine forest (17.3 & 15.3 Mg ha^{-1}, respectively). The values were greatest in summer season and lowest during winter season in all the forest types and at both the soil depths (Fig. 6.7). The mean seasonal SOC in the surface and sub-surface layers were highest in the old-growth broad-leaved forest (25.1 & 23.7 Mg ha^{-1}, respectively) and lowest in the regenerating pine forest (20.7 & 17.5 Mg ha^{-1}, respectively). The SOC pool declined significantly ($p<0.01$) with increase in the soil depth. Three-way ANOVA revealed significant variation due to forest type, season and depth ($p<0.01$) (Table 6.12).

The soil organic carbon pool upto a depth of 1 m varied significantly ($p<0.01$) among the four seasons in all the forest types. It was highest during summer and lowest during winter season (Fig. 6.8 – 6.11). SOC declined with the depth ($p<0.01$). The mean seasonal value of total SOC pool was highest in regenerating broad-leaved forest (102.4 Mg ha^{-1}) while the lowest SOC value was in the regenerating pine forest (35.2 Mg ha^{-1}) (Table 6.13). Three-way ANOVA revealed significant variation due to forest type, season and depth ($p<0.01$) (Table 6.14).

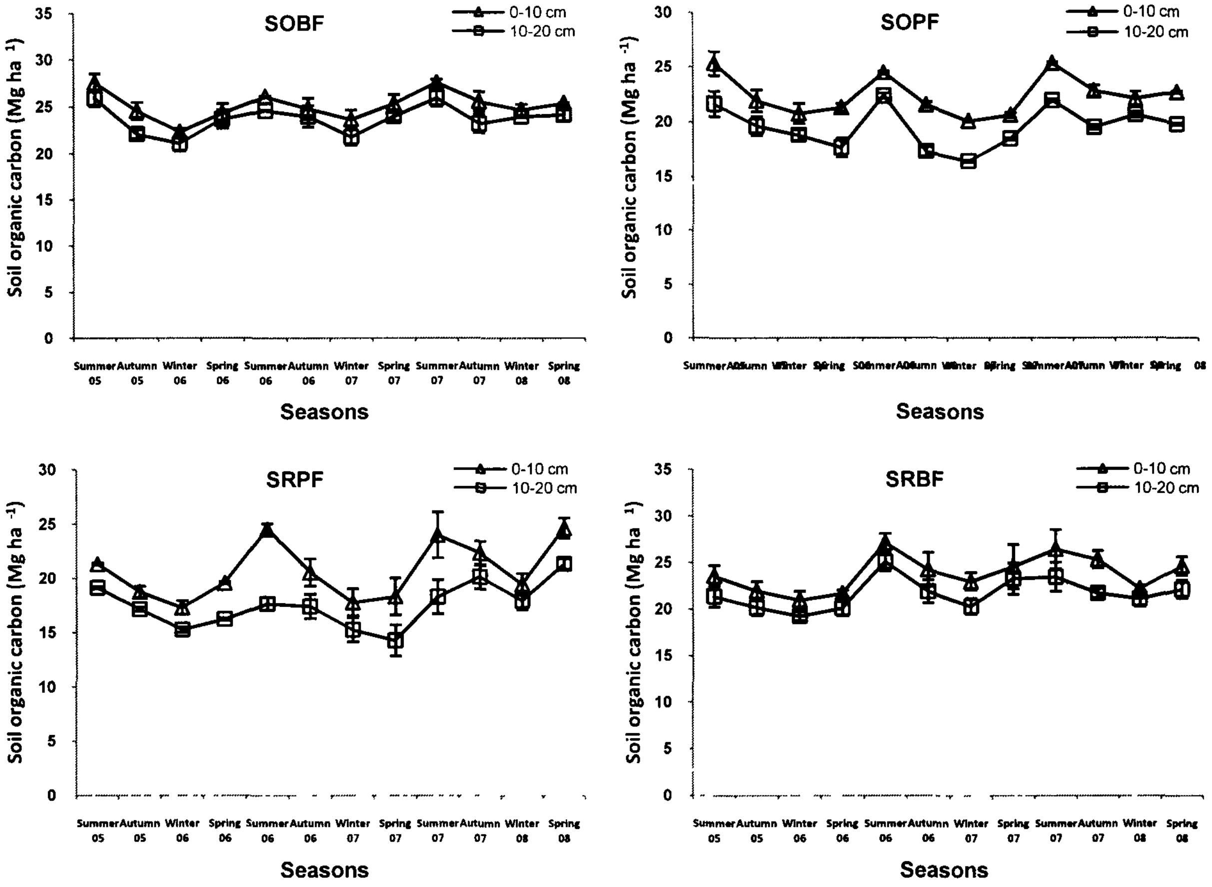


Fig. 6.7 Seasonal variation in the soil organic carbon (Mg ha^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.12 Three-way ANOVA showing effect of forest types, season and soil depth on soil organic carbon in different forest types in montane sub-tropical landscape of Upper Shillong (*P<0.01, **p<0.01)

Variation due to	df	SS	MS	F	*p*
Forest type	3	373.62	124.54	259.19**	0.000
Depth	1	140.37	140.37	292.13**	0.000
Season	11	215.08	19.55	40.69**	0.000
Forest type*Depth	3	11.14	3.71	7.73*	0.000
Forest type*Season	33	76.96	2.33	4.85*	0.000
Depth*Season	11	8.05	0.73	1.52	0.170
Forest type*Depth*Season	33	15.86	0.48	1.00	0.500

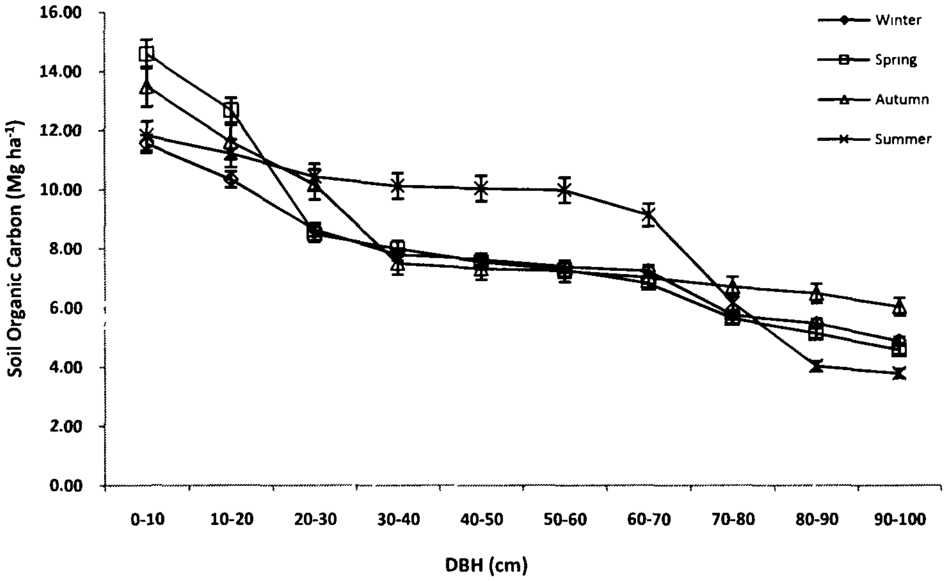


Fig. 6.8 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

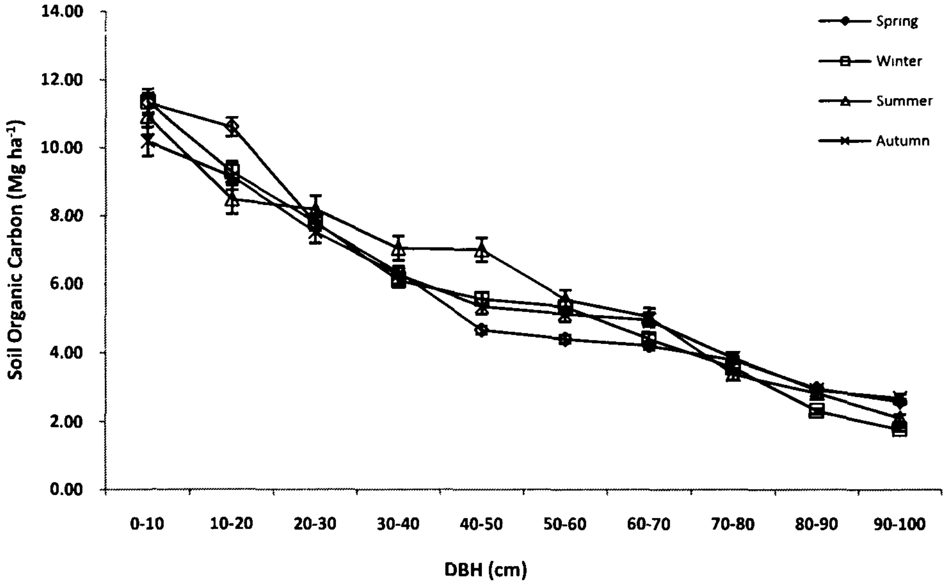


Fig. 6.9 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the old-growth pine (*Pinus kesiya*) forest in montane sub-tropical landscape of Upper Shillong

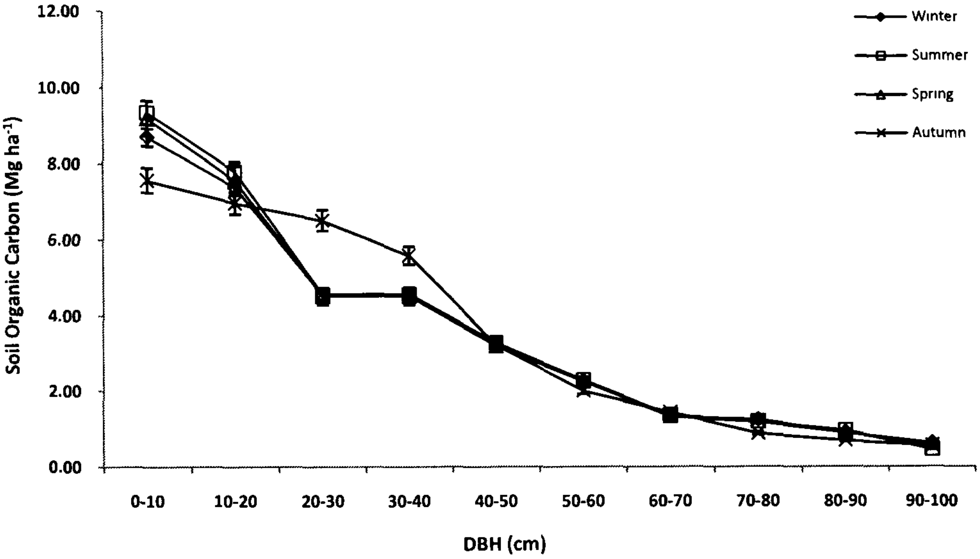


Fig. 6.10 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the regenerating pine (*Pinus kesiya*) forest in montane sub-tropical landscape of Upper Shillong

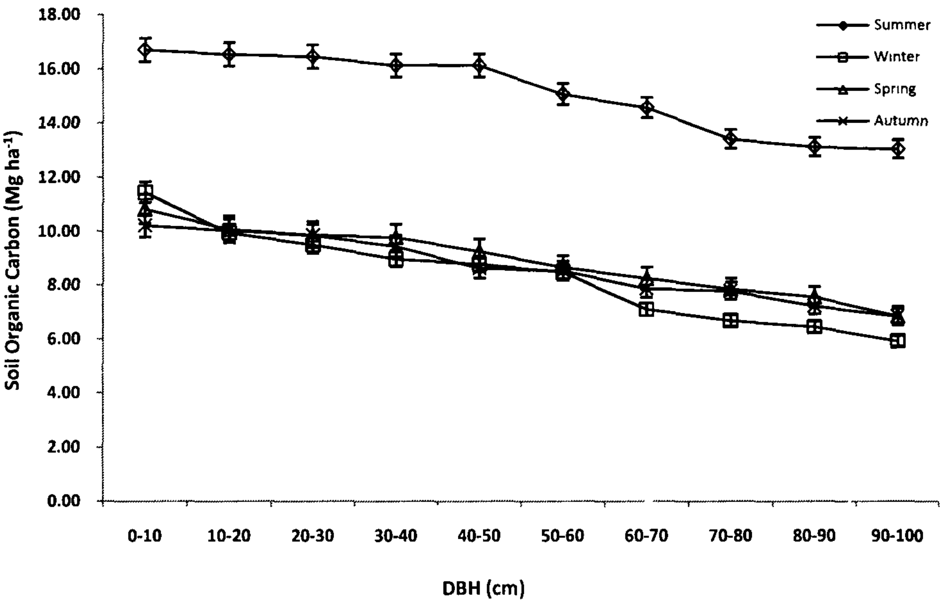


Fig. 6.11 Seasonal variation in soil organic carbon content (Mg ha^{-1}) (upto 1m) in the regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Table 6.13 Total SOC (Mg ha^{-1}) upto 1 m in different seasons in all montane sub-tropical forest types of Upper Shillong

Forest types	Spring	Summer	Autumn	Winter	Mean
SOBF	76.6	80.7	83.4	86.6	81.8
SOPF	58.6	60.5	58.0	58.7	57.5
SRPF	35.2	35.7	35.3	34.6	35.2
SRBF	88.9	151.0	86.3	83.2	102.4

Table 6.14 Three-way ANOVA showing effect of forest type, season and soil depth on soil organic carbon in different forest types in montane sub-tropical landscape of Upper Shillong (*$P<0.01$, **$p<0.01$)

Variation due to	df	SS	MS	F	*p*
Forest type	3	1010.33	336.78	823.73**	0.000
Depth	9	821.31	91.26	223.21**	0.000
Season	3	62.77	20.92	51.18**	0.000
Forest type*Depth	27	68.65	2.54	6.22*	0.000
Forest type*Season	9	260.46	28.94	70.78**	0.000
Depth*Season	27	16.36	0.61	1.48	0.091
Forest type *Depth*Season	81	33.12	0.41	1.00	0.500

C/ N ratio in the soil

The C/N ratio was greater in the sub-surface soil layer than the surface layer. In the montane sub-tropical landscape, highest C/N ratio was recorded in the old-growth broad-leaved forest (9.0 in the surface layer and 10.6 in the sub-surface layer) and lowest in the regenerating pine forest (4.8 in the surface layer and 4.9 in the sub-surface layer) (Table 6.15).

Table 6.15 Soil organic carbon, total kjeldahl nitrogen and C/N ratio in soils in different forest types in montane sub-tropical landscape of Upper Shillong. The values are mean of 12 seasons and five replicates each

Forest types	Depths (cm)	SOC (%)	TKN (%)	C/N
SOBF	0-10	2.51	0.28	8.96
	10-20	2.43	0.23	10.57
SOPF	0-10	2.41	0.46	5.24
	10-20	2.23	0.41	5.44
SRPF	0-10	2.24	0.47	4.77
	10-20	1.97	0.40	4.93
SRBF	0-10	2.06	0.30	6.87
	0-10	1.80	0.23	7.83

MICROBIAL BIOMASS CARBON POOL

The Microbial biomass carbon (MBC) both in the surface and sub-surface soil layers peaked during summer season (451.8 & 358.9 μg g^{-1}, respectively) in the old-growth broad-leaved forest and minimum during winter season (119.7 & 60.5 μg g^{-1}) in the regenerating pine forest. The mean seasonal MBC in the surface and sub-surface layers were highest in the regenerating broad-leaved forest (359.7 & 240.7 μg g^{-1}, respectively) and lowest in the regenerating pine forest (189.9 & 123.5 μg g^{-1}, respectively) (Fig. 6.12). Based on mean MBC concentration values in the surface soil layer, the regenerating broad-leaved forest had significantly greater MBC than the old-growth broad-leaved forest, old-growth pine forest, and regenerating pine forest. The surface soil layer had significantly ($p<0.01$) higher values than the sub-surface layer in all the forest types. Three-way ANOVA showed significant variation due to forest type, season and depth ($p<0.01$) (Table 6.16).

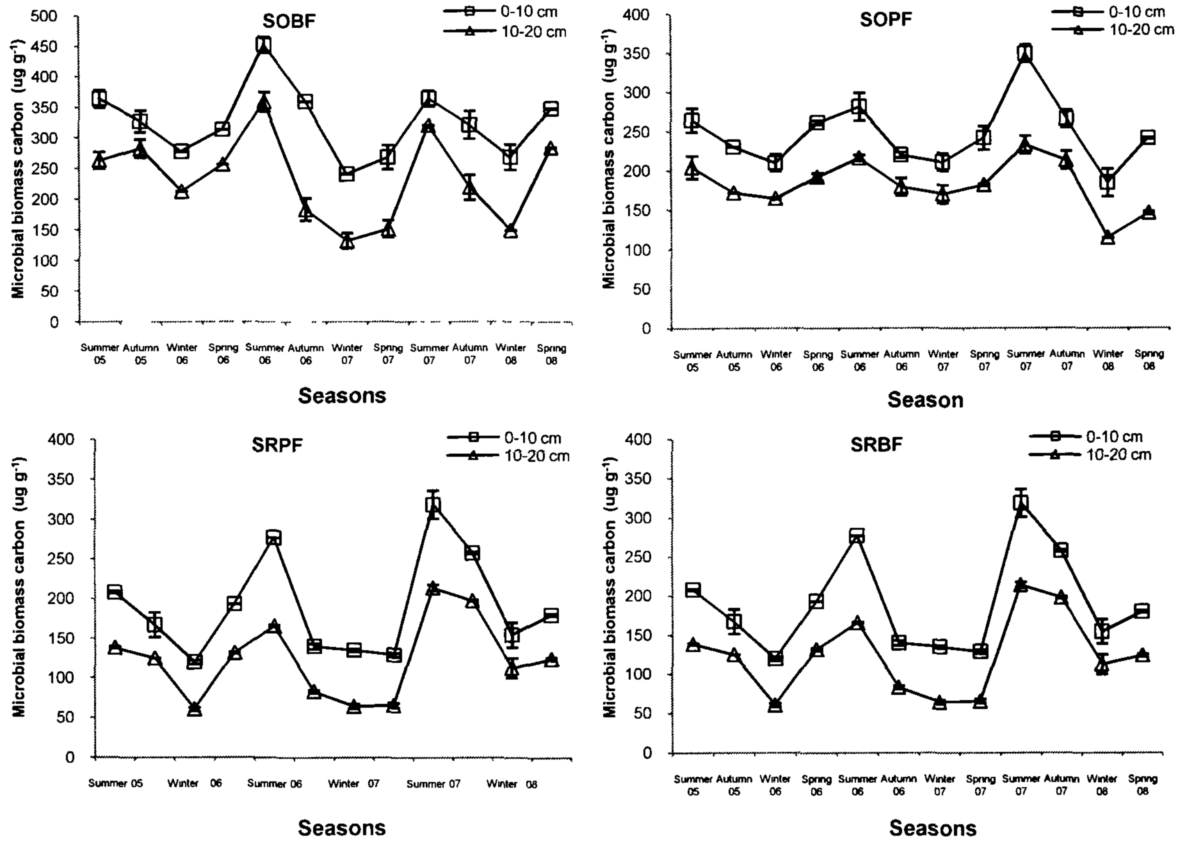


Fig. 6.12 Seasonal variation in the microbial biomass carbon (μg g^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.16 Three-way ANOVA showing the effect of forest type, season and depth on microbial biomass carbon in different forest types in montane sub-tropical landscape of Upper Shillong (*p<0.01, **p<0.01)

Variation due to	df	SS	MS	F	*p*
Forest type	3	306735.83	102245.28	224.13**	0.000
Depth	11	194147.67	17649.79	38.69**	0.000
Season	1	173975.27	173975.27	381.37**	0.000
Forest type*Depth	33	54188.54	1642.08	3.60*	0.000
Forest type*Season	3	11753.93	3917.98	8.59*	0.000
Depth*Season	11	5310.41	482.76	1.06	0.422
Forest type*Depth*Season	33	15054.07	456.18	1.00	0.500

Percentage contribution of MBC to SOC

During summer season the percentage contribution of MBC to SOC was highest in all the forest types and it was lowest during winter season. The percentage contribution was greater in the surface soil layer than that of sub-surface soil layer. The regenerating broad-leaved forest had greater percentage contribution (2.0), followed by old-growth broad-leaved forest (1.6), regenerating pine forest (1.6), and old-growth pine forest (1.4). In the sub-surface layer, it was highest in the regenerating broad-leaved forest (1.5) and lowest in old-growth pine forest (1.2) during summer season (Table 6.17).

Table 6.17 Percentage contribution of microbial biomass carbon to soil organic carbon content in different forest types in montane sub-tropical landscape of Upper Shillong. The seasons are mean of three years for the year 2005 to 2008

Forest types	Depth (cm)	Summer	Autumn	Winter	Spring
SOBF	0-10	1.6	1.2	1.1	1.3
	10-20	1.4	0.9	0.7	1.0
SOPF	0-10	1.4	1.0	0.9	1.2
	10-20	1.2	1.0	0.7	0.9
SRPF	0-10	1.6	1.1	1.1	1.1
	10-20	1.6	1.0	0.8	1.0
SRBF	0-10	2.0	1.5	1.3	1.4
	10-20	1.5	1.0	0.8	1.1

LITTER POOL

The total litterfall in different forest types ranged between 4.2 - 10.1 Mg ha^{-1} yr^{-1}. Leaf litter contribution ranged between 52.1 to 89.7 % to the total litterfall. Maximum (89.7%) contribution of leaf litter to the total litterfall was in the regenerating broad-leaved forest. The old-growth broad-leaved forest had the maximum litterfall (10.1 Mg ha^{-1} yr^{-1}), followed by old-growth pine (8.4 Mg ha^{-1} yr^{-1}), regenerating broad-leaved (6.1 Mg ha^{-1} yr^{-1}) and regenerating pine (4.2 Mg ha^{-1} yr^{-1}) forests, respectively (Fig. 6.13 & Table 6.18). The litterfall was maximum during winter season and minimum during summer season in all the forest types (Fig. 6.13). Two-way ANOVA revealed significant variation in litterfall among forest types (ANOVA F = 7.95; $p<0.01$) and seasons (ANOVA F = 2.82; $p<0.01$)

Litter decomposition

Litter showed an overall slow decomposition rate in all the forest types. Pine leaf litter could not be separated into distinct phases and both the pine forests showed a very slow rate of decomposition. However, in the old-growth broad-leaved forest and the regenerating broad-leaved forest where two phases could be distinguished. During the first 90 days, 45 - 74% of the litter remained undecomposed. This phase was followed by a period of slow weight loss lasting for 270 days (11 -31% weight remaining) (Fig. 6.14). The old-growth broad-leaved forest showed a faster rate of litter decomposition than the other forest types. The weight loss pattern followed the trend: SOBF>SRBF>SRPF>SOPF. The litter decay constant (k) was in the order SOBF>SRBF>SRPF>SOPF. The time required for 50% of litter to decay (t_{50}) varied among the forest types and ranged between 0.6 and 2.2. Similarly, t_{99} ranged between 4.0 and 16.1 (Table 6.19). The litter turnover rate varied among the different forest types. The SOBF, SOPF and SRBF showed high turnover rate (0.6, 0.5 & 0.5 yr^{-1}, respectively) and low in SRPF (0.4 yr^{-1}) (Table 6.20). The litter turnover time varied from 1.7 - 2.9 yr^{-1} (Table 6.21). Based on litter turnover rate, the forest types may be arranged in the order: SOBF>SOPF=SRBF>SRPF.

Table 6.18 Litterfall by different components (Mg ha^{-1}yr^{-1}) and their carbon content in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of 12 seasons and five replicate samples. The figures in parenthesis represent the percentage contribution of litter components to total litter

Forest types	Leaves	Twigs	Branches	Bark	Seed	Total	Total litter C
SOBF	7.0 (69.1)	2.4 (23.5)	0.5 (5.0)	0.1 (1.1)	0.1 (1.3)	10.1	4.6
SOPF	4.4 (52.1)	2.2 (26.2)	0.3 (3.2)	0.7 (8.1)	0.9 (10.4)	8.4	4.1
SRPF	2.5 (59.2)	0.8 (18.8)	0.1 (2.9)	0.2 (3.6)	0.7 (15.5)	4.2	2.1
SRBF	5.5 (89.7)	0.6 (9.8)	-	-	-	6.1	2.9

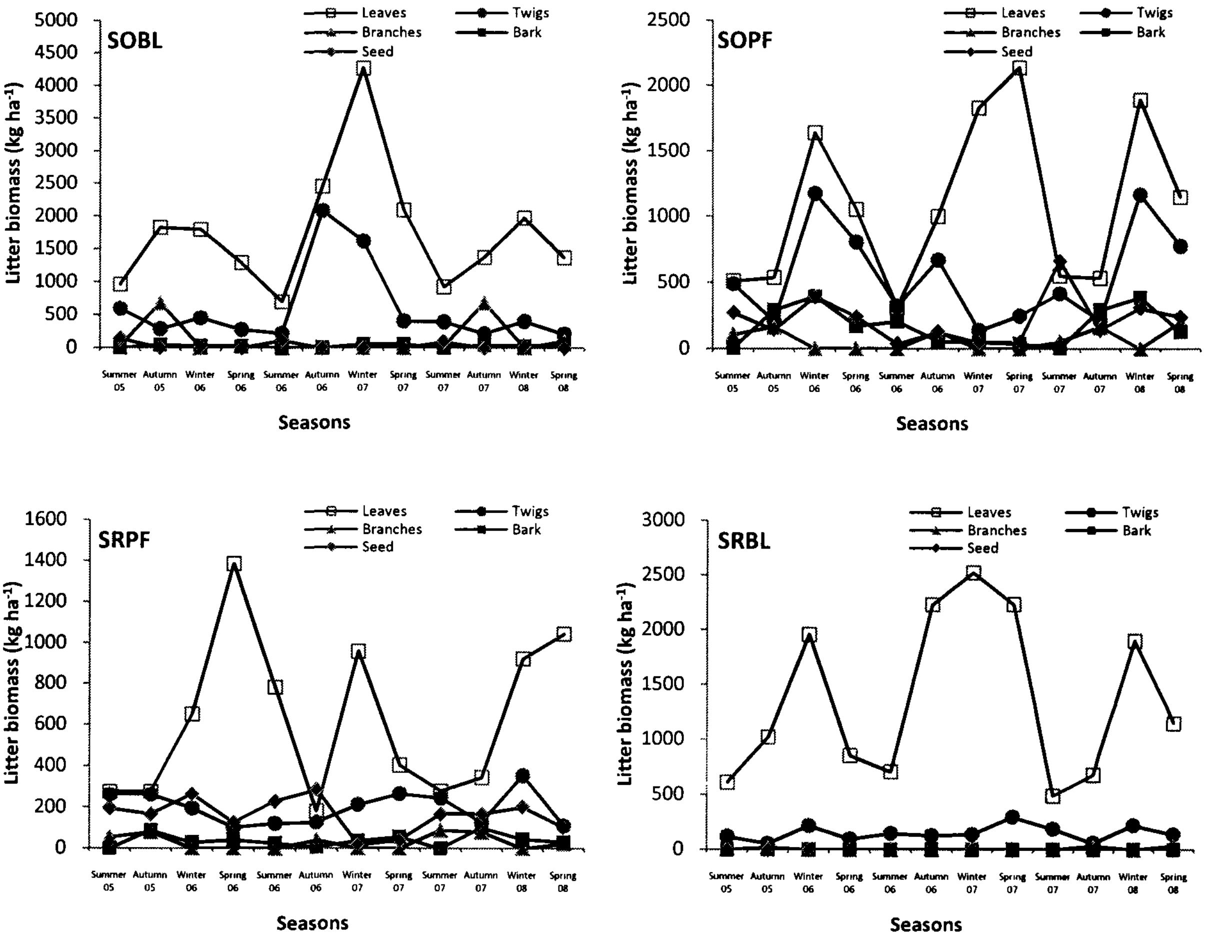


Fig. 6.13 Seasonal variation in litterfall (kg ha^{-1}) in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

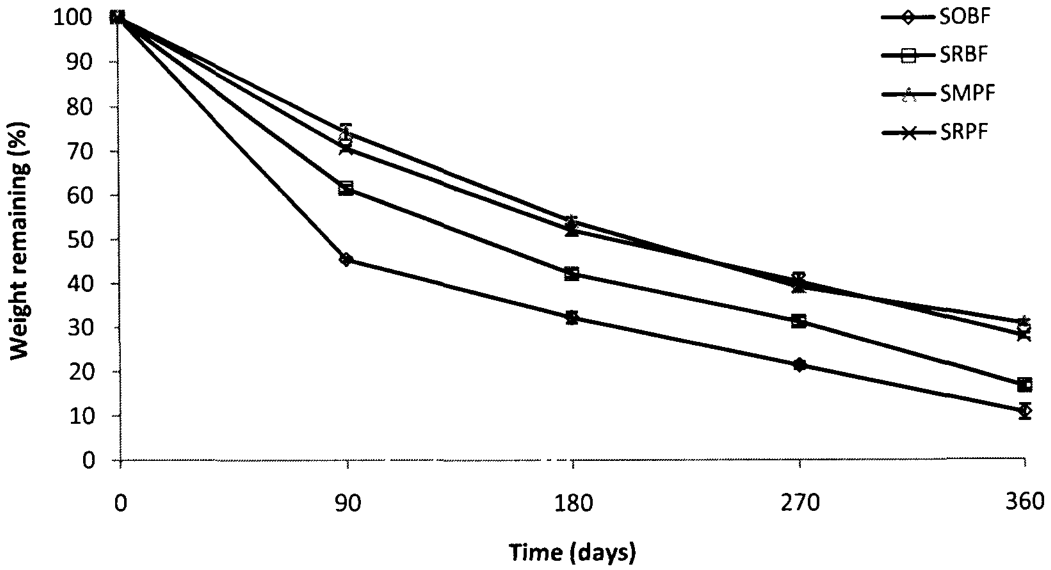


Fig. 6.14 Weight remaining (%) in different forest types in montane sub-tropical landscape of Upper Shillong *in situ* leaf litter decomposition. The values are mean (±SE) of five replicate samples

Table 6.19 Annual decay constant (k) of litter in different forest types in montane sub-tropical landscape of Upper Shillong

Parameters	SOBF	SOPF	SRPF	SRBF
Decay constants				
k	0.3	1.3	1.1	0.8
t_{50}	2.2	0.6	0.6	0.8
t_{99}	16.1	4.0	4.5	6.0

Table 6.20 Litter turnover rate (k_L yr^{-1}) in different forest types in montane sub-tropical landscape of Upper Shillong

Forest types	Litter turnover rate ($k_L yr^{-1}$)					
	Leaves	Twigs	Branches	Bark	Seed	Total
SOBF	0.6	1.1	2.0	1.2	1.7	0.6
SOPF	0.7	0.9	1.6	0.8	1.0	0.5
SRPF	0.7	0.4	1.2	1.1	0.4	0.4
SRBF	0.5	0.6	4.0	0.0	0.0	0.5

Table 6.21 Litter turnover time (t yr^{-1}) in different forest types in montane sub-tropical landscape of Upper Shillong

Forest types	Turnover time (t yr^{-1})					
	Leaves	Twigs	Branches	Bark	Seed	Total
SOBF	1.8	0.9	0.5	0.0	0.0	1.9
SOPF	1.5	1.1	0.6	0.0	0.0	1.7
SRPF	1.4	2.5	0.8	0.0	0.0	2.9
SRBF	2.0	1.6	0.3	0.0	0.0	2.2

Aboveground and belowground biomass pool

Old-growth broad-leaved forest

The allometric model of Chambers *et al.* (2001a) for broad-leaved tree aboveground biomass (AGB) was used to determine AGB of broad-leaved tree component in the old-growth broad-leaved forest. The model was:

$$Y_1 = \exp[-0.37 + 0.33*\ln(D) + 0.933*\ln(D)^2 - 0.122*\ln(D)^3]$$

Similarly, the model of Cairns *et al.* (1997) for broad-leaved tree belowground biomass (BGB) was to determine BGB of broad-leaved tree component in the forest. The model was:

$$Y_2 = \exp[-1.0587 + 0.8836*\ln(AGB)]$$

where, Y= biomass/tree, D= diameter at breast height, AGB= aboveground biomass.

The carbon (%) derived for the sub-tropical forest types were used for determination of carbon content (Table 6.22).

Table 6.22 Carbon content (%) in different tree components of montane sub-tropical landscape of Upper Shillong

Forest types	Aboveground (%)					Belowground (%)		Litter (%)			
	Leaf/ Needle	Twig	Branch	Cone	Stem	Fine root	Coarse root	Leaf/ Needle	Twig	Branch	Misc.
SOBF	.................49.....................					...... 47.....		47	47	48	47
SRBF	49	49	49	NA	49	.......47.....		47	47	47	47
SOPF	49	49	49	49	49	.......47.....		47	49	49	49
SRPF	48	48	48	48	49	.......47.....		47	47	47	47

Density diameter distribution

Total tree density in the forest was 980 individuals ha^{-1}. The diameter class >20-30 cm had the maximum number of trees (376 individuals ha^{-1}) contributing 38.4% to the total density in the forest. The higher diameter class i.e. >60 cm had 29 trees and contributed 3% to the total tree density in the forest (Fig. 6.15).

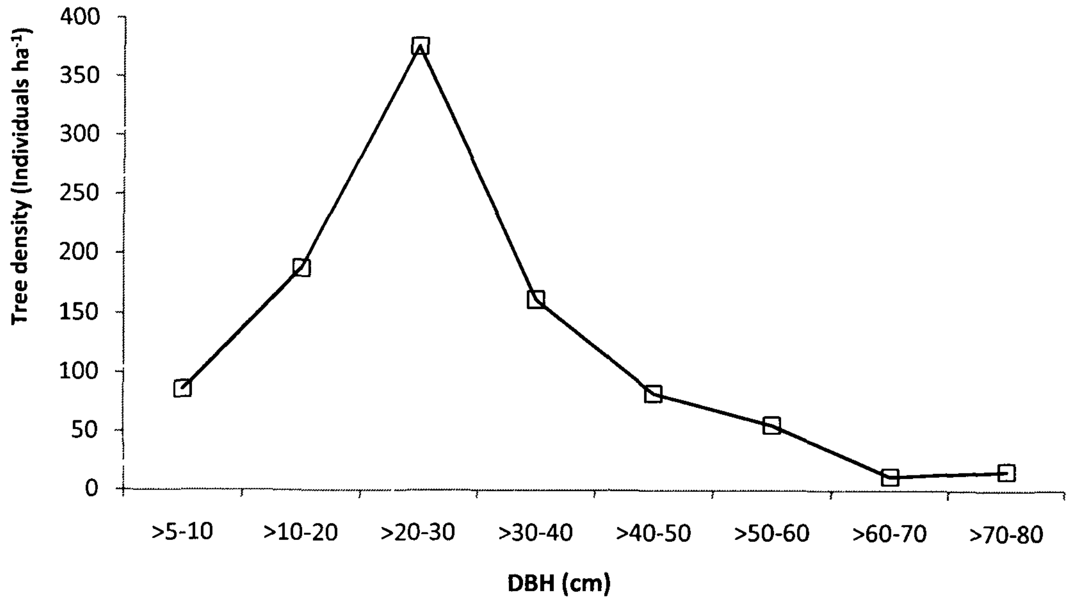


Fig. 6.15 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Aboveground biomass and carbon

Tree aboveground biomass in the old-growth broad-leaved forest was maximum in the >20-30 and >30-40 cm diameter classes. Both the diameter classes contributed 59.5% to the total tree aboveground biomass in the forest. The higher diameter classes i.e. >60 cm contributed 10.5% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon were 250.5 Mg ha^{-1} and 122.8 Mg C ha^{-1}, respectively (Table 6.23).

Table 6.23 Aboveground biomass and carbon in different diameter classes of old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Aboveground tree biomass (Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	1.1	0.5	0.4
>10-20	22.2	10.9	8.9
>20-30	86.3	42.3	34.4
>30-40	62.8	30.8	25.1
>40-50	28.3	13.9	11.3
>50-60	23.7	11.6	9.5
>60-70	15.5	7.6	6.2
>70-80	10.7	5.3	4.3
Total	250.5	122.8	100.0

Belowground biomass and carbon

Tree belowground biomass was highest in the >20-30 and >30-40 cm diameter classes. These diameter classes contributed 61.1% to the total belowground biomass in the forest. The higher diameter classes i.e. >60 cm contributed 8.9% to the total tree belowground biomass in the forest. The total tree belowground biomass and carbon were 57.1 Mg ha^{-1} and 26.8 Mg C ha^{-1}, respectively (Table 6.24).

Table 6.24 Belowground biomass and carbon in different diameter classes of old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Belowground tree biomass		
	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	0.3	0.2	0.6
>10-20	5.9	2.8	10.3
>20-30	20.8	9.8	36.4
>30-40	14.1	6.6	24.7
>40-50	6.1	2.9	10.6
>50-60	4.9	2.3	8.5
>60-70	3.0	1.4	5.3
>70-80	2.1	1.0	3.6
Total	57.1	26.8	100.0

Herb and Shrub biomass and carbon

The herb and shrub biomass were 11.2 and 40.3 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 5.3 and 19.4 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (260.7 Mg ha^{-1}) in the forest i.e. 0.004 and 0.02%, respectively (Table 6.25).

Table 6.25 Herb and shrub biomass and carbon and % contribution to total AGB in different forest types in montane sub-tropical landscape of Upper Shillong

Forest types	Herb biomass and carbon		Shrub biomass and carbon		%ABG	%AGB
	(kg ha^{-1})	(kg C ha^{-1})	(kg ha^{-1})	(kg C ha^{-1})	Herb	Shrub
SOBF	11.23	5.28	40.34	19.36	0.004	0.02
SOPF	120.32	56.55	560.32	268.95	0.028	0.13
SRPF	119.65	56.24	199.76	95.88	0.375	0.63
SRBF	9.78	4.60	39.78	19.09	0.105	0.43

Old-growth pine forest

The following model was fitted to estímate AGB and BGB of old-growth pine trees:

$$\text{Log}(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$$

where, Y= AGB (kg/tree), a, b, c, and d are regression coefficients, and D is the tree diameter at breast height. All the measures of coefficients of the models for tree components were statistically significant ($P < 0.001$). Models were evaluated by comparing coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE). The dry weights of various tree components of *Pinus kesiya* were obtained for the development of tree component regression models (Table 6.26) across a range of DBH. The component-wise models i.e. regression coefficients, along with R^2, SD, SSE, MSE and RMSE for the species are presented in Table 6.27. The goodness of fit of the models for various tree components of *Pinus kesiya* has been depicted in Fig. 6.16.

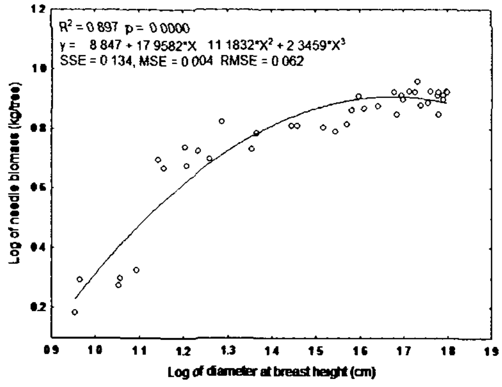
Table 6.26 Dry weight (kg tree^{-1}) of different components of old-growth *Pinus kesiya* trees used for developing regression models (AGB=aboveground biomass, BGB=total belowground biomass)

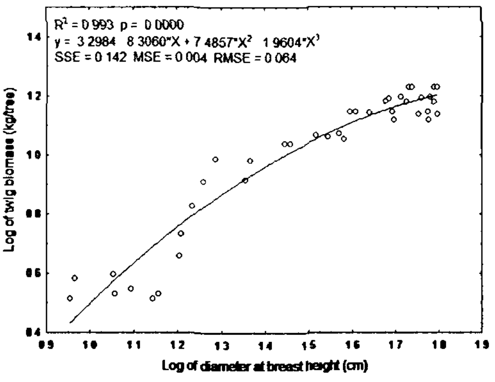
DBH (cm)	Stem	Branch	Twig	Needle	Cone	Fine root	Coarse root	Total BGB	Total AGB	Total tree biomass
9.0	15.2	5.0	3.3	1.5	0.4	0.3	2.2	2.5	25.4	27.9
9.2	15.6	6.3	3.8	2.0	0.4	1.0	2.5	3.5	28.1	31.6
11.3	19.6	8.1	4.0	1.9	0.9	0.3	3.2	3.4	34.5	37.9
11.4	20.1	8.5	3.4	2.0	1.0	0.3	3.6	3.9	35.0	38.9
12.4	22.2	9.7	3.5	2.1	0.8	0.6	6.8	7.3	38.4	45.7
13.9	32.5	13.0	3.3	5.0	2.2	1.0	11.7	12.6	56.0	68.6
14.3	48.3	14.0	3.4	4.7	2.4	1.1	12.8	13.9	72.7	86.6
16.0	41.1	19.6	4.6	5.5	2.3	1.4	17.5	18.9	73.1	92.0
16.1	63.1	23.9	5.4	4.8	2.6	1.5	17.9	19.4	99.8	119.2
17.1	81.4	28.3	6.7	5.3	2.5	1.7	20.6	22.3	124.2	146.6
18.1	88.8	35.5	8.2	5.0	3.3	1.9	23.2	25.1	140.8	165.9
19.3	72.9	22.5	9.7	6.7	2.6	2.1	26.1	28.2	114.4	142.6

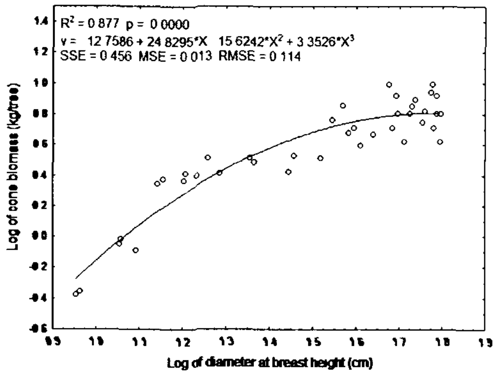
22.6	121.5	37.2	8.3	5.4	3.3	2.7	33.4	36.1	175.7	211.8
23.2	133.9	48.3	9.6	6.1	3.1	2.8	34.5	37.3	200.9	238.3
27.8	287.4	84.0	11.0	6.5	2.7	3.5	42.6	46.1	391.4	437.5
28.6	287.4	83.9	11.0	6.5	3.4	3.6	43.9	47.4	392.1	439.6
32.8	354.5	86.1	11.7	6.4	3.3	4.0	49.2	53.2	462.0	515.2
35.0	376.5	80.6	11.5	6.2	5.8	4.2	51.6	55.8	480.7	536.4
37.1	425.5	82.3	11.9	6.6	7.2	4.4	53.4	57.7	533.4	591.1
38.1	478.4	85.0	11.3	7.3	4.8	4.4	54.1	58.6	586.8	645.4
39.3	502.5	90.6	14.1	8.2	5.2	4.5	55.0	59.4	620.5	679.9
40.6	578.5	88.9	14.0	7.4	4.0	4.5	55.8	60.3	692.8	753.1
43.5	640.4	93.0	13.9	7.5	4.7	4.7	57.2	61.8	759.5	821.4
47.5	722.3	105.9	15.2	8.4	9.9	4.8	58.5	63.2	861.7	924.9
48.3	730.4	101.7	15.6	7.1	5.2	4.8	58.6	63.4	859.9	923.3
49.2	739.8	106.1	14.0	8.2	8.4	4.8	58.8	63.6	876.5	940.1
49.7	750.9	102.0	13.1	7.9	6.4	4.8	58.9	63.7	880.4	944.0
51.4	769.4	110.4	15.7	8.4	4.2	4.8	59.1	63.9	908.0	971.9
53.0	799.5	114.3	15.1	8.4	6.4	4.8	59.1	63.9	943.6	1007.6
53.6	815.1	105.9	17.0	9.1	7.1	4.8	59.2	64.0	954.3	1018.3
54.6	836.4	124.4	16.9	7.6	7.9	4.8	59.2	64.0	993.2	1057.1
56.7	961.7	115.5	13.8	7.7	5.6	4.8	59.1	63.9	1104.4	1168.2
57.4	1000.3	114.4	15.6	8.5	6.7	4.8	59.0	63.8	1145.5	1209.3
59.5	1046.0	93.0	14.0	8.2	8.8	4.8	58.8	63.6	1170.0	1233.6
59.9	1056.2	101.3	13.1	8.4	9.9	4.8	58.8	63.5	1188.9	1252.5
60.1	1080.3	105.9	15.7	7.1	5.2	4.8	58.8	63.5	1214.2	1277.7
61.3	1085.2	101.7	15.1	8.2	8.4	4.8	58.6	63.4	1218.5	1281.8
61.3	1104.7	106.1	17.0	7.9	6.4	4.8	58.6	63.4	1242.1	1305.5
62.4	1142.4	102.0	16.9	8.4	4.2	4.7	58.5	63.2	1273.9	1337.1
62.7	1184.1	110.4	13.8	8.4	6.4	4.7	58.4	63.2	1323.0	1386.1

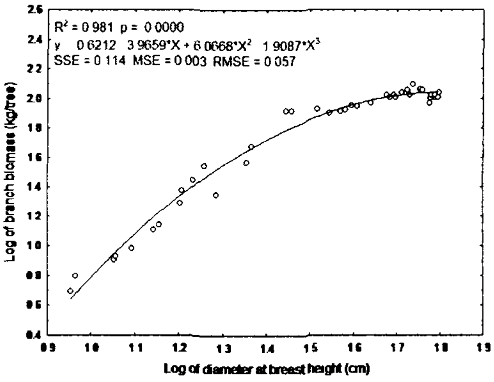
Table 6.27 Regression coefficients (a, b, c and d), coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) in respect of the models for biomass estimation of individual tree components and total tree biomass in montane old-growth pine forest of Upper Shillong. The model is of the form $Log(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$, where Y= biomass of individual tree components/BGB/AGB/total tree expressed in dry weight (kg $tree^{-1}$) , AGB/BGB = above- and below-ground biomass and D = diameter at breast height (n=40). The model validity is between 9 cm and 63 cm DBH

Dependent variables (Y)	Coefficients				R^2	SD	SSE	MSE	RMSE
	a	b	c	d					
Stem	1.8364	-5.3988	6.5046	-1.7410	0.993	0.610	0.096	0.003	0.052
Branch	0.6212	-3.9659	6.0668	-1.9087	0.981	0.396	0.114	0.003	0.057
Twigs	3.2984	-8.3060	7.4857	-1.9604	0.993	0.235	0.142	0.004	0.064
Needles	-8.8470	17.9582	-11.1832	2.3459	0.897	0.185	0.134	0.004	0.062
Cone	-12.7586	24.8295	-15.6242	3.3526	0.877	0.312	0.456	0.013	0.114
Fine root	1.5470	-8.2181	8.9111	-2.5697	0.904	0.348	0.442	0.013	0.112
Coarse root	-12.2742	22.8461	-12.3419	2.2129	0.985	0.386	0.084	0.002	0.049
Total BGB	-9.8635	17.8955	-8.9558	1.4511	0.978	0.379	0.119	0.003	0.058
Total AGB	1.3503	-3.4145	4.8678	-1.3520	0.993	0.540	0.073	0.002	0.046
Total tree biomass	-0.3686	0.3859	2.2618	-0.7785	0.994	0.522	0.063	0.002	0.043









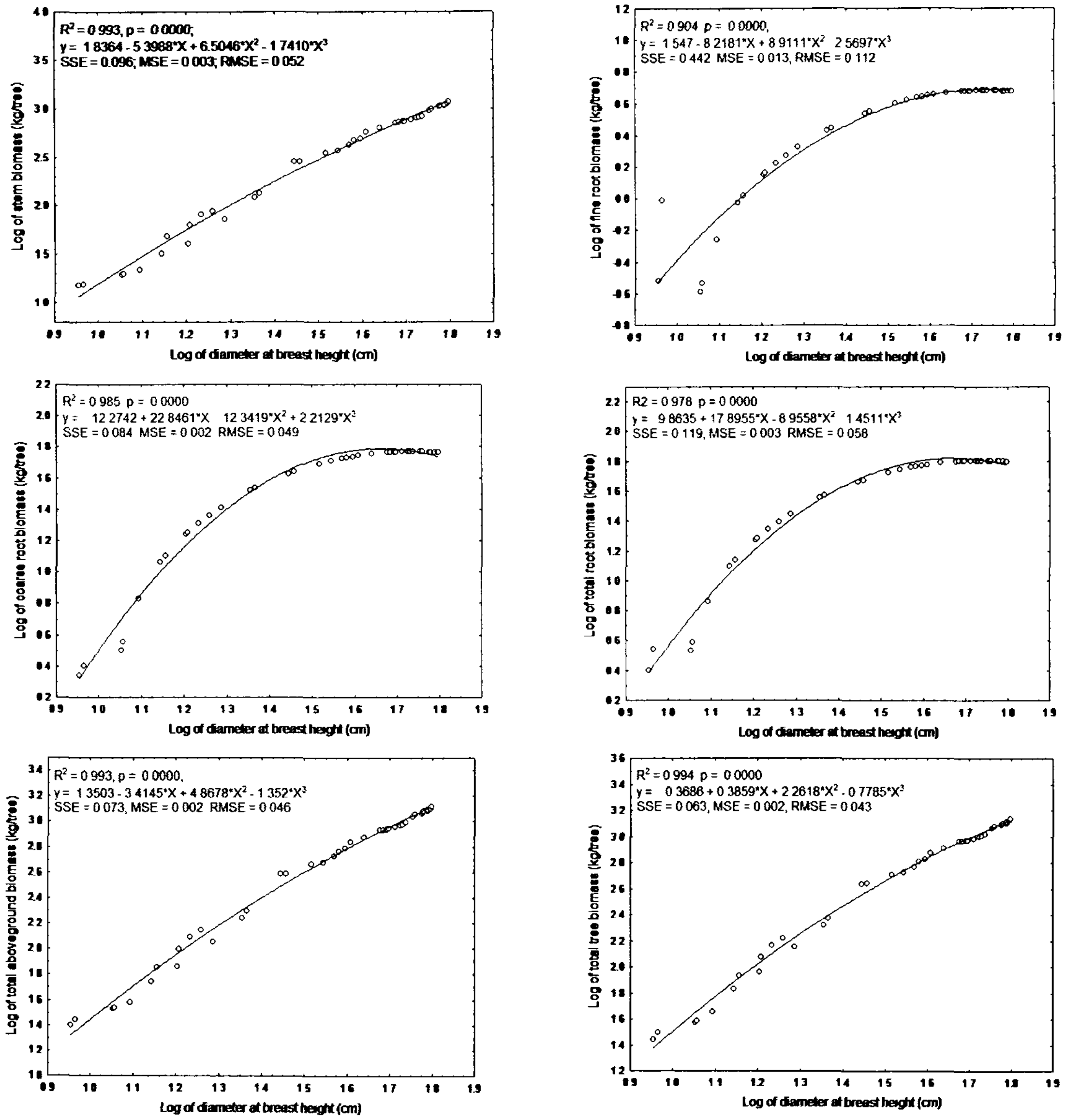


Fig. 6.16 Regression analyses between log of tree diameter at breast height (cm), and biomass of different tree components, total aboveground biomass and total tree biomass of old-growth *Pinus kesiya* forest

The allometric model of Chambers *et al.* (2001a) for broad-leaved tree aboveground biomass (AGB) was used to determine AGB of broad-leaved tree component in the old-growth pine forest. The model was:

$Y_l = \exp[-0.37 + 0.33*\ln(D) + 0.933*\ln(D)^2 - 0.122*\ln(D)^3]$

Similarly, the model of Cairns *et al.* (1997) for broad-leaved tree belowground biomass (BGB) was to determine BGB of broad-leaved tree component in the old-growth pine forest. The model was:

Y_2= exp [- 1.0587 + 0.8836*ln(AGB)]

where, Y= biomass/tree, D= diameter at breast height, AGB= aboveground biomass.

Density diameter distribution

The total tree density of the forest was 628 individuals ha^{-1}. The diameter class >50-60 cm had the maximum number of trees (260 individuals ha^{-1}) contributing 41.4% to the total density in the forest. The higher diameter class i.e. >60 cm had 24 trees contributing 3.8% to the total tree density in the forest (Fig. 6.17).

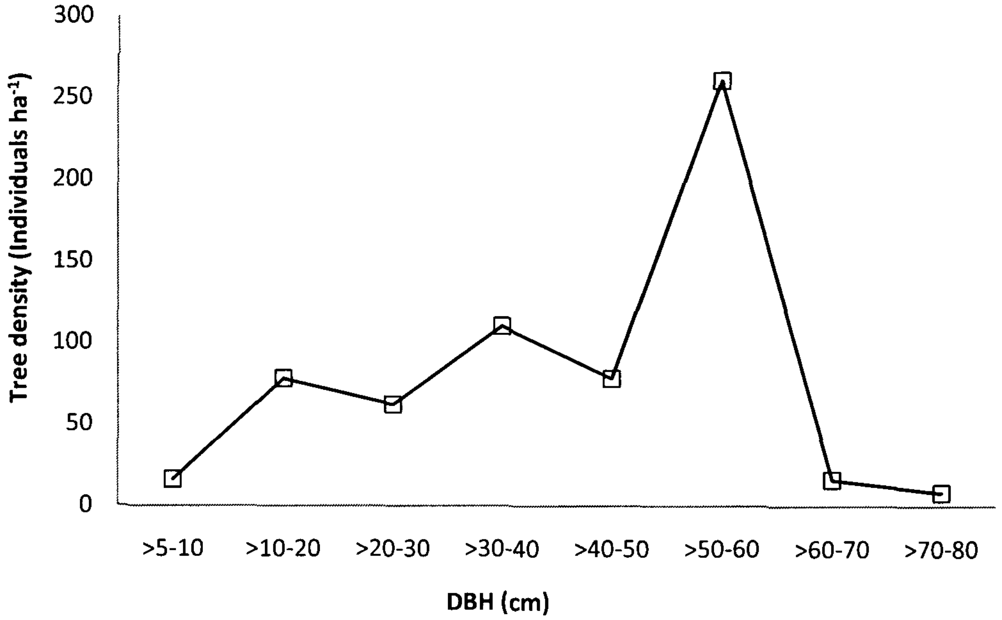


Fig. 6.17 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of old-growth pine forest in montane sub-tropical landscape of Upper Shillong

Aboveground biomass and carbon

Tree aboveground biomass was high in the >30-40 cm, >40-50 cm and >50-60 cm diameter classes. These diameter classes contributed 73.9% to the total tree aboveground biomass in the forest. The diameter classes i.e. >60 cm contributed 15.8% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon in the forest were 416.8 Mg ha^{-1} and 204.2 Mg C ha^{-1}, respectively (Table 6.28).

Table 6.28 Aboveground biomass and carbon in different diameter classes of old-growth *Pinus kesiya* forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Aboveground tree biomass		
	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	2.7	1.3	0.6
>10-20	7.4	3.6	1.8
>20-30	32.7	16.0	7.8
>30-40	111.6	54.7	26.8
>40-50	80.7	39.5	19.4
>50-60	115.9	56.8	27.8
>60-70	56.4	27.7	13.5
>70-80	9.5	4.7	2.3
Total	416.8	204.2	100.0

Belowground biomass and carbon

The tree belowground biomass in the old-growth pine forest was high in the >30-40 cm, >40-50 cm and >50-60 cm diameter classes. These diameter classes contributed 64.1% to the total belowground biomass in the forest. The higher diameter classes i.e. >60 cm contributed 19.2% to the total tree belowground biomass in the forest. The total tree belowground biomass and carbon were 40.7 Mg ha^{-1} and 18.7 Mg C ha^{-1}, respectively (Table 6.29).

Table 6.29 Belowground biomass and carbon in different diameter classes of old-growth *Pinus kesiya* forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Belowground tree biomass		
	(Mg ha^{-1})	(Mg C ha^{-1})	%
>5-10	0.6	0.3	1.5
>10-20	1.5	0.7	3.8
>20-30	4.6	2.1	11.4
>30-40	11.0	5.0	26.9
>40-50	7.4	3.4	18.0
>50-60	7.8	3.6	19.2
>60-70	6.5	3.0	15.9
>70-80	1.4	0.6	3.4
Total	40.8	18.7	100.0

Herb and Shrub biomass and biomass carbon

The herb and shrub biomass were 120.3 and 560.3 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 56.5 and 268.9 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (425.8 Mg ha^{-1}) in the forest i.e. 0.03 and 0.13%, respectively (Table 6.25).

Regenerating Pinus kesiya forest

The following models were fitted to estimate AGB and BGB of regenerating *Pinus kesiya* trees:

$$\text{Log}(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$$

where, Y= AGB (kg/tree), a, b, c, and d are regression coefficients, and D is the tree diameter at breast height. All the measures of coefficients of the models for tree components were statistically significant ($P < 0.001$). Models were evaluated by comparing coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE). The dry weights of various tree components of

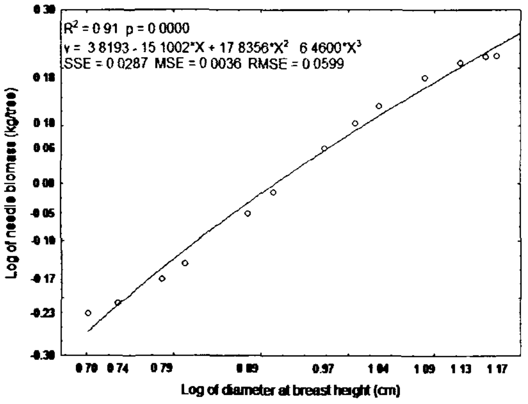
Pinus kesiya were obtained for the development of tree component regression models (Table 6.30) across a range of DBH. The component-wise models i.e. regression coefficients, along with R^2, SD, SSE, MSE and RMSE for the species is presented in Tables 6.31. The goodness of fit of the models for various tree component s of *Pinus kesiya* has been depicted in Fig. 6.18.

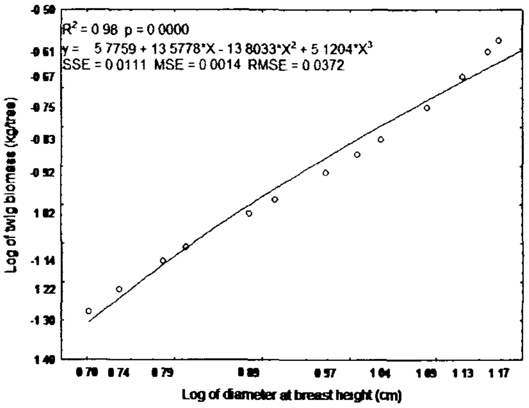
Table 6.30 Dry weight (kg tree^{-1}) of different components of regenerating pine trees of Upper Shillong used for developing regression models (AGB=aboveground biomass, BGB=total belowground biomass)

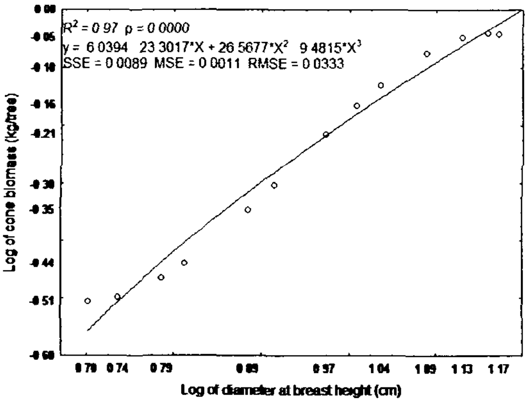
DBH (cm)	Dry wt. (kg tree^{-1})									
	Stem	Branch	Twig	Needle	Cone	Total AGB	Fine root	Coarse root	Total BGB	Total tree biomass
5.0	6.5	1.6	0.1	0.7	0.2	9.1	0.3	1.5	1.8	10.9
5.5	6.9	1.5	0.1	0.7	0.3	9.4	0.3	1.6	1.9	11.3
6.1	7.3	1.6	0.1	0.7	0.3	10.1	0.3	1.8	2.1	12.1
6.5	8.8	1.8	0.1	0.6	0.4	11.5	0.3	1.8	2.4	13.9
7.7	9.7	1.4	0.1	0.9	0.4	12.5	0.6	2.3	2.9	15.4
8.2	10.2	2.0	0.1	1.1	0.5	13.9	0.6	2.4	3.0	17.0
9.4	10.7	2.6	0.1	1.2	0.7	15.2	0.7	2.5	3.2	18.4
10.2	11.3	3.1	0.1	1.3	0.8	16.7	0.7	2.5	3.2	19.8
10.9	12.2	3.1	0.2	1.4	0.7	17.6	0.7	2.3	3.0	20.6
12.3	13.4	3.6	0.2	1.4	0.8	19.4	0.7	2.4	3.1	22.5
13.5	14.2	3.6	0.2	1.4	0.8	20.3	0.8	2.4	3.1	23.4
14.5	14.9	3.2	0.2	1.7	0.9	20.9	0.8	2.7	3.5	24.3
14.9	15.2	3.7	0.3	1.9	1.0	22.0	0.8	2.7	2.7	24.7

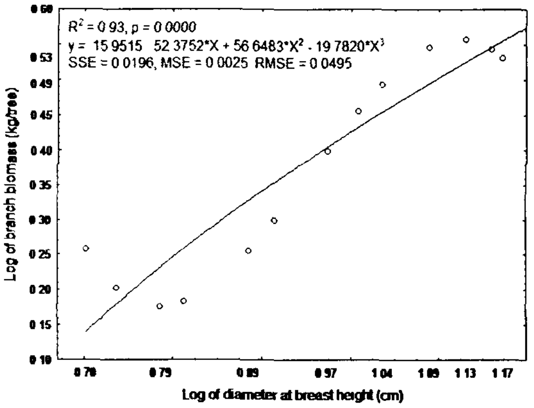
Table 6.31 Regression coefficients (a, b, c and d), coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) in respect of the models for biomass estimation of individual tree components and total tree biomass in montane sub-tropical regenerating pine forest of Upper Shillong. The model is of the form $Log(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$, where Y= biomass of individual tree components/BGB/AGB/total tree expressed in dry weight (kg $tree^{-1}$) and D = diameter at breast height, AGB/BGB = above- and below round biomass (n=13). The model validity is between 5 cm and 15 cm DBH

Dependent variables (Y)	Coefficients				R^2	SD	SSE	MSE	RMSE
	a	b	c	d					
Stem	-2.823	10.378	-9.744	3.249	0.980	0.120	0.002	0.002	0.015
Branch	15.952	-52.375	56.648	-19.782	0.930	0.160	0.020	0.003	0.050
Twig	-5.776	13.578	-13.803	5.120	0.980	0.220	0.011	0.001	0.037
Needles	3.819	-15.100	17.836	-6.460	0.910	0.170	0.029	0.004	0.060
Cones	6.039	-23.302	26.568	-9.482	0.970	0.180	0.009	0.001	0.033
Total AGB	1.087	-1.906	3.341	-1.309	0.990	0.130	0.001	0.000	0.012
Fine root	8.974	-30.702	32.539	-11.048	0.930	0.180	0.170	0.021	0.146
Coarse root	-4.372	14.582	-15.129	5.278	0.890	0.080	0.039	0.005	0.069
Total BGB	-1.765	6.417	-6.520	2.326	0.830	0.090	0.053	0.007	0.081
Totaltree biomass	0.520	0.421	0.693	-0.351	0.990	0.120	0.003	0.000	0.018









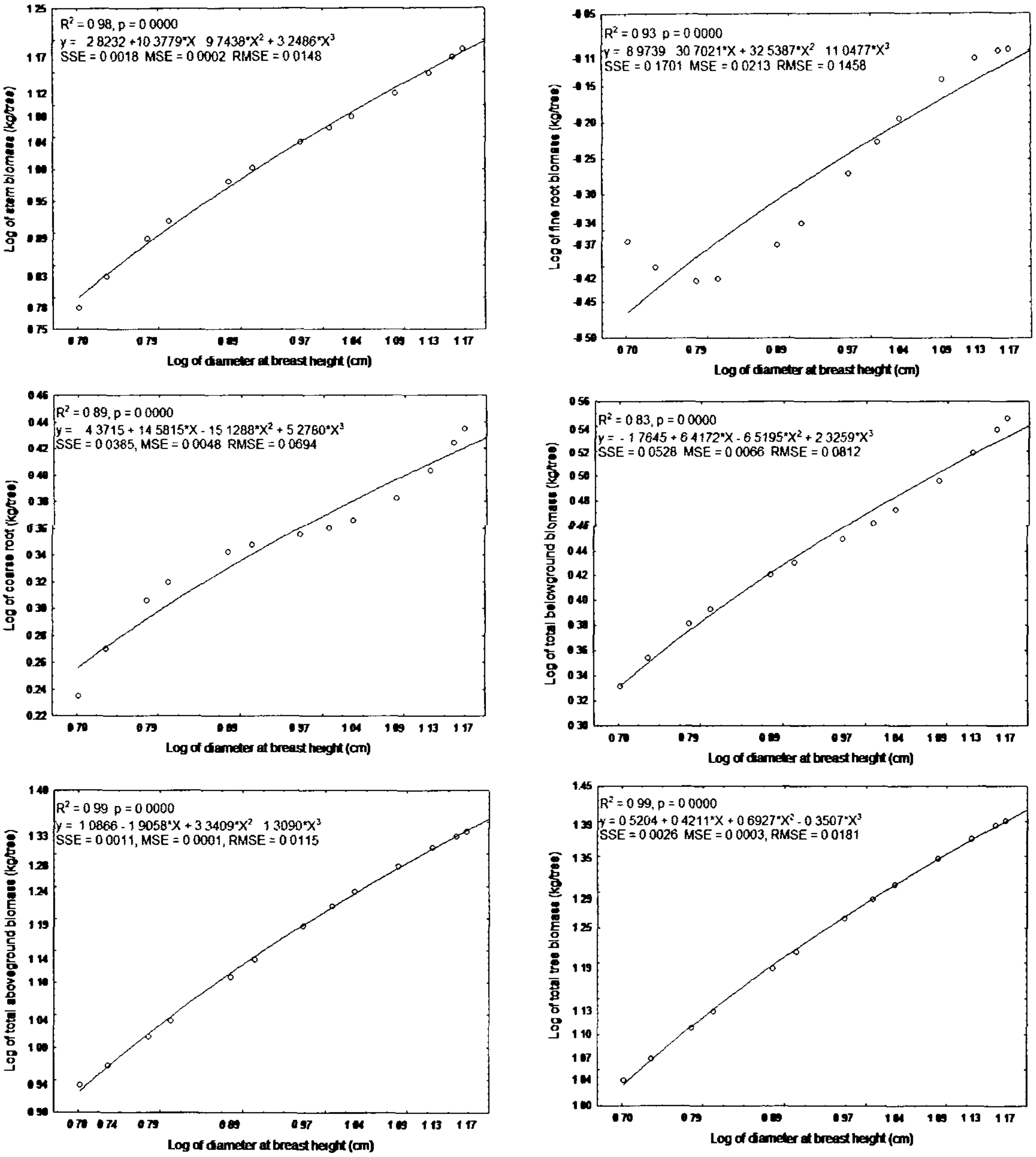


Fig. 6.18 Regression analyses between log of tree diameter at breast height (cm), and biomass of different tree components, total above ground biomass and total tree biomass of regenerating *Pinus kesiya* forest

The allometric model of Chambers *et al.* (2001a) for broad-leaved aboveground biomass (AGB) was used to determine AGB of broad-leaved tree component in the regenerating pine forest. The model was:

$Y_1 = \exp[-0.37 + 0.33*\ln(D) + 0.933*\ln(D)^2 - 0.122*\ln(D)^3]$

Similarly, the model of Cairns *et al.* (1997) for broad-leaved belowground biomass (BGB) was to determine BGB of broad-leaved tree component in the regenerating pine forest. The model was:

$Y_2 = \exp[-1.0587 + 0.8836*\ln(AGB)]$

where, Y= biomass/tree, D= diameter at breast height, AGB= aboveground biomass.

Density diameter distribution

Total tree density in the regenerating pine forest was 1768 individuals ha^{-1}. The diameter class >10-15 cm had the maximum number of trees (1068 individuals ha^{-1}) contributing 60.4% to the total density in the forest. The lower diameter class <10 cm had 540 trees contributing 30.5% to the total tree density in the forest (Fig. 6.19).

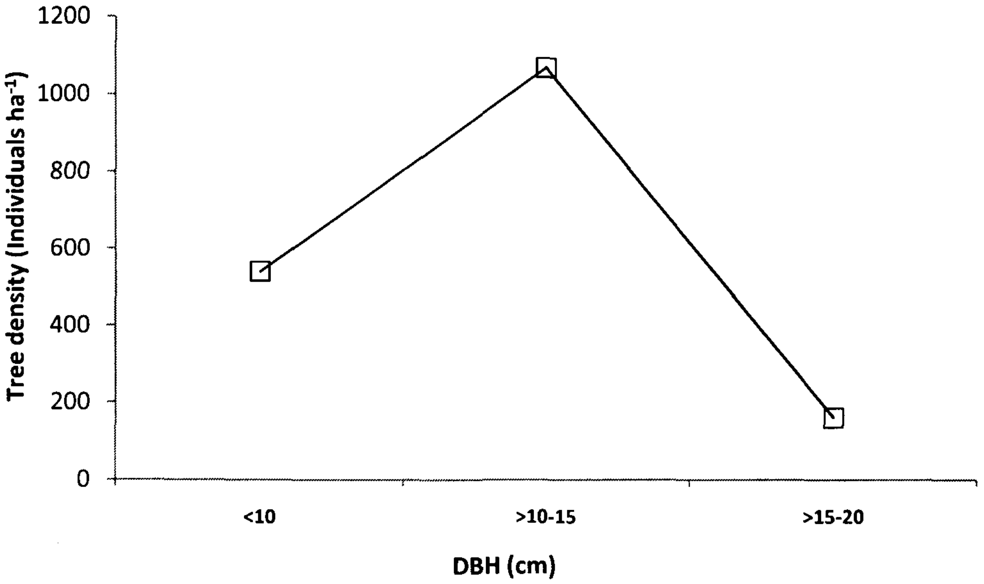


Fig. 6.19 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of regenerating pine forest in montane sub-tropical landscape of Upper Shillong

Aboveground biomass and carbon

In the regenerating pine forest, the tree aboveground biomass was maximum in the >10-15 cm diameter class indicating the young age of the forest. This diameter class contributed 63.9% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon in the forest were 27.3 Mg ha^{-1} and 13.16 Mg C ha^{-1}, respectively (Table 6.32).

Table 6.32 Aboveground biomass and carbon in different diameter classes of regenerating *Pinus kesiya* forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Aboveground biomass and carbon of *Pinus kesiya* (Mg ha^{-1})							
	Stem	**Branch**	**Twig**	**Needle**	**Cone**	**Total**	**Carbon**	**%**
<10	5.1	1.1	0.1	0.5	0.3	7.0	3.4	25.6
>10-15	11.3	2.9	0.2	1.3	0.7	16.3	7.8	59.4
>15-20	1.3	0.3	0.0	0.1	0.1	1.8	0.9	6.6
Total	17.7	4.3	0.3	1.9	1.1	25.3	12.1	91.7
	Aboveground biomass and carbon of other species							
	(Mg ha^{-1})					**(Mg C ha^{-1})**		**%**
<10	0.8					0.4		2.9
>10-15	1.2					0.6		4.3
>15-20	0.3					0.1		1.1
Total	2.3					1.1		100.0

Belowground biomass and carbon

The tree belowground biomass in the regenerating pine forest was maximum in the >10-15 cm diameter class. This diameter class contributed 61.1% to the total belowground biomass in the forest. The total tree belowground biomass and carbon were 4.4 Mg ha^{-1} and 2.1 Mg C ha^{-1}, respectively (Table 6.33).

Table 6.33 Belowground biomass and carbon in different diameter classes of regenerating *Pinus kesiya* forest in montane sub-tropical landscape of Upper Shillong

DBH class	Belowground tree biomass (Mg ha^{-1})				
(cm)	Fine root	Coarse root	Total	Carbon	%
<10	0.2	1.2	1.4	0.7	32.0
>10-15	0.6	2.1	2.7	1.3	61.1
>15-20	0.1	0.2	0.3	0.2	7.0
Total	0.9	3.5	4.4	2.1	100.0

Herb and Shrub biomass and biomass carbon

The herb and shrub biomass in the forest were 119.7 and 199.8 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 56.2 and 95.9 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (31.9 Mg ha^{-1}) in the forest contributing only 0.4 and 0.6%, respectively (Table 6.25).

Regenerating broad-leaved forest

The following models were used to estimate AGB and BGB for regenerating broad-leaved tree species:

$$\text{Log}(Y) = a + b \log D + c (\log D)^2 + d (\log D)^3$$

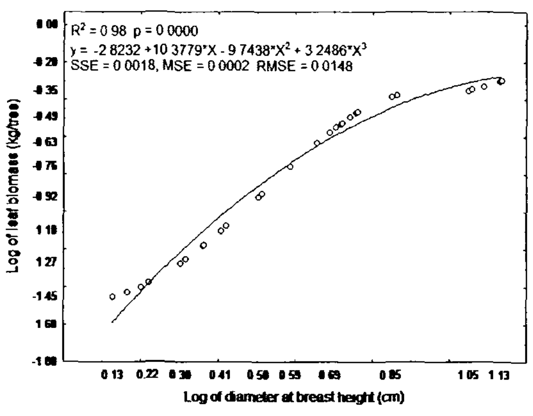
where, Y= AGB (kg/tree), a, b, c, and d are regression coefficients, and D is the tree diameter at breast height. All the measures of coefficients of the models for tree components were statistically significant ($P < 0.001$). Models were evaluated by comparing coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE). The dry weights of various components of tree species were obtained for the development of tree component regression models (Table 6.34) across a range of DBH. The regression coefficients, R^2, SD, SSE, MSE and RMSE for various tree components are presented in Table 6.35. The goodness of fit of the models for various tree components of regenerating broad-leaved forest is presented in Fig. 6.20.

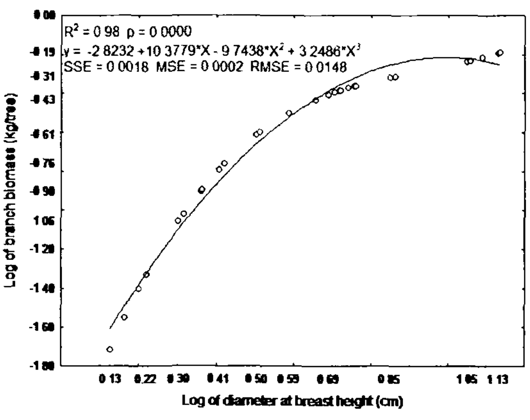
Table 6.34 Dry weight (kg tree^{-1}) of different components of regenerating broad-leaved trees of Upper Shillong used for developing regression models (AGB=aboveground biomass, BGB=belowground biomass)

DBH (cm)	Dry wt. (kg tree^{-1})							
	Stem	Branch	Leaves	Total AGB	Fine root	Coarse root	Total BGB	Total biomas
1.3	0.2	0.1	0.0	0.3	0.1	0.1	0.2	0.4
1.5	0.2	0.1	0.1	0.3	0.1	0.1	0.2	0.5
1.6	0.4	0.0	0.0	0.4	0.1	0.1	0.2	0.7
1.7	0.4	0.0	0.1	0.5	0.1	0.2	0.2	0.7
2.0	0.4	0.1	0.0	0.5	0.1	0.2	0.3	0.8
2.1	0.2	0.2	0.1	0.5	0.1	0.2	0.3	0.7
2.3	0.2	0.2	0.1	0.5	0.1	0.2	0.2	0.7
2.3	0.5	0.1	0.0	0.6	0.1	0.2	0.3	0.9
2.6	0.2	0.2	0.1	0.5	0.1	0.2	0.2	0.7
2.6	0.3	0.2	0.1	0.6	0.1	0.2	0.3	0.9
3.2	1.1	0.2	0.1	1.4	0.2	0.3	0.6	2.0
3.3	0.3	0.3	0.1	0.7	0.1	0.2	0.3	1.0
3.9	0.2	0.4	0.3	0.9	0.2	0.3	0.4	1.4
4.5	0.3	0.4	0.4	1.0	0.2	0.3	0.4	1.4
4.9	0.3	0.5	0.3	1.0	0.2	0.3	0.5	1.5
5.1	1.4	0.3	0.4	2.1	0.3	0.6	0.9	3.0
5.2	1.6	0.4	0.3	2.3	0.4	0.6	1.0	3.3
5.3	0.3	0.4	0.3	0.9	0.2	0.3	0.4	1.3
5.5	0.3	0.5	0.3	1.1	0.2	0.3	0.5	1.6
5.7	1.6	0.5	0.3	2.4	0.3	0.7	1.0	3.4
5.7	1.7	0.4	0.3	2.3	0.3	0.7	1.0	3.3
7.0	1.8	0.4	0.3	2.5	0.4	0.7	1.0	3.5
7.3	1.9	0.5	0.3	2.7	0.4	0.7	1.1	3.8
11.2	2.5	0.6	0.4	3.6	0.4	1.0	1.4	5.0
11.4	2.6	0.7	0.5	3.7	0.4	1.0	1.5	5.1
12.3	2.9	0.7	0.5	4.0	0.5	1.1	1.6	5.6
13.5	3.3	0.6	0.5	4.4	0.5	1.2	1.7	6.1
13.7	3.4	0.6	0.5	4.4	0.6	1.1	1.7	6.1

Table 6.35 Regression coefficients (a, b, c and d), coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) in respect of the models for biomass estimation of individual tree components and total tree biomass in montane sub-tropical regenerating broad-leaved forest of Upper Shillong. The model is of the form: $Log(Y) = a + b\ logD + c\ (logD)^2 + d\ (logD)^3$, where Y= biomass of individual tree components/BGB/AGB/total tree expressed in dry weight (kg $tree^{-1}$) and D = diameter at breast height, AGB = aboveground biomass, BGB = belowground biomass (n=28). The model validity is between 1.3 cm and 13.7 cm DBH

Dependent variables (Y)	Coefficients				R^2	SD	SSE	MSE	RMSE
	a	b	c	d					
Stem	-0.0554	-3.2773	6.2102	-2.5264	0.80	0.448	1.618	0.070	0.265
Branch	-2.3891	6.0453	-6.1073	2.1930	0.90	0.414	0.455	0.020	0.141
Leaf	-1.4459	-0.6030	4.8067	-3.0400	0.92	0.411	0.361	0.016	0.125
Total AGB	-0.3899	-0.5476	2.9576	-1.4815	0.91	0.366	0.316	0.014	0.117
FR	-1.2057	0.2167	1.4561	-0.8295	0.92	0.288	0.224	0.010	0.099
CR	-0.8038	-1.0752	3.5994	-1.7429	0.94	0.345	0.258	0.011	0.106
Total BGB	-0.6831	-0.4797	2.6129	-1.3105	0.91	0.324	0.244	0.011	0.103
Total tree biomass	-0.211	-0.531	2.856	-1.430	0.91	0.353	0.293	0.013	0.113





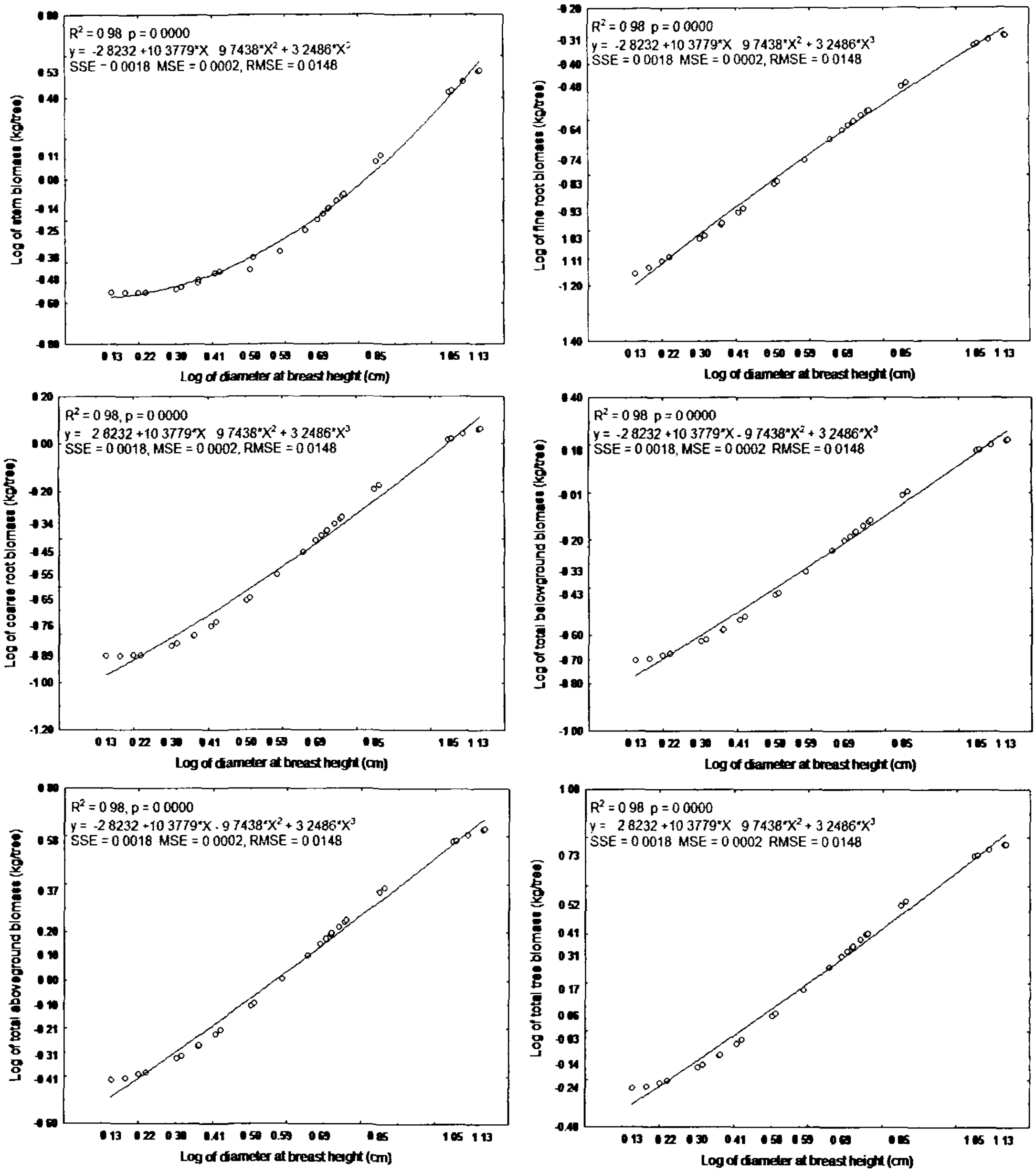


Fig. 6.20 Regression analyses between log of tree diameter at breast height (cm), and biomass of different tree components, total above ground biomass and total tree biomass of regenerating broad-leaved forest

Density diameter distribution

The total tree density in the regenerating broad-leaved forest was 2106 individuals ha^{-1}. The diameter class <10 cm had maximum number of trees (1902 individuals ha^{-1}) contributing 90.3% to the total tree density in the forest (Fig. 6.21).

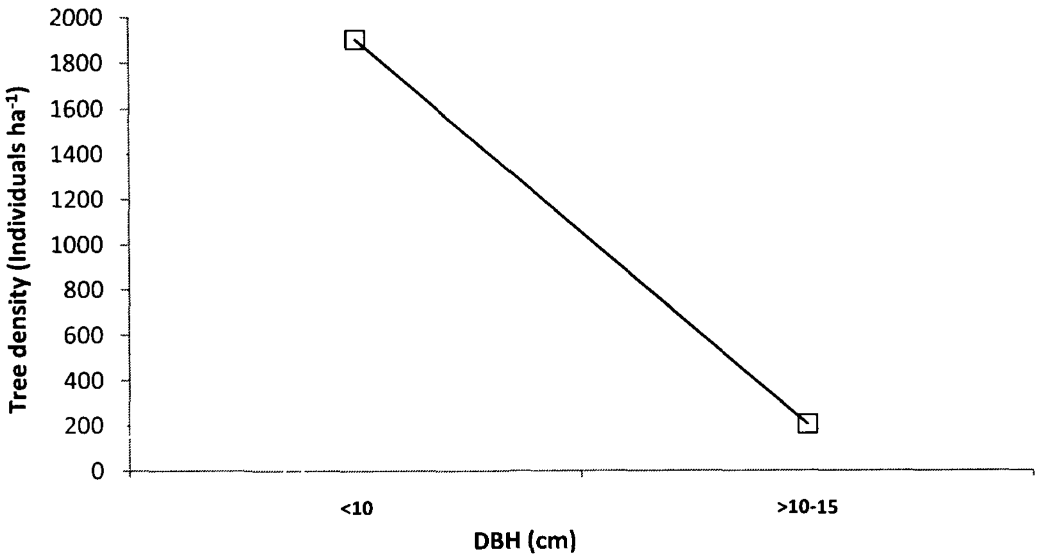


Fig. 6.21 Tree density distribution (Individuals ha^{-1}) in different diameter classes (cm) of regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Aboveground biomass and carbon

The tree aboveground biomass in the regenerating broad-leaved forest was maximum in the >10-15 cm diameter class. This diameter class contributed 62.9% to the total tree aboveground biomass in the forest. The tree aboveground biomass and carbon in the forest were 4.2 Mg ha^{-1} and 2.1 Mg C ha^{-1}, respectively (Table 6.36).

Table 6.36 Aboveground biomass and carbon in different diameter classes of regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Aboveground biomass (Mg ha^{-1})					
	Stem	Branch	Leaf	Total	Carbon	%
<10	0.9	0.3	0.4	1.6	0.8	37.1
>10-15	1.7	0.6	0.4	2.6	1.3	62.9
Total	2.5	0.9	0.8	4.2	2.1	100.0

Belowground biomass and carbon

The tree belowground biomass in the regenerating broad-leaved forest was maximum in <10 cm diameter class and contributed 50.6% to the total belowground biomass in the forest.

The total tree belowground biomass and carbon were 7.9 Mg ha^{-1} and 3.7 Mg C ha^{-1}, respectively (Table 6.37).

Table 6.37 Belowground biomass and carbon in different diameter classes of regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

DBH class (cm)	Belowground tree biomass (Mg ha^{-1})				
	Fine root	Coarse root	Total	Carbon	%
<10	2.3	1.7	4.0	1.9	50.6
>10-15	2.1	1.9	3.9	1.8	49.4
Total	4.4	3.5	7.9	3.7	100.0

Herb and Shrub biomass and carbon

The herb and shrub biomass in the forest were 9.8 and 39.8 kg ha^{-1}, respectively. The corresponding figures for biomass carbon were 4.6 and 19.1 kg C ha^{-1}. The herbs and shrubs contributed a negligible amount to the total aboveground biomass (9.3 Mg ha^{-1}) in the forest contributing only 0.1 and 0.4%, respectively (Table 6.25).

ESTIMATION OF CARBON FLUX IN MONTANE SUB-TROPICAL FOREST ECOSYSTEMS

Soil respiration/CO_2 efflux

The highest soil CO_2 efflux value of 1.8 mg CO_2 kg^{-1} day^{-1} was in regenerating broad-leaved forest during summer season of 2005 and it was lowest in regenerating pine forest (0.2 mg CO_2 kg^{-1} day^{-1}) during winter season of 2008 (Fig. 6.22). The CO_2 efflux was maximum during the summer season and minimum during winter season in all the forest types. The mean CO_2 efflux in the surface layer during three years period i.e. 2005-2008 was highest in the regenerating broad-leaved (1.3 mg CO_2 kg^{-1} day^{-1}) and lowest in the regenerating pine forest (0.5 mg CO_2 kg^{-1} day^{-1}). CO_2 efflux declined significantly ($p<0.01$) with the increase in the soil depth. Three-way ANOVA showed significant variation due to forest types, season and depth (Table 6.38). CO_2 efflux in different forest types followed the order: SRBF>SOBF>SOPF=SRPF.

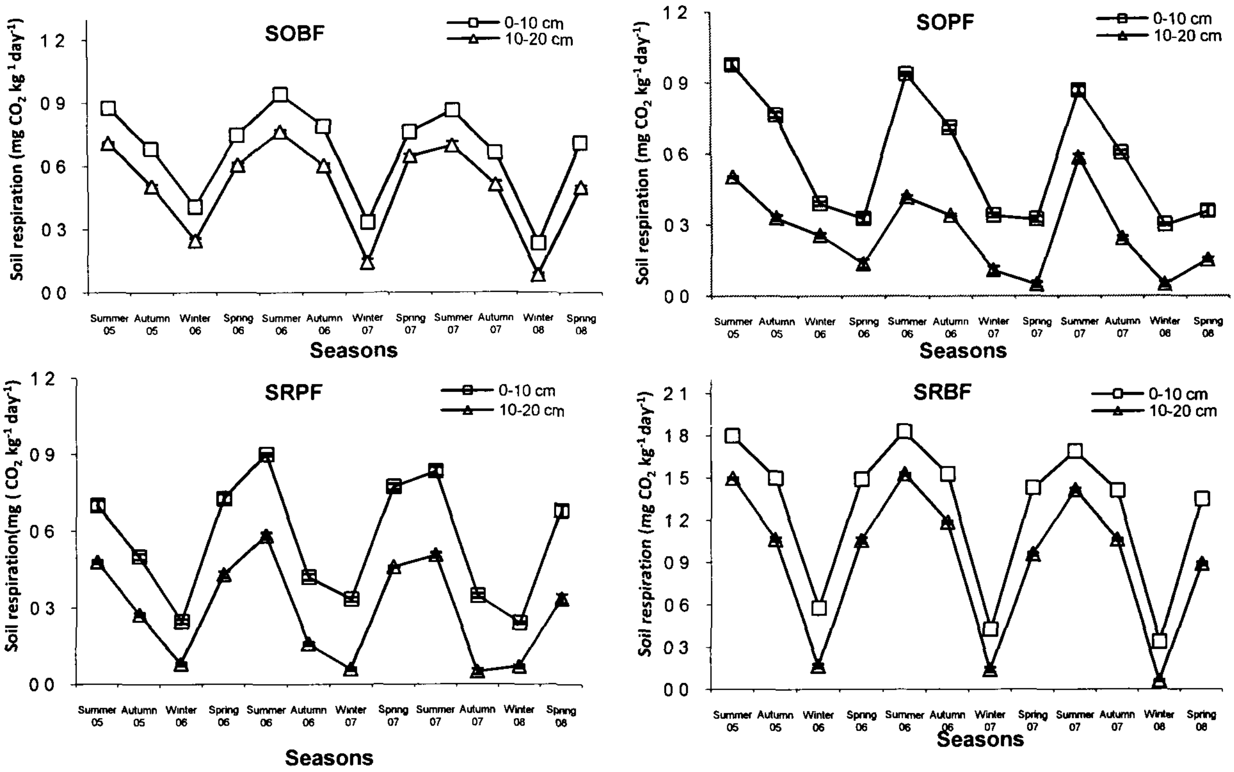


Fig. 6.22 Seasonal variation in the soil respiration (mg CO_2 kg^{-1} day^{-1}) in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers in different forest types in montane sub-tropical landscape of Upper Shillong during 2005-2008. The values are mean (±SE) of five replicate samples

Table 6.38 Three-way ANOVA showing the effect of forest type, season and depth on soil respiration in different forest types in montane sub-tropical landscape of Upper Shillong (*p<0.01, **p<0.01)

Variation due to	df	SS	MS	F	*p*
Forest type	3	21.68	7.23	1392.72**	0.000
Depth	1	3.09	3.09	595.03**	0.000
Seasons	11	10.50	0.95	183.93**	0.000
Forest type*Depth	3	0.15	0.05	9.64*	0.000
Forest type*Seasons	33	5.63	0.17	32.87**	0.000
Depth*Seasons	11	0.08	0.01	1.40	0.218
Forest type*Depth*Seasons	33	0.17	0.01	1.00	0.500

ESTIMATION OF ECOSYSTEM LEVEL BIOMASS AND NPP

Old-growth broad-leaved forest

Total ecosystem biomass of the old-growth broad-leaved forest was 317.8 Mg ha^{-1}, of which 82% was in the aboveground compartment and 18% in the belowground compartment. Trees contributed 78.8%, herbs 0.003%, shrubs 0.01%, and litter 3.2% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 260.7 Mg ha^{-1}. The tree AGB and BGB were 250.5 and 57.1 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts accounted for 69.1, 23.5, 5 and 2.4%, respectively to the total litterfall. The total ecosystem NPP of the forest was 16.6 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed 50.6, 37.9, 11.1 and 0.5% to the total litter production. The total ecosystem carbon content of the forest was 236 Mg C ha^{-1} (Table 6.39). The soil organic carbon was 81.8 Mg C ha^{-1} contributing 34.7% to the total ecosystem carbon. The diameter class >20-30 and >30-40 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 86.3 Mg ha^{-1} and 62.8 Mg ha^{-1}, respectively. These two diameter classes accounted for 34.4% and 25.1 % of the total AGB of the forest (Table 6.23). The aboveground NPP of the forest was 90.3%, while belowground NPP was 9.7%.

Table 6.39 Total ecosystem, above and belowground biomass, carbon content and net primary production of old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Components	Biomass and carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Tree above ground biomass	250.5	122.8	78.8	7.8	47.1
Herbs	0.01	0.005	0.003	0.001	0.01
Shrubs	0.04	0.02	0.013	0.002	0.01
Detrital biomass					
Leaves	7.0	3.2	2.2	3.6	21.8
Twigs	2.4	1.1	0.8	2.7	16.4
Branches	0.5	0.2	0.2	0.8	4.8
Misc.	0.2	0.1	0.1	0.0	0.2
Total detrital biomass	10.1	4.6	3.2	7.2	43.2
Total aboveground biomass	**260.7**	**127.3**	**82.0**	**15.0**	**90.3**
Tree belowground biomass	57.1	26.8	18.0	1.6	9.7
Total forest	317.8	154.1	100.0		
Total soil organic carbon		81.8			
Total Ecosystem	**317.8**	**236.0**	**100.0**	**16.6**	**100.0**
BNPP/NPP				0.1	9.7
ANPP/NPP				0.9	90.3

Old-growth Pinus kesiya forest

Total ecosystem biomass of the old-growth *Pinus kesiya* forest was 466.6 Mg ha^{-1}, of which 91.3% was in the aboveground compartment and 8.7% in the belowground compartment. Trees of *Pinus kesiya* and broad-leaved tree species contributed 76% and 13.3%, respectively to the total forest biomass. The herbs contributed 0.003%, shrubs 0.01%, and litter 3.2% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 208.7 Mg ha^{-1}. The tree AGB and BGB were 425.8 and 40.7 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts accounted for 52, 26.2, 3.2 and 18.6%, respectively to the total litterfall. The total ecosystem NPP of the forest was 21.9 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed 33.5, 26.1, 6.3 and 34.1% to the total litter production. The total ecosystem carbon content of the forest was 236 Mg C ha^{-1} (Table 6.40). The soil organic carbon was 81.8 Mg C ha^{-1} contributing 34.7% to the total ecosystem carbon. The diameter class >30-40, >40-50 and >50-60 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 111.6, 80.7 and 115.9 Mg ha^{-1}, respectively. These diameter classes accounted for 26.8, 19.4 and 27.8% to the total AGB of the forest (Table 6.28). The aboveground NPP of the forest was 93.6%, while belowground NPP was 6.4%.

Table 6.40 Total ecosystem, above and belowground biomass, carbon content and net primary production of old-growth pine forest in montane sub-tropical landscape of Upper Shillong

Components	Biomass and carbon			Net Production	
	(Mg ha^{-1})	(Mg C ha^{-1})	%	(Mg ha^{-1} yr^{-1})	%
Tree aboveground biomass Of *Pinus kesiya*					
Stem	291.8	143.0	62.5	9.8	44.7
Branches	48.2	23.6	10.3	1.5	6.8
Twigs	7.3	3.6	1.6	0.7	3.2
Needles	4.3	2.1	0.9	0.5	2.3
Cones	3.1	1.5	0.7	0.3	1.4
Tree aboveground biomass of other species	62.2	30.5	13.3	3.2	14.6
Herbs	0.12	0.06	0.03	0.001	0.01
Shrubs	0.56	0.27	0.12	0.002	0.01
Detrital biomass					0
Needle	4.4	2.1	0.9	1.5	6.9
Twigs	2.2	1.1	0.5	1.2	5.4
Branches	0.3	0.1	0.1	0.3	1.3
Misc.	1.6	0.8	0.3	1.6	7.1
Total detrital biomass	8.4	4.1	1.8	4.6	20.7
Total aboveground biomass	**425.9**	**208.7**	**91.3**	**20.6**	**93.6**
Tree belowground biomass of *Pinus kesiya*					
Fine roots (<2 mm)	2.4	1.1	0.5	0.3	1.3
Coarse roots (>2mm)	29.4	13.5	6.3	0.7	3.3
Belowground biomass of others species	9.0	4.1	1.9	0.4	1.8
Total forest	466.6	227.4	100.0		
Total soil organic carbon		58.7			
Total Ecosystem	**466.6**	**286.1**	**100.0**	**22.0**	**100.0**
BNPP/NPP				0.1	6.4
ANPP/NPP				0.9	93.6

Regenerating Pinus kesiya forest

Total ecosystem biomass of the regenerating *Pinus kesiya* forest was 36.3 Mg ha^{-1}, of which 87.9% was in the aboveground compartment and 12.1% in the belowground compartment. *Pinus kesiya* and broad-leaved trees species contributed 69.2 and 6.2%, respectively to the total forest biomass. The herbs contributed 0.3%, shrubs 0.6%, and litter 11.6% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 31.9 Mg ha^{-1}. The tree AGB and BGB were 27.4 and 4.4 Mg ha^{-1}, respectively. The leaves, twigs, branches and miscellaneous parts contributed 69.1, 23.5, 5 and 2.4%, respectively to the total litterfall. The total ecosystem NPP of the forest was 4.4 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch and miscellaneous parts contributed 59.2, 18.8, 2.9 and 19.1% to the total litter production. The total ecosystem carbon content of the forest was 52.6 Mg C ha^{-1} (Table 6.41). The soil organic carbon was 35.2 Mg C ha^{-1} contributing 67% to the total ecosystem carbon. The diameter class >10-15 cm had the highest tree biomass among all the diameter classes and contributed maximum biomass of 16.3 Mg ha^{-1}, respectively. This diameter class accounted for 59.4 % of the total AGB of the forest (Table 6.32). The aboveground NPP of the forest was 71.5%, while belowground NPP was 28.5%.

Table 6.41 Total ecosystem, above and belowground biomass, carbon content and net primary production of regenerating pine forest in montane sub-tropical landscape of Upper Shillong

Components	Biomass and carbon			Net Production	
	$(Mg\ ha^{-1})$	$(Mg\ C\ ha^{-1})$	%	$(Mg\ ha^{-1}\ yr^{-1})$	%
Above ground biomass of *Pinus kesiya*					
Stem	17.7	8.5	48.8	1.5	33.9
Branches	4.2	2.0	11.6	0.3	6.8
Twigs	0.2	0.1	0.6	0.1	2.3
Needles	1.9	0.9	5.3	0.2	3.9
Cones	1.0	0.5	2.8	0.0	0.1
Above ground biomass of other species	2.3	1.1	6.2	0.1	1.1
Herbs	0.1	0.06	0.3	0.001	0.02
Shrubs	0.2	0.10	0.6	0.002	0.05
Detrital biomass					
Needles	2.5	1.2	6.9	0.7	15.0
Twigs	0.8	0.4	2.2	0.1	2.4
Branches	0.1	0.1	0.3	0.2	3.4
Misc.	0.8	0.4	2.2	0.1	2.6
Total detrital Biomass	4.2	2.1	11.6	1.0	23.4
Total aboveground biomass	**31.9**	**15.3**	**87.9**	**3.2**	**71.5**
Belowground biomass of *Pinus kesiya*					
Fine roots (<2 mm)	0.9	0.4	2.5	0.3	6.8
Coarse roots (>2mm)	3.5	1.6	9.6	0.9	19.5
Belowground biomass of others species	0.01	0.01	0.03	0.1	2.3
Total forest	36.3	17.4	100.0		
Total soil organic carbon		35.2			
Total Ecosystem	**36.3**	**52.6**	**100.0**	**4.4**	**100.0**
BNPP/NPP				0.3	28.5
ANPP/NPP				0.7	71.5

Regenerating broad-leaved forest

Total ecosystem biomass of the forest was 18.2 Mg ha^{-1}, of which 56.6% was in the aboveground compartment and 43.4% in the belowground compartment. Trees contributed 23.1%, herbs 0.1%, shrubs 0.2%, and litter 33.3% to the total forest biomass. The total AGB of the forest including litter, herb and shrub components was 10.3 Mg ha^{-1}. The leaves, twigs and branches accounted for 89.7, 9.8 and 0.5%, respectively to the total litterfall. The total ecosystem NPP of the forest was 6 Mg ha^{-1} yr^{-1}. The leaf litter, twig and branch contributed 91.7, 6.3 and 2% of the total litter production. The total ecosystem carbon content of the forest was 111 Mg C ha^{-1} (Table 6.42). The soil organic carbon was 102.4 Mg C ha^{-1} contributing a maximum of 92.2% to the total ecosystem carbon. The diameter class <10 cm had the highest tree biomass and contributed 4 Mg ha^{-1} accounting for 50.6% to the total AGB of the forest (Table 6.36). The aboveground NPP of the forest was 66.1%, while belowground NPP was 33.9%.

Table 6.42 Total ecosystem, above and belowground biomass, carbon content and net primary production of regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

Components	Biomass and carbon			Net Production	
	($Mg\ ha^{-1}$)	($Mg\ C\ ha^{-1}$)	%	($Mg\ ha^{-1}\ yr^{-1}$)	%
Tree aboveground biomass					
Stem	2.5	1.2	13.8	0.2	3.5
Branches	0.9	0.5	5.0	0.1	1.7
Leaves	0.8	0.4	4.2	0.1	1.3
Herbs	0.01	0.005	0.05	0.002	0.03
Shrubs	0.04	0.02	0.22	0.004	0.07
Detrital biomass					
Leaves	5.4	2.6	29.9	3.3	54.5
Twigs	0.6	0.3	3.3	0.2	3.7
Branches	0.03	0.01	0.2	0.1	1.2
Misc.	0.001	0.0005	0.010	0.001	0.02
Total detrital biomass	6.1	2.9	33.3	3.5	59.4
Total aboveground biomass	**10.3**	**4.9**	**56.6**	**3.9**	**66.1**
Tree belowground biomass					
Fine root	4.4	2.1	24.0	1.5	24.3
Coarse root	3.5	1.7	19.4	0.6	9.6
Total forest	18.2	8.7	100.0		
Total soil organic carbon		102.4			
Total Ecosystem	**18.2**	**111.0**	**100.0**	**6.0**	**100.0**
BNPP/NPP				0.3	33.9
ANPP/NPP				0.7	66.1

DISCUSSION

The structure and function of forest ecosystem is mostly determined by the plant (Richards 1996). The important characteristics of the tropical and subtropical humid forests are their species richness and complex community organization. These forests, found all over tropical and sub-tropical region, harbour maximum diversity of plant species found on the earth (WCMC 1992). Species diversity is an important attribute of a natural community that influences functioning of an ecosystem (Hengeveld 1996). High species content per unit area is largely due to presence of synuisae in the forest (Richards 1996). Forest vegetation and soils constitute a major terrestrial carbon pool with the potential to absorb and store carbon dioxide (CO_2) from the atmosphere. The CO_2 sink and source of forest ecosystem go hand in hand with the dynamics of trees grow, death and decay and are subjected to disturbance and forest management practices.

Vegetation characteristics of the montane sub-tropical forest ecosystem

The species richness was high in the regenerating broad-leaved forest (35 species), followed by old-growth broad-leaved forest (21 species), old-growth pine forest (9 species) and regenerating pine forest (6 species). The tree density in the montane sub-tropical landscape was highest in the regenerating broad-leaved forest (2106 individuals ha^{-1}), followed by regenerating pine forest (1768 individuals ha^{-1}), old-growth broad-leaved forest (980 individuals ha^{-1}) and old-growth pine forest (628 individuals ha^{-1}).

Lindera latifolia (IVI 60.5) and *Rhododendron arboretum* (IVI 37.6) were the dominant tree species in old-growth broad-leaved and regenerating broad-leaved forests. *Pinus kesiya* was the dominant tree species in the old-growth pine and regenerating pine forests. The basal area of the forest was high in the old-growth broad-leaved forest, followed by old-growth broad-leaved forest, regenerating pine forest and lowest in regenerating broad-leaved forest. The relatively high total basal area and low density are characteristic features of old growth forest (Golley *et al.* 1965) which is evident in the present study. The Shannon diversity, Simpson dominance and Species evenness indices in the montane sub-tropical landscape were high in the regenerating broad-leaved forest and followed the order:

SRBF>SOBF>SOPF>SRPF. Alpha diversity was highest in the regenerating broad-leaved forest and followed the order: SRBF>SOBF>SOPF>SRPF.

Soil properties as a function of forest ecosystem carbon

Physico-chemical properties of forest soils vary in time and space due to variation in topography, climate, weathering processes, vegetation cover, microbial activities and several other biotic and abiotic factors (Paudel and Sah, 2003). Vegetation also plays an important role in soil formation (Champan and Reiss, 1992).

The soil was sandy in all the montane sub-tropical forest types of Upper Shillong. The clay content was higher in the sub-surface soil layer whereas water holding capacity (WHC) was higher in the surface soil layer, which had greater accumulation of organic matter thereby indicating a stronger influence of SOC on WHC than the clay particles. Higher WHC has been reported from old forest regrowth in north-eastern India where SOM was as high as 11% (Arunachalam *et al.* 1996a; Maithani 1996). The clay percentage ranged between 7.2 – 10.3% in the surface layer and 8.2 – 11.4% in the sub-surface layer. The silt percentage varied between 0.4 – 1.5% and 0.9 – 2.1% respectively, in the corresponding layers. The corresponding figures for sand ranged between 88.2 – 92.5 and 86.6 – 89.7%, respectively. These values were higher than the reported values of 3 - 31%, 21 - 41% and 36 - 66% for clay, silt and sand percentage (Sakin *et al.* 2011). In all the forest types the amounts of clay and silt increased along depth. The less clay and silt values in the montane sub-tropical landscape were due to leaching and run off of the fine particles due to heavy rainfall.

Soil bulk density (BD) in the surface and subsurface layer was highest in the old-growth broad-leaved forest and lowest in the old-growth pine forest. The bulk density increased with depth in all the forest types and followed the order: SOBF>SRBF>SRPF>SOPF. The increase in BD in all the forest types may be attributed to increase in sand percentage in the soil (Guerrero *et al.* 2000). A strong negative correlation was observed between BD and SOC in the montane sub-tropical landscape (R^2 = 0.96; p = 0.003) (Fig. 6.23). The SOC declined significantly with increase in BD ($p<0.01$). A strong negative correlation was also observed between BD and SOC (R^2 = -0.78, $p<0.01$) by Li *et al.* (2007). Curtis and Post (1964) also observed a reverse correlation between organic matter and bulk density.

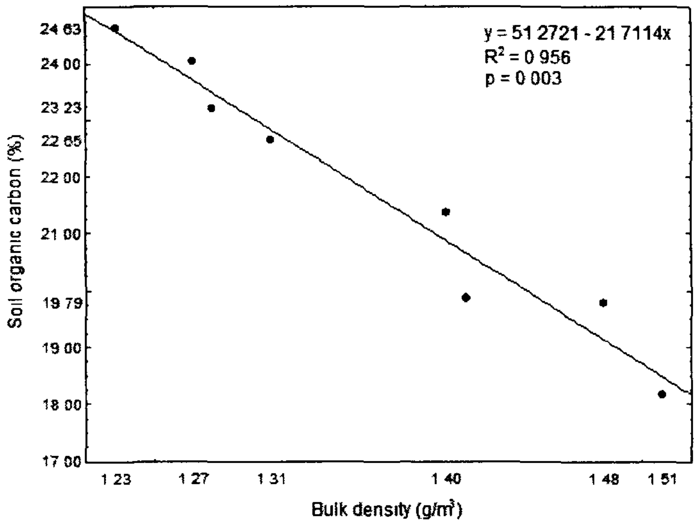


Fig. 6.23 Relationship between BD and SOC in the montane sub-tropical landscape of Upper Shillong

The surface soil moisture content (SMC) exhibits a high degree of spatial and temporal variability. Soil moisture variability is influenced by a number of factors. These include variations in topography, soil properties, vegetation type and density, mean moisture content, depth to water table, precipitation depth, solar radiation and other meteorological factors (Famiglietti *et al.* 1995). The SMC in the montane sub-tropical forest types was higher in the surface soil layer than the sub-surface soil layer. The greater moisture content in the surface soil layer may be ascribed to the greater accumulation of litter on the forest floor that check evaporation losses and higher SOM that helps in retention of moisture.

The soils of the montane sub-tropical landscape was acidic (pH = 4.2 – 5.9) in nature. John *et al.* (2002) reported pH of 5.0 – 5.4 in the sub-tropical pine forest of Meghalaya. Singh *et al.* (1995) recorded higher soil pH in natural forests of north-eastern India. This type of soil as stated by Rathore (1971) is usually rich in organic matter and nitrogen. These two soil nutrients were also highly related to each other as shown by Shrestha (1979). A drop in the pH during the rainy season could be due to the result of excessive leaching of basic cations through heavy rainfall (Wild 1996).

The higher TKN concentration in the surface and sub-surface layer could be due to the higher organic matter concentration in this layer. A sharp decline in TKN value from 0.5% in old-growth broad-leaved forest to 0.3% in old-growth pine forest and regenerating pine forest may be due to runoff losses caused by heavy rainfall. The higher concentration of TKN during winter season for all the forest types were in accordance with the findings of Kamei (2007). A strong positive correlation was observed between SOC and TKN in the montane sub-tropical landscape ($R^2 = 0.834$; $p < 0.01$) (Fig. 6.24).

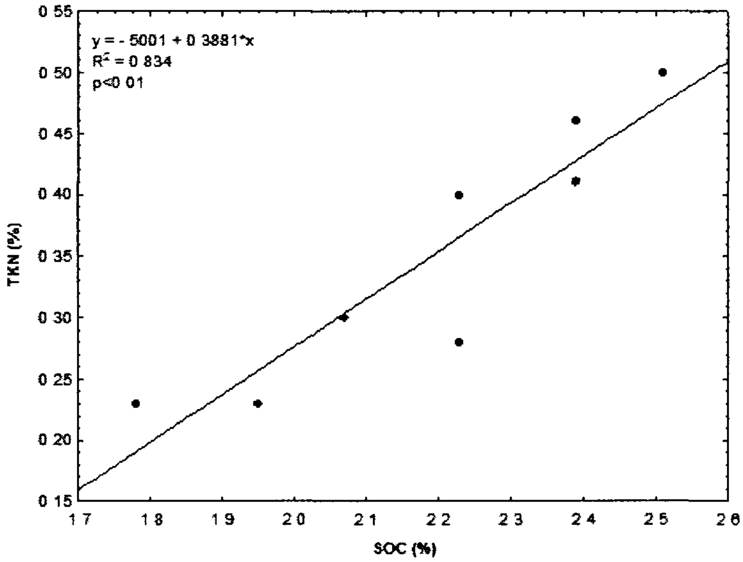


Fig. 6.24 Relationship between SOC and TKN in the montane sub-tropical landscape of Upper Shillong

Available phosphorus is present at very low concentration in the soils of north-eastern region of the country. Some earlier workers (Arunachalam *et al.* 1998a; Saikh *et al.* 1998; Singh 2002; Ralte 2004) observed significant seasonal variation in available P content in different land-use systems. The present findings also showed significant ($p< 0.01$) variation due to forest type, depth and season. Greater input of phosphorus through litter during winter and spring season has been reported by Arunachalam *et al.* (1998a) in the humid montane sub-tropical region of India. The SOC did not show positive correlation with available P in the montane sub-tropical landscape. Therefore, no significant role of organic matter in P availability was observed.

Exchangeable Potassium (K) promotes photosynthesis, controls stomata opening, improves the utilization of N, promotes the transport of assimilates and consequently increases crop yields (Yawson *et al.* 2011). High exchangeable K (239.5 $\mu g\ g^{-1}$) was observed during the winter season in old-growth broad-leaved forest and low in the old-growth pine forest (74 $\mu g\ g^{-1}$) during summer season. Potassium concentration declined further in the next rainy season in all forest types due to excessive leaching and runoff losses. Fluctuation in the concentration of K is because K cycles through vegetation and soil and solely acts as unbound ion, and is easily leached from living and decomposing plant tissues compared to other nutrients (Bradley *et al.* 2001). The exchangeable K concentration in the surface soil layer of old-growth broad-leaved forest and regenerating broad-leaved forest of montane sub-tropical landscape was higher because K was retained and cycled dynamically in the forest ecosystem. Exchangeable K showed a positive correlation with SOC ($R^2 = 0.751$; $p < 0.01$) (Fig. 6.25).

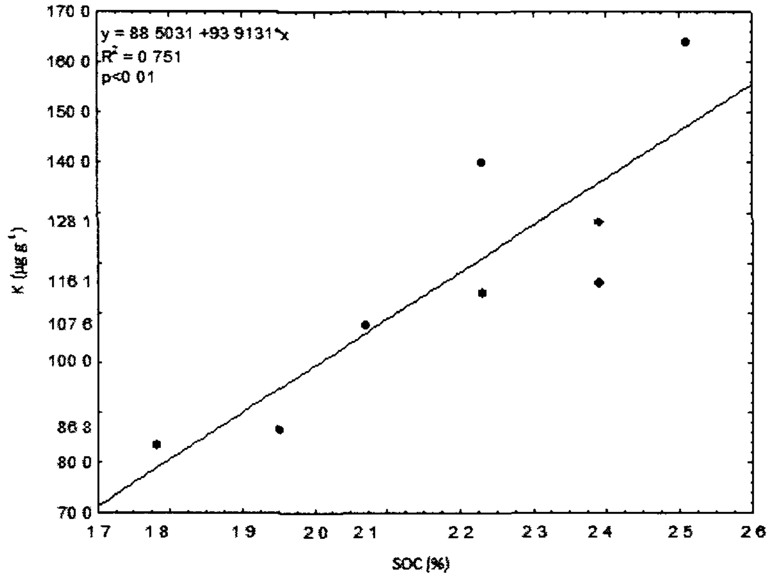


Fig. 6.25 Relationship between SOC and K in montane sub-tropical landscape of Upper Shillong

Soils play a central role in the dynamics of carbon in the biosphere because they store the largest carbon stock in terrestrial ecosystems. Soil C sequestration is considered an important ecosystem service and the enhancement of above and a belowground C stock has become a recognized forest management strategy (Neumann-Cosel *et al.* 2011) Soil carbon pools govern soil properties and nutrient cycling in forest ecosystems and play a critical role on forest productivity. Soil C pool forms a large and dynamic reservoir of C, which is an important part of the global C cycle and a potential sink for atmospheric CO_2. Soil C pool is important for nutrient cycling in forest ecosystems and global C balance and has been of interest in understanding the effect of forest soil management on soil C pool (Ussiri and Johnson, 2007).

The soil organic carbon pool in the regenerating broad-leaved forest was 102.4 Mg C ha^{-1}, followed by old-growth broad-leaved forest (81.8 Mg C ha^{-1}), old-growth pine forest (58.7 Mg C ha^{-1}) regenerating pine forest (35.2 Mg C ha^{-1}). The values for regenerating broad-leaved forest (102.4 Mg C ha^{-1}) and the old-growth broad-leaved forest (81.8 Mg C ha^{-1}), are comparable with the findings (72 – 149 Mg C ha^{-1}) of Glaser *et al.* (2003) for the Amazonian rain forest near Belterra and of Lu *et al.* (2010) in the tropical seasonal forest of China (84 to 102 Mg C ha^{-1}). The magnitude of soil organic carbon pool and finally carbon mineralization depend on many factors. The impact of land-use change on organic carbon pool in the mineral soil depends on long-term site-specific factors (e.g. climate, topography and parent material) and is often overridden by the high spatial heterogeneity of soil organic carbon (Brown and Lugo, 1982; Smithson *et al.* 2002; Schwendenmann *et al.* 2007). The low organic carbon in both the pine forest was due to the acidity of soil, low soil and air temperature thereby slowing the decomposition of organic matter as is evident from the litter decomposition study. Beside high content of lignin in the pine needles (Arunachalam *et al.* 1996b) adds to the slow decomposition process in the both the pine forests. Jha *et al.* (2003) reported that the SOC pool in *Pinus kesiya* to be 116.4 Mg ha^{-1}, which is higher than the present findings.

The values are higher than the reported values (34 – 56 Mg C ha^{-1}) by Sombroek *et al.* (1993) for Amazonian forests, but are comparable with the SOC values of old-growth pine forest (58.7 Mg C ha^{-1}) and regenerating pine forest (35.2 Mg C ha^{-1}). The SOC values for old-growth pine forest are comparable with the reported values for *Pinus ponderosa* forests (86.3 Mg C ha^{-1}) by Laclau (2003). The present SOC values are less than the reported values for the tropical forests of Asia (139 Mg C ha^{-1}) by Dixon *et al.* (1994) and the Amazonian rain forests near Manaus (147 - 506 Mg C ha^{-1}) by Glaser *et al.* (2003).

The soil organic matter upto 1 m depth in the regenerating broad-leaved forest was highest (176.5 Mg ha^{-1}), followed by old-growth broad-leaved forest (141.0 Mg ha^{-1}), old-growth pine forest (101.2 Mg ha^{-1}) and lowest in the regenerating pine forest (60.7 Mg ha^{-1}). The soil organic matter is mainly responsible for ecosystem functioning and determines whether soil act as sink or source of carbon in the global carbon cycle (Brown and Lugo, 1982; Smithson *et al.* 2002).

The C/N ratio is a good indicator of the degree of decomposition and quality of the organic matter in soil. It reflects the site quality and forest species composition owing to its variability for litter among species (Yamakura and Sahunalu, 1990). The soil C/N ratio varied among the forest types and increased with depth in all the forest types (Table 6.16). The higher C/N ratios in deeper soil layers are mainly due to the decreasing organic C concentration along the soil profile. The C/N ratio of the surface (0-10 cm) and sub-surface (10-20 cm) soil layers of montane sub-tropical landscape ranged between 4.8 – 9.0 and 4.9 – 10.6, respectively. These values are within the reported C/N range (3.3 – 14.3) for montane sub-tropical forests of the world (Post *et al.* 1985). The higher decomposition rate in the old-growth broad-leaved forest (k = 0.3) and the regenerating broad-leaved forest (0.8) than the two pine forests was because of higher C/N ratio.

Seasonal and spatial dynamics of microbial biomass carbon (MBC)

The concentration of MBC obtained in the present study is within the reported range (61-2000 μg g^{-1}) for various temperate and tropical forest soils (Henrot and Robertson, 1994; Diaz-Ravina *et al.* 1995; Vance *et al.* 1987). The low value during winter season is in accordance with the findings of Lynch and Panting (1982) and Sarathchandra *et al.* (1989). Higher values of microbial biomass during summer (rainy) season have also been reported from pasture and dry tropical forest soils (Sarathchandra *et al.* 1984; Singh *et al.* 1989). In this study, minima and maxima of SMC and MBC were observed during winter and rainy seasons, respectively. These results are in conformity with those of Singh (2002) who reported higher MBC during warm-wet months and lower during dry-cold period, in the undegraded forest of Arunachal Pradesh. The MBC showed a strong positive correlation with SMC (R^2=0.68, p<0.005) (Fig. 6.26).

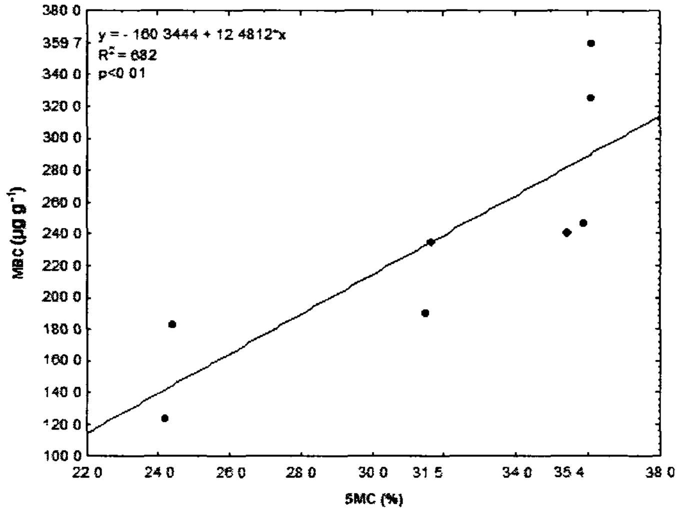


Fig. 6.26 Relationship between SMC and MBC in montane sub-tropical landscape of Upper Shillong

Contribution of microbial biomass carbon to SOC

The percentage contribution of MBC to SOC in the tropical landscape indicated the evolution of a more complex system of substrate-use in these forest types by microbial communities. In the present study, the contribution of MBC to soil organic carbon in the surface soil layer during summer season (1.38 - 1.97%) in montane sub-tropical landscape was within the reported range (1.8 - 2.9%) for sub-tropical forests (Vance *et al.* 1987). The results are consistent with previously reported studies (Arunachalam and Arunachalam, 2000; Sharma *et al.* 2004; Wright *et al.* 2005). The SOC showed a strong positive correlation with MBC (R^2=0.91, p<0.001) (Fig. 6.27) in the montane sub-tropical landscape.

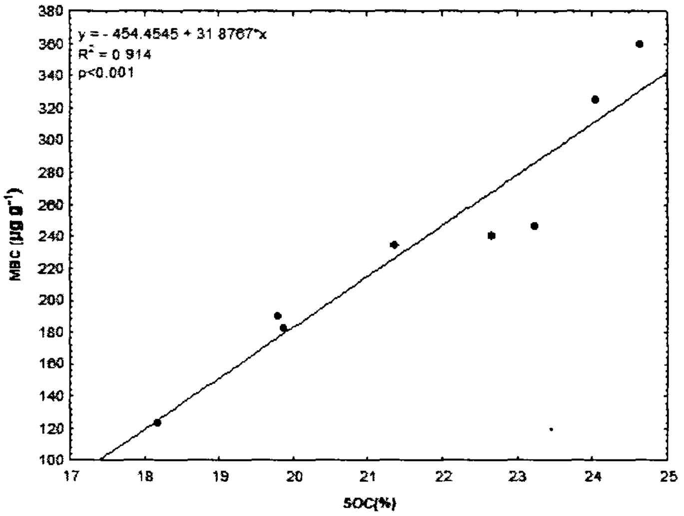


Fig. 6.27 Relationship between SOC and MBC in montane sub-tropical landscape of Upper Shillong

Litter production and accumulation in the montane sub-tropical landscape

The total annual litterfall obtained in the different forest types (4.2 – 10.1 Mg ha^{-1}) was within the reported range for various montane sub-tropical forests (2.2 - 22.6 Mg ha^{-1}) (Vogt *et al.* 1986), 2.4 - 10.3 Mg ha^{-1} for the Amazonian tropical forest (Cuveas and Medina, 1986), 9.5 – 12.8 Mg ha^{-1} for the montane sub-tropical forest of Meghalaya (Kamei 2007). The values were however lower (13 – 15 Mg ha^{-1}) than the moist deciduous forest of Indian Western Ghats (Swamy and Procter, 1994). The litterfall values obtained in the montane sub-tropical forest types were within the reported range of 6.8 to 10.0 Mg ha^{-1} for the humid tropical forest of West Africa (Cornforth 1970) and three Colombian forests (Folster and Salas, 1976). The highest litterfall was obtained in the old-growth broad-leaved forest (10.1 Mg ha^{-1}) followed by old-growth pine forest (8.4 Mg ha^{-1}), regenerating broad-leaved forest (6.1 Mg ha^{-1}) and lowest in regenerating pine forest (4.2 Mg ha^{-1}). The high litter production in the old-growth broad-leaved forest may be because of high tree density and leaf dry weight. The annual litterfall in the old-growth *P. kesiya* forest (8.4 Mg ha^{-1}) was greater than that in the 22 years old forest (1.5 Mg ha^{-1}) reported by Arunachalam *et al.* (1996b). The proportion of leaf litter in the total litter was 52% and the twigs, branches and reproductive parts accounted for 26.2%, 3.2% and 18.6%, respectively. Arunachalam *et al.* (1996b) reported that leaf litter constituted as high as 75% of the total litter in the young 22 years old *P. kesiya* forest stand. Therefore, it is evident that with increasing age of *P. kesiya* forest, the relative proportion of leaf litter in the litter decreases (Baishya and Barik, 2011).

The proportion of leaf (52 - 89%) in the litterfall of montane sub-tropical forest types was in accordance with the reports of Lian and Zhang (1998) for the sub-tropical forest types of China (50 - 80%), of Arunachalam *et al.* (1998a) (78 - 88%), and of Kamei (2007) (71 - 81%) for the montane sub-tropical forest of Meghalaya. The peak litterfall as observed in winter and spring is similar to the observation of earlier studies in evergreen broad-leaved forest (Arunachalam *et al.* 1998a; Kamei 2007; Khiewtam and Ramakrishnan, 1993; Yang *et al.* 2004).

The litter accumulation value of the different forest types in montane sub-tropical landscape (3,867 – 6,015kg ha^{-1}) is close to the values reported by Kamei (2007) (4,334-6,147 kg ha^{-1}). The greater litter accumulation in the montane sub-tropical forest types was due to slower decomposition rate ($k = 0.31 – 1.25$) and turnover rate (0.35 – 0.58 yr^{-1}) (Table 5.19). The values are close to the reported figures by Kamei (2007), where decomposition rate was 0.53 yr^{-1} and turnover rate was 2.04 yr^{-1}. The high litter accumulation in old-growth broad-leaved forest (6,015 kg ha^{-1}) could be attributed to these reasons.

Litter decomposition in the montane sub-tropical landscape

The climatic conditions and initial chemical composition of litter influences decomposition rate (Couteaux *et al.* 1995). Rainfall and air temperature determine the rate of decomposition in unfavourable weather conditions (Arunachalam *et al.* 1998b). The seasonal pattern of litterfall is related to the change in climatic variables and intrinsic generic characters of species (Jamaludheen and Kumar, 1999).

Litter decomposition in all the forest types of montane sub-tropical landscape did not show any distinct phase of decomposition, except in the old-growth broad-leaved forest and the regenerating broad-leaved forests where two distinct phases were observed. The initial phase lasted for about 3 months (June – August) during which 26 - 55% of the litter decomposed due to favourable environmental conditions. The second phase had undergone slow weight loss lasting for 9 months where, 11 - 31% of the litter remained undecomposed. The old-growth broad-leaved forest showed a faster rate of litter decomposition than both the pine forests, where it showed a very slow rate of litter decomposition. The weight loss pattern followed the order: SOBF>SRBF>SRPF>SOPF. The decomposition rate in the old-growth broad-leaved forest was higher than the other forest types due to higher N content, C/N ratio, soil moisture, temperature and microbial biomass. Increased litter thickness and litter accumulation on the forest floor at high altitude may be due to low rate of litter decomposition followed by low microbial activity as compared to the low altitude. The nutrients released from litter through decomposition process is recognized as an important process of nutrient cycle whereby essential elements are made available for plant growth (MacLean and Wein, 1978).

Carbon flux in the montane sub-tropical forest ecosystem

Soil respiration from terrestrial biosphere represents the second largest global flux of CO_2 to the atmosphere after the ocean (Schlesinger and Andrew, 2000). Soil CO_2 efflux is mainly regulated by the oxidation of soil organic matter during litter decomposition by heterotrophic microorganisms and respiration by plant roots (Pandey *et al.* 2010). The soil microorganisms and soil abiotic factors (*viz.*, temperature, moisture, soil organic matter) are the major factor responsible for emission of CO_2 from the soil (Muhr *et al.* 2008). Forest management practice like thinning is reported to influence soil CO_2 efflux as a result of changes in micro-environmental conditions, mainly soil microclimate and root dynamics (Tian *et al.* 2009). Soil CO_2 efflux value was highest in the regenerating broad-leaved forest (1.3 mg CO_2 kg^{-1} day^{-1}), followed by old-growth broad-leaved forest (0.7 mg CO_2 kg^{-1} day^{-1}), old-growth pine forest (0.6 mg CO_2 kg^{-1} day^{-1}) and regenerating pine forest (0.5 mg CO_2 kg^{-1} day^{-1}). These values are higher than the values (0.02 - 0.1 mg CO_2 $kg^{-1}day^{-1}$) reported by Aghasi *et al.* (2011). Favourable soil conditions during summer season promote maximum microbial activity in the soil. The decrease in soil CO_2 efflux values during winter season is in accordance with the report of Mo *et al.* (2005). The soil respiration varied with the forest type, depth and season, which could be attributed to the differences in temperature, soil moisture content, root biomass and soil microbial biomass (Yi *et al.* 2007). Mallik and Hu (1997) reported that soil organic matter influences soil microbial respiration and is one of the important factors controlling it. The different land-use affects the formation of organic matter, SOC and microbial biomass C, which in turn affects soil microbial respiration.

Soil CO_2 efflux showed a positive correlation with microbial biomass carbon in the montane sub-tropical landscape ($R^2 = 0.81$; $p<0.01$) (Fig. 6.28). The findings are in support of positive correlation found between CO_2 efflux and microbial biomass in the forests of Puerto Rico by Li *et al.* (2005).

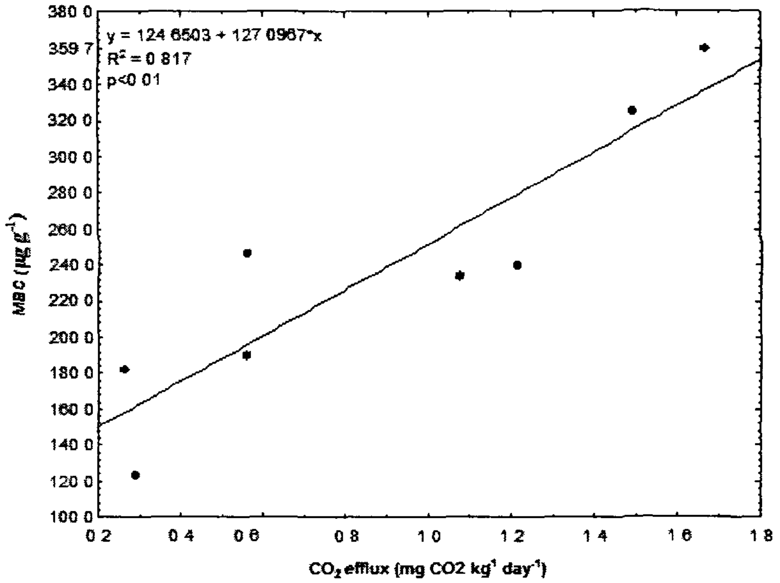


Fig. 6.28 Relationship between CO_2 efflux and microbial biomass carbon in the montane sub-tropical landscape of Upper Shillong

Biomass and carbon sequestration potential in the montane sub-tropical forest ecosystems

In the old-growth broad-leaved forest, the observed AGB value of 260.7 Mg ha^{-1} is within the reported values for the tropical rain forests of Malaysia (225 - 446 Mg ha^{-1}) and Cameroon (238 - 341 Mg ha^{-1}). The values are also within the range (153 - 221 Mg ha^{-1}) reported from Sri Lankan tropical rain forests by Brown and Lugo (1982), 96 - 276 Mg ha^{-1} for the tropical rain forests of Thailand by Terakunpisut *et al.* (2007). The AGB value obtained in the present study coincides with the AGB for tropical Africa (260 Mg ha^{-1}), but more than 215 Mg ha^{-1} for tropical Asia and 150 Mg ha^{-1} for total tropics as reported by Brown and Lugo (1984). The present AGB values for the old-growth broad-leaved forest is also less than the reported range for the primary rainforests of Southeast Asia, ranging from 300 - 500 Mg ha^{-1} (Kira and Shidei, 1967; Kato *et al.* 1978; Laumonier *et al.* 2010; Yamakura *et al.* 1986). The AGB of the old-growth broad-leaved forest are comparable with the AGB (215 – 468.2 Mg ha^{-1}) of the moist temperate forest of Garhwal Himalaya, Uttarakhand (Gairola *et al.* 2011).

The AGB in the old-growth *Pinus kesiya* forest (425.9 Mg ha^{-1}) was about 38% greater than that of the 22-years old young forest studied by Das and Ramakrishnan (1987) (308.7 Mg ha^{-1}). Increase in AGB with age was also depicted through the studies conducted by Ovington and Madgwick (1959), and Delrio *et al.* (2008) for *Pinus sylvestris* L. for young and old-growth forests, respectively. Similar increase was also noted from the studies of Karizumi (1974) and Tanabe *et al.* (2003) for *Pinus densiflora* forest of Japan. The AGB obtained in this study is comparable with 41 - 60 years old *P. sylvestris* forest in Spain (359.7 – 529.3 Mg ha^{-1}) (Delrio *et al.* 2008). However, it was much greater than that of 71-80 years old *Pinus koraiensis* forest of Japan (317.9 Mg ha^{-1}) (Son *et al.* 2001). The AGB of *P. kesiya* was greater than the *Pinus roxburghii* forest (115 - 236 Mg ha^{-1}) of central Himalaya, India (Chaturvedi and Singh, 1987). The AGB obtained in the present study was also greater than most tropical forests studied. Muller (1982) obtained an AGB of 330 Mg ha^{-1} for the tropical broad-leaved forests of eastern hardwood region of USA, and Brown *et al.* (1989) reported 238-341 Mg ha^{-1} for Cameroon and 153 - 221 Mg ha^{-1} for Sri Lanka. However, the AGB of the present study is comparable with the findings of Brown *et al.* (1989) for the tropical rain forests of Malaysia (225 - 446 Mg ha^{-1}). The ecosystem level biomass of the *Pinus kesiya* forest was 466.6 Mg ha^{-1}. The AGB in the regenerating pine forests 31.9 Mg ha^{-1} is comparable with the 18 years old *Pinus densiflora* forest of Japan (26.7 Mg ha^{-1}) reported by Karizumi (1974). The AGB of 10.3 Mg ha^{-1} obtained in the regenerating broad-leaved forest was lower than the 20 years old (118 Mg ha^{-1}) and for the 30 years (177 Mg ha^{-1}) old tropical secondary forest of Ecuador (Fehse *et al.* 2002). Although aboveground biomass of regenerating broad-leaved forest (12 - 15 years) in the study is in close range with the 15 – 20 years old tropical secondary forests (14 – 272 $\mathrm{Mgha^{-1}}$) (Marin-Spiotta *et al.* 2008). The montane sub-tropical regenerating broad-leaved forest being of younger age, the AGB was less than the other reported values for secondary forests of the world.

The AGB carbon stored in the old-growth broad-leaved forest (127.3 Mg C $\mathrm{ha^{-1}}$) is in close range with the findings of Flint and Richards (1996) for Southeast Asia countries including India, Thailand, Cambodia, Malaysia and Indonesia, who reported 17 - 350 Mg C $\mathrm{ha^{-1}}$ in the matured tropical rain forests. Ogawa *et al.* (1965) reported 60 - 179 Mg C $\mathrm{ha^{-1}}$ in different tropical forest types of Thailand. The biomass carbon values are comparable with the findings (108 – 234.1 Mg C $\mathrm{ha^{-1}}$) of Gairola *et al.* (2011) for the moist temperate forest of Garhwal Himalaya, Uttarakhand, India. The values were however, greater than the tropical forests of Sri Lanka (77 Mg C $\mathrm{ha^{-1}}$) and for the primary Amazon forest (110 Mg C $\mathrm{ha^{-1}}$) (Phillips *et al.* 1998), but lower than the relatively undisturbed matured tropical rain forest of Malaysia (223 Mg C $\mathrm{ha^{-1}}$) as reported by Brown and Lugo (1982). The tropical old-growth forest of Ivory Coast had also AGBC of 151.5 - 256.5 Mg C $\mathrm{ha^{-1}}$ as reported by Clark *et al.* (2001b) and the Amazonian forest (123.5 Mg C $\mathrm{ha^{-1}}$) by Girardin *et al.* (2010).

The AGB carbon stock in the old-growth pine forest 208.7 Mg C $\mathrm{ha^{-1}}$ was within the reported range (169 – 254 Mg C $\mathrm{ha^{-1}}$) for *Pinus sylvestris* forests of Russia (Kurbanov 2000). The AGB carbon in the regenerating pine forest (15.3 Mg C $\mathrm{ha^{-1}}$) and the regenerating broad-leaved forest (4.9 Mg C $\mathrm{ha^{-1}}$) had much lower biomass carbon owing to their young age.

The ecosystem level carbon stock in the montane sub-tropical landscape were high in the old-growth pine forest (286.1 Mg C $\mathrm{ha^{-1}}$), followed by old-growth broad-leaved forest (236.0

Mg C ha^{-1}), regenerating broad-leaved forest (111.0 Mg C ha^{-1}) and regenerating pine forest (52.6 Mg C ha^{-1}).

The higher diameter class i.e. >60 cm DBH contributed 10.5% and 15.8%, respectively to the total aboveground biomass in the old-growth broad-leaved forest and old-growth pine forest. The regenerating pine and the broad-leaved forests had trees only in the lower diameter class <20 cm. Brown *et al.* (1995) and other co-workers reported that the higher diameter class (>70 cm) can contribute upto 50% to the total AGB, which in this case was very low. These results show that the higher diameter classes can add substantial amount of carbon in the future, provided these forests are protected from anthropogenic disturbances. Analyses have shown that forests with reduced biomass have either had their large trees removed by past human disturbance or represent regenerating secondary forests which do not yet have large trees (Brown *et al.* 1989).

The BGB percentage contribution to the total tree biomass in the old-growth broad-leaved, old-growth pine, regenerating pine and regenerating broad-leaved forests were 18.0, 8.7, 12.2 and 43.4%, respectively which were within the range reported by earlier workers. The BGB of an ecosystem can reach up to 25% of the total tree biomass (Cairns *et al.* 1997). The contribution of herbs (0.003 – 0.3) and shrubs (0.01 – 0.6) components to the total forest biomass in all the forest types was negligible. Brown (1997) concluded that herbs and shrubs can contribute up to 3% of the total forest AGB.

Fine and coarse root biomass of *P. kesiya* contributed 6.0% and 72.1% to the total root biomass, respectively. The fine root biomass was (2.4 Mg ha^{-1}) lower than the values reported by Arunachalam *et al.* (1996b) (3.4 - 5.1 Mg ha^{-1}) and by John *et al.* (2001) (4.6 Mg ha^{-1}) in a 22 years old and 23 years old *P. kesiya* plantation forest indicating the reduction of fine root production in old-growth forest. The fine root biomass obtained in the present study was within the global range of 1.0 - 17.7 Mg ha^{-1} for various ecosystems (Vogt *et al.* 1986). The fine root biomass obtained in the regenerating pine forest (0.9 Mg ha^{-1}) was also lower than the earlier reports but within the reported values of Vogt *et al.* (1986). However, the fine root biomass (4.4 Mg ha^{-1}) in regenerating broad-leaved forest was within the reported range of earlier reports (Arunachalam *et al.* 1996b; John *et al.* 2002; Vogt *et al.* 1986). Low availability of soil nutrients and water has been reported to promote high production and accumulation of fine roots (Vogt *et al.* 1986). The extremely low BGB (40.8 Mg ha^{-1}) as obtained in this study may be attributed to this reason (Baishya and Barik, 2011).

In agreement with the findings of earlier workers, all the montane sub-tropical forest types with less AGB in higher diameter class had greater potential to accumulate significant quantities of biomass, and thus sequester more atmospheric CO_2 in the future. However, given the fact that all the forest types had large tree populations under smaller dbh classes <60 cm, it can be argued that these forests have high potential for carbon sequestration.

Net primary productivity in the montane sub-tropical forest ecosystem

The old-growth forests are considered to be carbon neutral that is the amount of carbon uptake during photosynthesis equals the release through plant and soil respiration (Melillo *et al.* 1996). However, increasing evidence shows that old-growth, undisturbed forests can still accumulate carbon even as net primary productivity (NPP) declines (Carey *et al.* 2001; Luyssaert *et al.* 2008). About half of primary forests are located in the tropical and montane sub-tropical regions, and the rate of net carbon uptake by these forests and their response to increasing CO_2 levels has been shown to be much higher than those of temperate or boreal forests (Wang and Polglase, 1995). Therefore, total net carbon uptake by old-growth forests may account for more than 10% of total land carbon uptake globally (Tang *et al.* 2011). The total aboveground NPP in the various montane sub-tropical forest types was 3.2 – 20.6 Mg ha^{-1} yr^{-1} with highest aboveground NPP in old-growth pine forest (20.6 Mg ha^{-1} yr^{-1}) and lowest in regenerating pine forest (3.2 Mg ha^{-1} yr^{-1}). The old-growth broad-leaved forest and the regenerating broad-leaved forest had aboveground NPP of 15.0 and 3.9 Mg ha^{-1} yr^{-1}, respectively. The aboveground NPP value in the old-growth broad-leaved forest is comparable with the findings of Clark *et al.* (2001b) for the old-growth tropical forests of Ivory Coast (7.9 – 14.3 Mg ha^{-1} yr^{-1}). The aboveground NPP value however was higher than the aboveground NPP (9.9 Mg ha^{-1} yr^{-1}) for the old-growth broad-leaved forest of Thailand and the tropical forests (3.9 – 4.1 Mg ha^{-1} yr^{-1}) of India (Clark *et al.* 2001b). The aboveground NPP of the regenerating broad-leaved forest was within the range for tropical forests of Thailand (Clark *et al.* 2001b).

The total ecosystem NPP of the old-growth pine forest was high (22 Mg ha^{-1} yr^{-1}), which may be attributed to relatively higher soil NPK content and moisture regime in this pine forest. Gower *et al.* (1994) and Haynes and Gower (1995) reported that N fertilization increases NPP of forest, and carbon allocation to belowground components decreases with increase in soil N availability. Studies on even-aged forest stands have revealed that aboveground NPP peaks in the early stand development process and then gradually declines as a result of decreasing soil nutrient availability and increasing stomatal limitation, leading to reduced photosynthetic rates (Gower *et al.* 1996). Vogt *et al.* (1996) reported that root NPP does not depend only on nutrient availability for many pine species. Relatively low root NPP as observed in this study (1.4 Mg ha^{-1} yr^{-1}) in comparison to the other works (Karizumi 1974) supports this argument.

The aboveground NPP of old-growth *Pinus kesiya* (20.6 Mg ha^{-1} yr^{-1}) is comparable with the findings of Tanabe *et al.* (2003) for the *Pinus densiflora* (10 – 38.2 Mg ha^{-1} yr^{-1}) forest from Japan. The aboveground NPP in this old-growth pine forest was however greater than the younger (5 - 22 years old) *P. kesiya* forest (30.1 - 20.1 Mg ha^{-1} yr^{-1}) reported by Das and Ramakrishnan (1987). The total aboveground NPP was higher than most pine forests around the world. For example, Chaturvedi and Singh (1982, 1987) and Rana *et al.* (1989) calculated the aboveground NPP for *Pinus roxburghii* forest (6.1 - 15.6 Mg ha^{-1} yr^{-1}) in the central Himalayas of India. Ma (1988) reported NPP of 3.5 - 17.5 Mg ha^{-1} yr^{-1} for 33-70 years old *Pinus tabulaeformis* forest from China. The aboveground NPP of the regenerating pine forest is comparable with the aboveground NPP values obtained by Ma (1988) for the 33 - 70 years old *Pinus tabulaeformis* forest from China. The high AGB and NPP in the old-growth pine forest may be due to addition of detrital biomass and NPP to the aboveground biomass. The high productivity of old-growth *P. kesiya* may be attributed to high net assimilation rate due to prolonged photosynthetic activity. The calculated aboveground NPP value in the *P. kesiya* forest (20.6 Mg $ha^{-1}yr^{-1}$) was greater than the 6.1 - 15.6 Mg $ha^{-1}yr^{-1}$ ANPP for *Pinus roxburghii* forest in the central Himalayas of India (Rana *et al.* 1989). The *P. kesiya* aboveground NPP was also higher than the studied ecosystem level NPP in the *Pinus densiflora* forest in Japan (10.0 - 14.6 Mg $ha^{-1}yr^{-1}$), but lower than 3.5 - 4.9 Mg $ha^{-1}yr^{-1}$ reported by Karizumi (1974). The belowground NPP value reported by Karizumi (1974) however was within the calculated value for the regenerating pine forest of Upper Shillong (4.4 Mg $ha^{-1}yr^{-1}$).

The regenerating pine and regenerating broad-leaved forests of montane sub-tropical landscape with high belowground NPP may be due to high clay content. It has been reported that forests with higher clay content tend to allocate a lower proportion of their NPP to below-ground (Aragao *et al.* 2009).

The study on biomass and carbon sequestration potential of old-growth broad-leaved forest, and the old-growth pine forest revealed more carbon stock is there in these two forest types than the regenerating pine and regenerating broad-leaved forest. However, the regenerating pine and regenerating broad-leaved forests with low biomass and NPP values due to their younger age has tremendous potential for carbon sequestration in the future. Therefore, both these forests need to be conserved as the old-growth forests after maturity becomes carbon neutral with the amount of carbon sequestered being equal to the amount of carbon release through dead and decomposition.

CHAPTER VII

GENERAL DISCUSSION

Forest ecosystems are the largest pool of biomass and carbon among all the terrestrial ecosystem. They absorb large quantities of CO_2 form the atmosphere through photosynthesis, and also return a large quantity of sequestered carbon (C) back to the atmosphere through autotrophic and heterotrophic CO_2 efflux (Lorenz and Lal, 2010). It has been estimated that about 234 Pg C are stored in the aboveground compartment, 62 Pg C in the belowground compartment, 42 Pg C in the dead woody compartment, 23 Pg C in litter compartment, and a maximum carbon sink of 398 Pg C in the forest soils (Kindermann *et al.* 2008). These carbon pools are dynamic and change with change in land-use. Thus, forest degradation and deforestation have a major impact on the forest carbon stock. The tropical forests store large amount of carbon and are therefore important in the global terrestrial carbon cycle (Houghton 1996; Houghton *et al.* 2001). However, very little is known about the levels of carbon storage in different tropical forests (Baishya *et al.* 2009). Assessment of the rates of carbon accumulation by different tropical and montane sub-tropical forests would help in estimating the fact that the tropical and sub-tropical landscape, serve as sinks of atmospheric carbon. The estimates of carbon stock are also important for generating empirical data on forest primary productivity and nutrient cycling.

Physico-chemical properties of soil and soil organic carbon

The soil texture played an important role in all the tropical and montane sub-tropical forest types of Nongkhyllem wildlife sanctuary and Upper Shillong. The clay content was higher in the sub-surface soil layer whereas WHC was higher in the surface soil layer in both the landscape, which had greater accumulation of organic matter thereby indicating a stronger influence of SOC on WHC than the clay particles. The water holding capacity was higher than the reported value of 52 – 67% in the Kumaun Hilamalaya (Rikhari *et al.* 1991). The clay percentage ranged between 9.2 – 13.3% in the surface layer and 11.2 – 17.3% in the sub-surface layer of tropical landscape. The corresponding figures for montane sub-tropical landscape were 7.2 – 9.4% and 9.2 – 11.4%, respectively.

Soil bulk density is an important indicator of soil compaction and the ability to function for structural support, water, solute movement and soil aeration (Acosta-Martinez *et al.* 1999). The bulk density was more in the tropical than the montane sub-tropical landscape. The strong negative correlation between bulk density and SOC was observed in both the tropical ($R^2 = 0.84$; $p = 0.002$) and montane subtropical forests ($R^2 = 0.96$; $p = 0.003$). These findings are similar to the findings of Li *et al.* (2007).

The acidic condition of soils of the tropical and montane sub-tropical forests were related to soil organic matter and nitrogen contents. These two soil nutrients were also highly related to each other as shown by Shrestha (1979).

The soil nitrogen availability to plant is largely determined by the mineralization process during decomposition of organic matter (Jordan 1985; Swift *et al.* 1979) and thereby affecting tree growth (Yamakura and Sahunalu, 1990). Relatively higher concentration of TKN in the tropical and montane sub-tropical forest soils than the earlier studies in the tropics could be due to higher organic matter concentration in the soils. A sharp decline in TKN value from 0.22% in regenerating broad-leaved forest to 0.18% in teak plantation forest in the tropical landscape and from 0.50% in old-growth broad-leaved forest to 0.30% in old-growth pine forest and regenerating pine forest of montane sub-tropical landscape may be due to runoff losses caused by heavy rainfall. The SOC did not show any correlation with

TKN in the tropical landscape ($R^2 = 0.237$; $p = 0.154$) but was positively correlated in the sub-tropical landscape ($R^2 = 0.834$; $p < 0.01$).

Available phosphorus in the soils of north-eastern India is always low. Therefore, being a limiting factor, its availability in soil is critical for significant carbon fixation. In general, the available phosphorus was higher in the montane sub-tropical landscape than the tropical landscape.

The exchangeable K values were also greater in tropical than montane sub-tropical. It was highest in the sal plantation forest (454 $\mu g\ g^{-1}$) and lowest in the teak plantation forest (117.5 $\mu g\ g^{-1}$) of tropical landscape and highest in the old-growth broad-leaved forest (239.5 $\mu g\ g^{-1}$) and lowest in the old-growth pine forest (74 $\mu g\ g^{-1}$) of montane sub-tropical landscape. Exchangeable Potassium showed a positive correlation with SOC in both the tropical ($R^2 = 0.751$; $p < 0.01$) and montane sub-tropical ($R^2 = 0.415$; $p < 0.05$) landscapes. The relative lower values in montane sub-tropical soils were attributed to the origin of the soils, and high rainfall (Yawson *et al.* 2011).

Soil carbon pool in tropical and montane sub-tropical forest ecosystems

Carbon input, magnitude of soil organic carbon pools, and finally carbon mineralization depend on many factors. The impact of land-use changes on organic carbon pools in the mineral soil depends on long-term site-specific factors (e.g. climate, topography and parent material) and is often overridden by the high spatial heterogeneity of soil organic carbon (Brown and Lugo, 1982; Smithson *et al.* 2002; Schwendenmann *et al.* 2007).

In both the tropical and montane sub-tropical landscapes, the soil organic carbon pool upto a depth of 1 m varied significantly ($p<0.01$) among the four seasons in all the forest types. It was highest during summer and lowest during winter season. SOC declined with the depth ($p<0.01$).

The C/N ratios in the surface (0-10 cm) and sub-surface (10-20 cm) soil layers of tropical and montane sub-tropical landscapes ranged between 7.1 – 9.6 and 7.3 – 10.3 and 5 – 9 and 4.9 – 10.6, respectively. The faster rate of decomposition ($k = 0.06$) in the mixed bamboo forest was due to high C/N ratio as is evident from litter decomposition studies. In the montane sub-tropical landscape, higher decomposition rate in the old-growth broad-leaved forest ($k = 0.31$) and the regenerating broad-leaved forest ($k = 0.84$) was because of relatively higher C/N ratio than the two pine forests. The tropical forest types had higher C/N ratio than the montane sub-tropical forest types. The soil C/N ratio is an index of N mineralization, decomposition and quality of the organic matter in soil (Springob and Kirchmann, 2003).

Microbial biomass pool in the tropical and montane sub-tropical forest ecosystems

Ecosystems with high organic matter input and easily available organic compound tend to have higher microbial biomass contents and activities because they are preferred energy source for the microorganisms (Hassink 1994). The quantity and composition of microbial biomass is sensitive to change in the soil chemical and physical environment (Bauhus and Khanna, 1994; Wardle 1992; Wolters and Joergensen, 1991).

The microbial population was generally high in the surface soil layer owing to higher organic matter content, better aeration and greater nutrient availability in both the tropical and montane sub-tropical landscapes. The seasonal change in soil moisture, soil temperature and available residue could have a strong effect on soil microbial biomass and its activity (Diaz-Ravina *et al.* 1995; Insam 1990).

In the tropical landscape, the MBC pool was highest in the sal plantation forest (420.4 μg g^{-1}) and lowest in the regenerating broad-leaved forest (278.4 μg g^{-1}). Based on mean MBC concentration values in the surface soil layer, the sal plantation forest had significantly greater MBC than mixed bamboo, old-growth broad-leaved, teak plantation and regenerating broad-leaved forests. In the montane sub-tropical landscape, the MBC pool was highest in the regenerating broad-leaved forest (359.7 μg g^{-1}) and lowest in the regenerating pine forest (189.9 μg g^{-1}).

The contribution of MBC to the soil organic carbon in the surface soil layer of tropical landscape was 1.2 - 4.9% and the proportion in the montane sub-tropical landscape was 0.9 – 2.0%. The greater percentage contribution of MBC to SOC in the tropical landscape suggests that microbial communities at these sites have evolved a more complex system of substrate-use. High MBC was observed during summer season and low during winter season. These results are in conformity with those of Singh (2002) who reported higher MBC during warm-wet months and lower during dry-cold period in the undegraded forest of Arunachal Pradesh. The SOC showed a strong positive correlation with MBC in the tropical (R^2=0.9, p<0.001) and montane sub-tropical (R^2=0.91, p<0.001) landscapes.

These differences in the microbial biomass C in the tropical and montane sub-tropical forest types may be due to the climatic conditions, differences in ground cover vegetation, the number of roots, soil types and properties, types of land use and management (Anderson and Domsch, 1989; Murrieta *et al.* 2007). Apart from the soil moisture, detrital material on the forest floor and relatively high temperature are important factors that favour growth of microbial population in soil during rainy season. Contrary to these conditions, when soil moisture and temperature decline during winter, conditions become unfavourable for microbial growth. This suggests greater role of soil moisture and ambient temperature in seasonal variation of MBC on the forest floor. The ratio of microbial biomass C to soil organic C provides an insight into the C status of soil and a decline in the ratio indicates loss of soil organic C (Anderson and Domsch, 1989).

Carbon flux through litterfall in tropical and montane sub-tropical landscapes

The species composition plays an important role in the litter production of forest ecosystem in the same climatic range (Facelli and Pickett, 1991). The similarity in seasonal pattern of litterfall in all the forest types in spite of difference in species composition indicates strong control of climatic factors on this process. In the present study, the peak litterfall observed during winter and spring season in both the landscapes was similar the observations made in evergreen broadleaved forest (Arunachalam *et al.* 1996b; Khiewtam and Ramakrishnan, 1993; Yang *et al.* 2004).

The litter production in the tropical and montane sub-tropical landscapes ranged between 6.5 – 10.8 Mg ha^{-1} yr^{-1} and 4.2 – 10.1 Mg $ha^{-1}yr^{-1}$, respectively. The litter production was highest in sal plantation in the tropical landscape and old-growth broad-leaved forest in montane sub-tropical landscape. A significant proportion of the terrestrial NPP is recycled between the trees as litterfall and the forest floor (Morison 1991). The high litter production in both the forest type may be because of high density, basal area, leaf size and leaf dry weight. The life span, size and weight of leaf play an important rate in the litter production (Singh 1980).

The litter accumulation in the tropical (2,172 – 5,865 kg ha^{-1}) and the montane sub-tropical forest types (3,867 – 6,015kg ha^{-1}) indicated the slow decomposition process in the montane sub-tropical landscape. The greater litter accumulation in the montane sub-tropical forest types was due to slower decomposition rate (k = 0.31 – 1.25) and turnover rate (0.35 – 0.58 yr^{-1}) than the tropical forest types (k = 0.06 – 0.12) and turnover rate (0.45 – 1.01 yr^{-1}). The litter accumulation in the tropical forest types was inversely related to its turnover rate. This finding are consistent with the results of Vogt *et al.* (1986), who assigned rapid decomposition of litter as an important cause of lower litter accumulation on the forest floor of wet tropical forest. Pargasan and Parthasarathy (2005) also reported that the significant differences in total litter accumulation between two tropical dry evergreen forest sites on the Coromandel coast of south India, was due to differences in temperature, soil moisture and decomposition rates besides vegetation variables.

Litter decomposition plays a crucial role in the nutrient budget of the tropical forest ecosystems where vegetation depends mainly on the recycling of nutrients present in the plant detrital matter (Vogt *et al.* 1986). The decomposition rate is strongly influenced by climatic conditions and initial chemical composition of the litter (Couteaux *et al.* 1995). Among the climatic variables, rainfall and air temperature determine the rate of decomposition in areas subjected to unfavourable weather conditions (Arunachalam *et al.* 1998b). Litter decomposition in forests of the humid tropics is generally perceived as a rapid process, because of favorable climatic conditions for microbial activity. The tropical forest types showed a rapid phase of decomposition followed by stabilization. The faster initial phase of decomposition reflects the rapid process of decomposition and leaching of soluble compounds (Swift *et al.* 1979; Loranger *et al.* 2002). The findings of the present study revealed that the weight loss was much faster in the tropical forest types than the montane sub-tropical forest types.

The physical environment, especially soil moisture, temperature, relative humidity and microbial biomass are important in litter decay as these factors regulate the biological activity of soil (Sujhata *et al.* 2003). The rapid rate of decomposition after an initial lag-phase may reflect the leaching of soluble compounds and the decay of easily degradable compounds and tissues (Swift *et al.* 1979; Loranger *et al.* 2002). After the initial phase, decomposition rate appeared to be rather constant in the tropical forest types due to the higher fraction of cellulose, lignin and tannin present in the residue during the advanced stage of leaf decay. The decomposition rate was fastest in the mixed bamboo forest (k=0.06) of tropical landscape and the old-growth broad-leaved forest of montane sub-tropical landscape (k = 0.31). The lower decomposition rates in the montane sub-tropical landscape elements may be attributed to lower temperature and reduced microbial activity and presence of organic rich residues on the forest floor due to its openness. Weight loss of litter is related to microbial activity favoured by suitable environmental conditions and release of nutrient and accumulation of organic carbon on the forest floor.

CO_2 efflux in the tropical and montane sub-tropical forest ecosystems

In most ecosystems, microorganisms are responsible for soil respiration (Li *et al.* 2005). Microorganisms are generally considered as the driving force behind soil decomposition processes (Smith and Paul, 1995). It has been observed that seasonal change in microbial biomass could have important ramifications for nutrient cycling and ecosystem functioning (Lipson *et al.* 1999; Singh *et al.* 1989; Wardle, 1998; Zak *et al.* 1990). Although there are a number of studies on soil CO_2 efflux and microbial biomass comparative values among various tropical and sub-tropical forests, only a few studies have characterized soil CO_2

efflux and microbial biomass under tropical plantations and secondary forests (Amatya *et al.* 2002; Tufekcioglu *et al.* 2001).

The CO_2 efflux rate in the tropical forest types was higher than the montane sub-tropical forest types. The CO_2 efflux rate was high in the mixed bamboo forest of tropical landscape and the regenerating broad-leaved forest of montane sub-tropical landscape. Favourable soil conditions during summer season promote maximum microbial activity in the soil. The decrease in soil CO_2 efflux values during winter season is in accordance with the report of Mo *et al.* (2005). The soil respiration varied with the forest type, depth and season, which could be attributed to the differences in temperature, soil water content, root biomass and soil microbial biomass (Yi *et al.* 2007). Mallik and Hu (1997) reported that soil organic matter influences soil microbial respiration and is one of the important factors controlling it.

Relationship between soil CO_2 efflux and microbial biomass

Soil CO_2 efflux showed a positive correlation with microbial biomass carbon in both the tropical and montane sub-tropical landscapes indicating a strong influence of microbial biomass on CO_2 efflux. The findings are in support of similar results obtained in the forests of Puerto Rico by Li *et al.* (2005).

Ecosystem level biomass and carbon stock in the tropical and montane sub-tropical forest ecosystems

The carbon pool in different compartment of the ecosystem and flux rate in the different tropical and montane sub-tropical forest types are presented in Fig. 7.1 - 7.9.

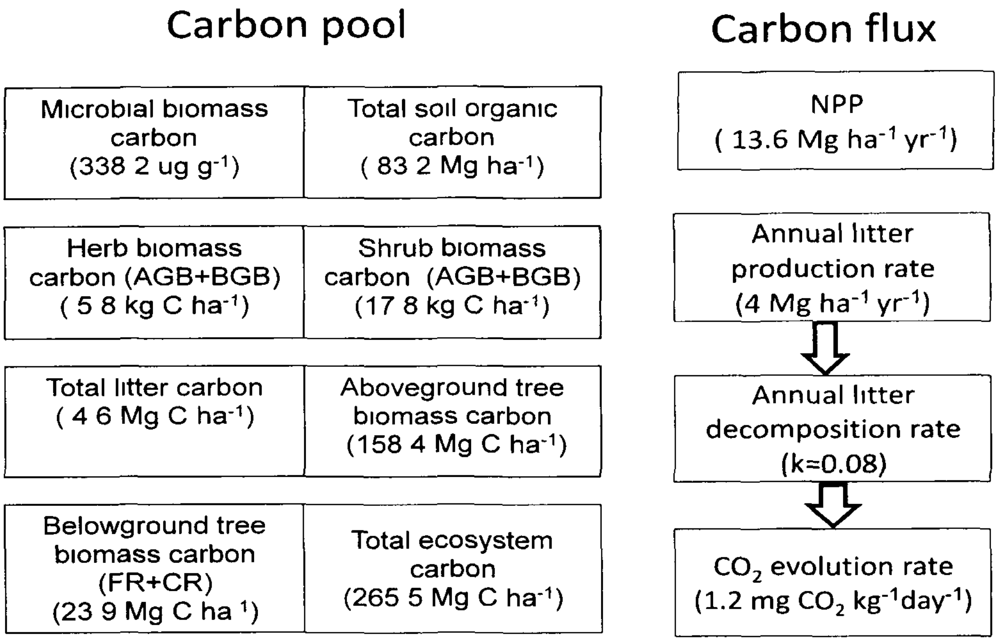


Fig. 7.1 Diagrammatic representation of carbon pool size and flux in the old-growth broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

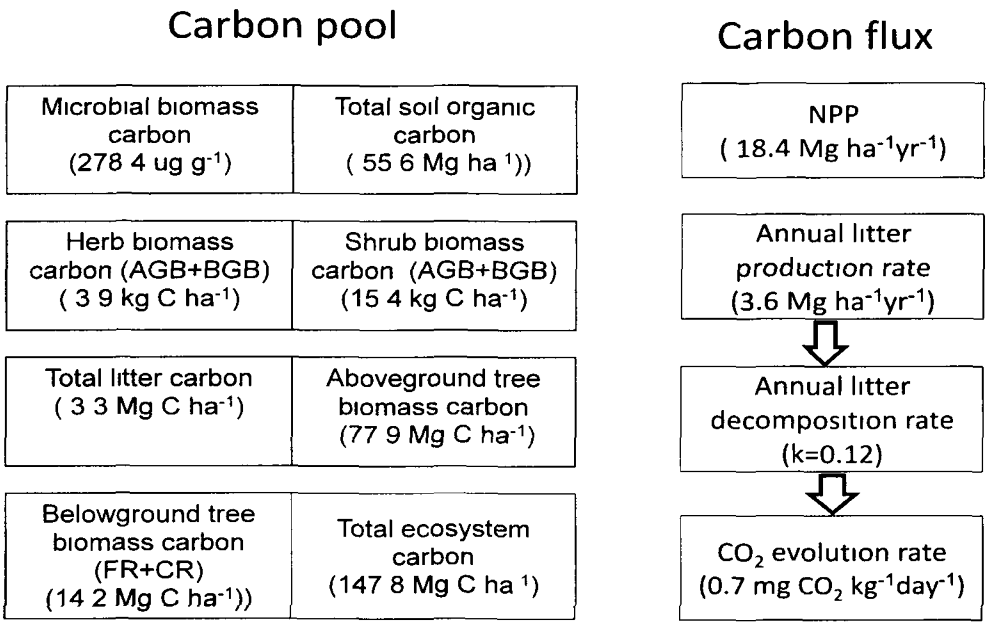


Fig. 7.2 Diagrammatic representation of carbon pool size and flux in the regenerating broad-leaved forest in tropical landscape of Nongkhyllem wildlife sanctuary

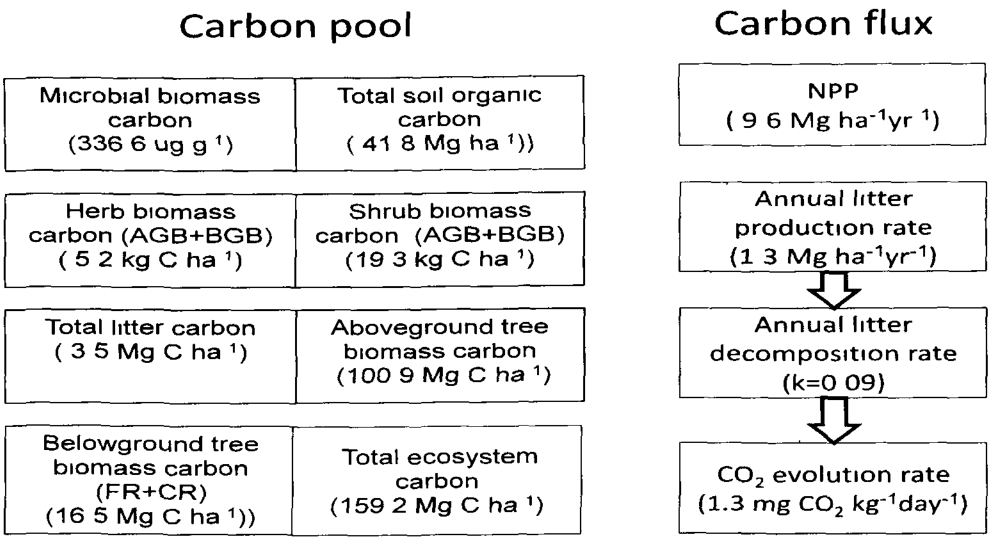


Fig. 7.3 Diagrammatic representation of carbon pool size and flux in the teak (*Tectona grandis*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

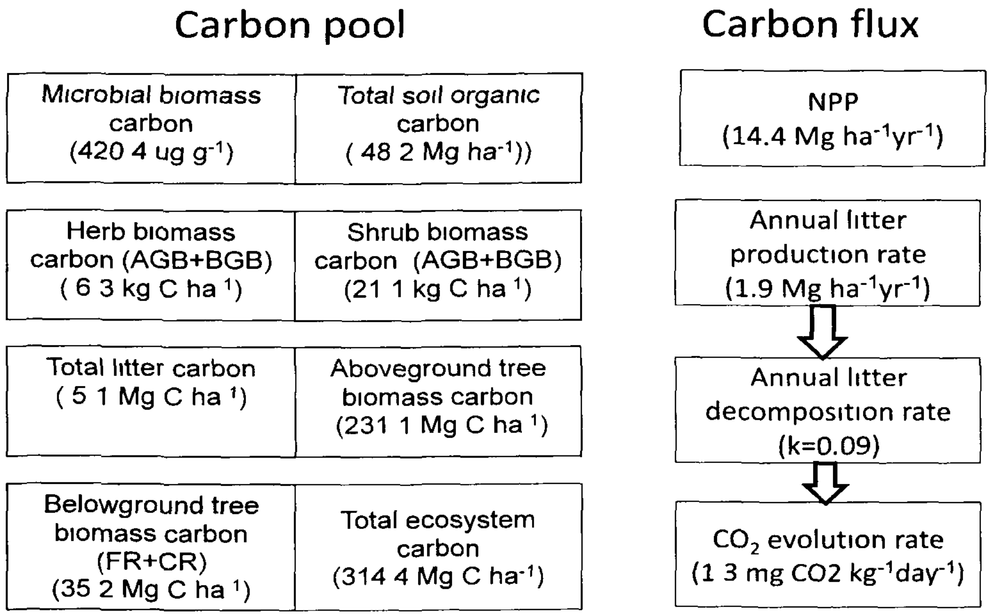


Fig. 7.4 Diagrammatic representation of carbon pool size and flux in the sal (*Shorea robusta*) plantation forest in tropical landscape of Nongkhyllem wildlife sanctuary

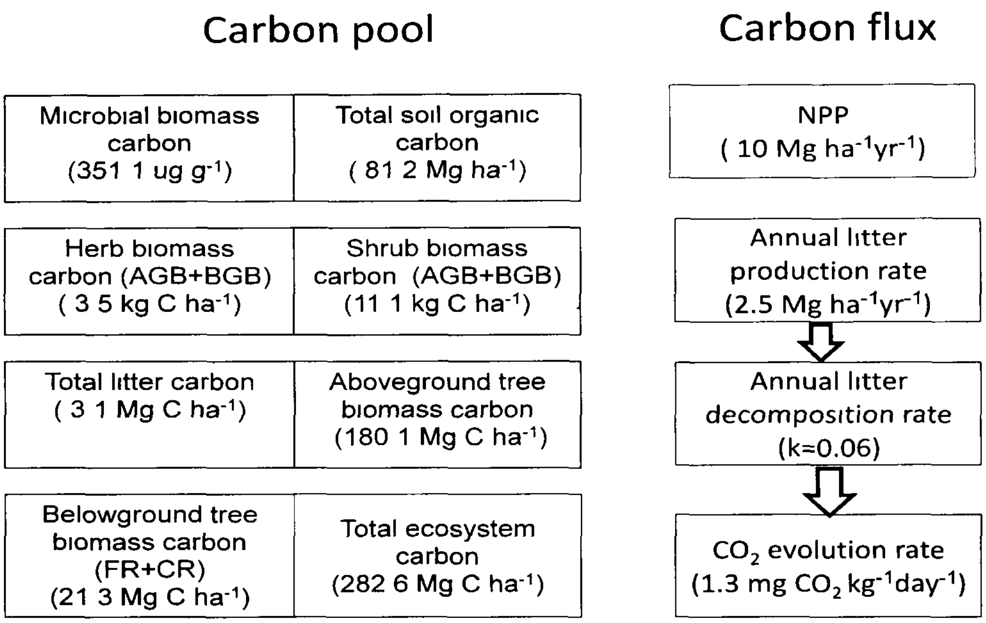


Fig. 7.5 Diagrammatic representation of carbon pool size and flux in the mixed bamboo forest in tropical landscape of Nongkhyllem wildlife sanctuary

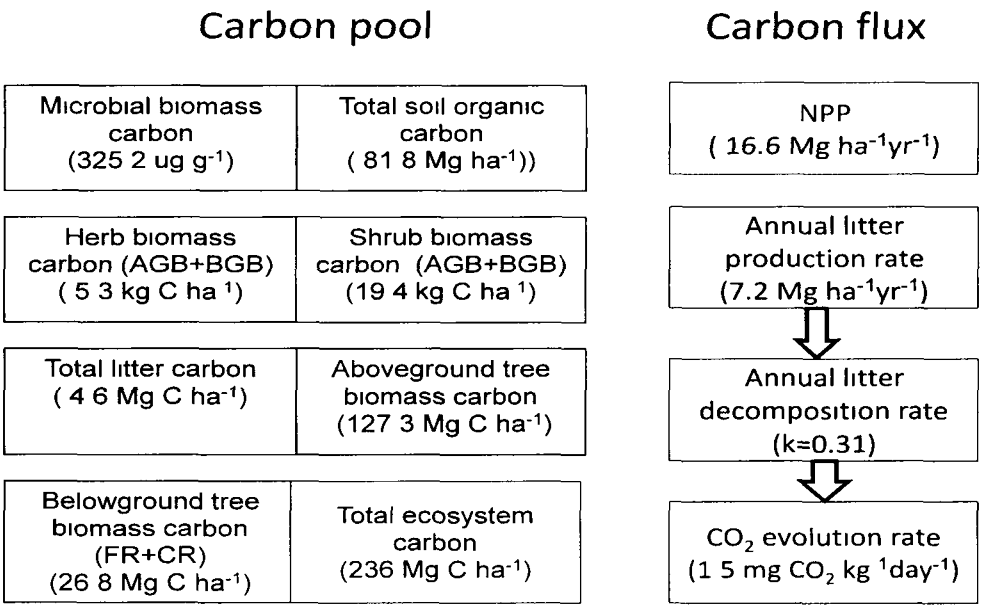


Fig. 7.6 Diagrammatic representation of carbon pool size and flux in the old-growth broad-leaved forest in montane sub-tropical landscape of Upper Shillong

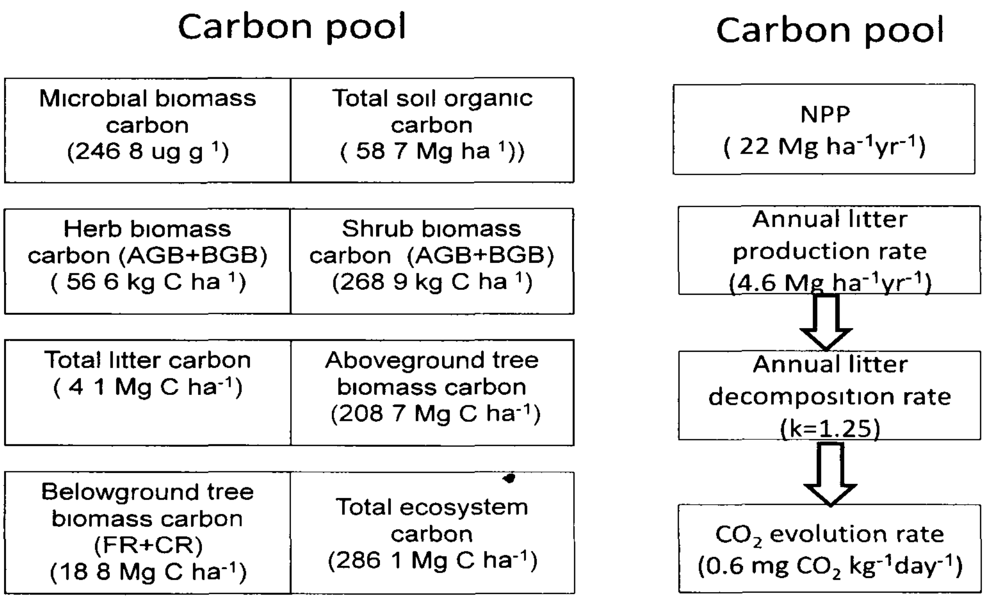


Fig. 7.7 Diagrammatic representation of carbon pool size and flux in the old-growth pine (*Pinus kesiya*) forest in montane sub-tropical landscape of Upper Shillong

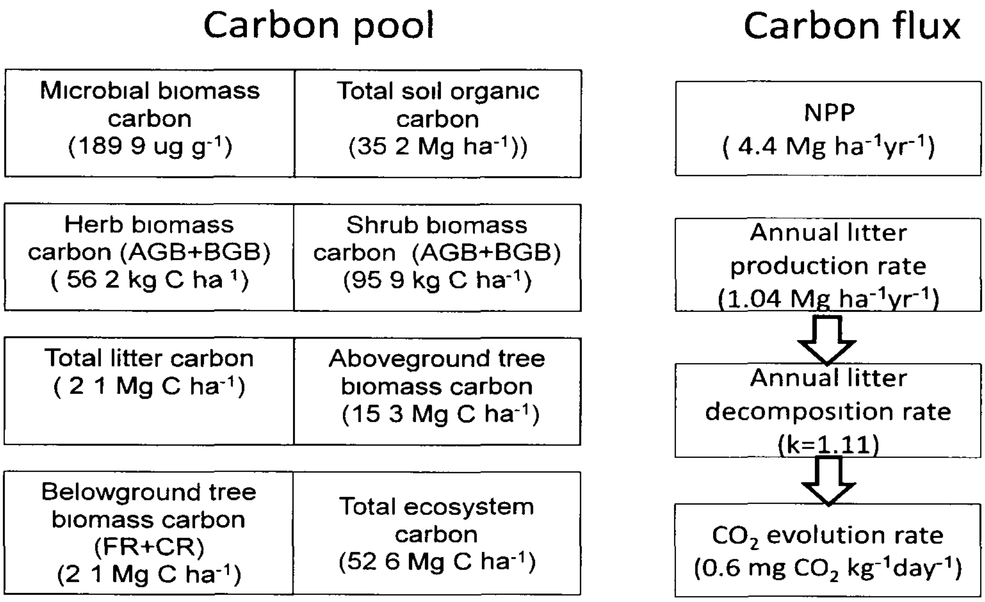


Fig. 7.8 Diagrammatic representation of carbon pool size and flux in the regenerating pine (*Pinus kesiya*) forest in montane sub-tropical landscape of Upper Shillong

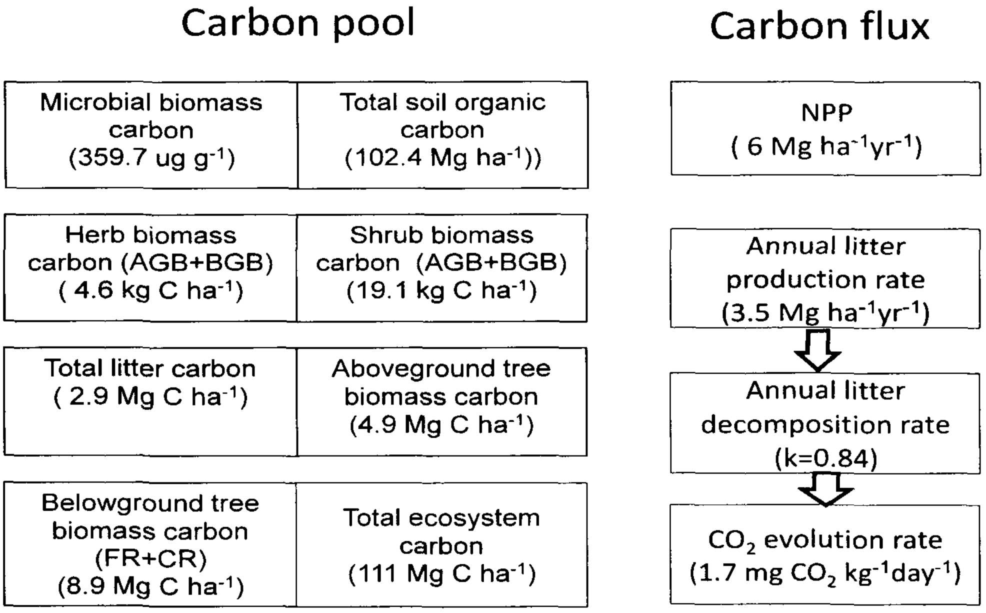


Fig. 7.9 Diagrammatic representation of carbon pool size and flux in the regenerating broad-leaved forest in montane sub-tropical landscape of Upper Shillong

The potential of forests to sequester carbon depends on the forest type, age of forest and diameter size class of trees (Terakunpisut *et al.* 2007). The biomass stock in the tropical old-growth forests remove carbon dioxide from the atmosphere at rates that vary with climate (Carey *et al.* 2001; Pregitzer and Euskirchen, 2004; Zhou *et al.* 2006). Old-growth forests accumulate carbon for long period and contain large quantities of carbon. The sequestered carbon dioxide is stored in live woody tissues and slowly decomposing organic matter in litter and soil. Old-growth forests therefore serve as a global carbon dioxide sink, but they are not protected by international treaties, because it is generally thought that ageing forests cease to accumulate carbon (Kira and Shidei, 1967).

The total aboveground biomass (TAGB) in the tropical forest types ranged between 159.3 – 472.0 Mg ha^{-1} with the highest being in the sal plantation forest, followed by mixed bamboo forest, old-growth broad-leaved forest, teak plantation forest and lowest in the regenerating broad-leaved forest. The TAGB in the montane sub-tropical forest types ranged between 10.3 – 425.9 Mg ha^{-1} with highest biomass stock in the old-growth pine forest and old-growth broad-leaved forest. The regenerating pine forest and regenerating broad-leaved forest had very low TAGB due to its young age structure. On the basis of TAGB stock, the sal plantation forest and the mixed bamboo forest had higher TAGB stock in the tropical landscape and the old-growth pine forest and the old-growth broad-leaved forest in the montane sub-tropical landscape.

The tropical forests carry more than 90% of the total biomass in the aboveground compartment, while the temperate forests stores >50% biomass in the belowground compartment due to net productivity differences which alters the allocation pattern of biomass in the forest system (Brown *et al.* 1989). Therefore, tropical forest inventories, which ignore dead matter, will be a small loss of proportion to total aboveground biomass than similar inventories in the temperate zone (Terakunpisut *et al.* 2007). The total belowground biomass (TBGB) in tropical landscape ranged between 30.3 – 73.2 Mg ha^{-1}, with the highest being in the sal plantation forest. The highest contribution of BGB to the total ecosystem biomass ranged between 11 – 16%, with highest being in the regenerating broad-leaved forest. This is in agreement with the findings of Cairns *et al.* (1997), who reported that the belowground biomass can reach a maximum of 25% in tropical forest ecosystems.

In the montane sub-tropical landscape, the TBGB ranged between 4.4 – 57.1 Mg ha^{-1}, with the highest being in the old-growth broad-leaved forest. The BGB contributed a maximum of 8.7 – 43.4% to the total ecosystem biomass, with the highest being in the regenerating broad-leaved forest. The high BGB compared to the total ecosystem biomass in the regenerating broad-leaved forest and regenerating pine forest was due to net productivity difference and low clay content in the soil. Moreover, low availability of soil nutrients and water has been reported to promote high production and accumulation of fine roots (Vogt *et al.* 1986).

The ecosystem level biomass was highest in the sal plantation forest (545.2 Mg ha^{-1}) of tropical landscape and the old-growth pine forest (466.6 Mg ha^{-1}) of montane sub-tropical landscape. The ecosystem level biomass of both the old-growth broad-leaved forests of tropical and montane sub-tropical landscape was significantly lower than the sal plantation and old-growth pine forests. The ecosystem level carbon stock in the tropical landscape was highest in the sal plantation forest and mixed bamboo forest. The corresponding figures in the montane sub-tropical landscape were high in the old-growth pine forest and old-growth broad-leaved forest.

A higher proportion of AGB was in the higher diameter classes (>60 cm) in old-growth broad-leaved (43%), sal plantation forest (32%) mixed bamboo forest (52%) of tropical landscape, and the old-growth pine forests (16%) of montane sub-tropical landscape. This trend does indicate the important role of large trees in carbon storage, but does not undermine the role of small trees (<60 cm dbh) which would enhance the future carbon stock because of their high carbon sequestration potential. The teak plantation forest, regenerating pine and the regenerating broad-leaved forests of both the landscapes with less biomass stock have the maximum carbon sequestration potential as they will store substantial amount of carbon as they grow till maturity. The old-growth forests after maturity turns into carbon source due to death of older trees in the forest (Chambers *et al.* 2001c; Lal and Singh, 2000. It is well established that forest plantations sequester carbon till maturity that varies from 25 to 75 years depending upon the forest type. Beyond maturity, the trees generally have marginal carbon sequestration capability (Lal and Singh, 2000). The higher AGB in these forests may be attributed to the more or less uniform stand structure that resulted from a combination of site factors, adopted management and silvicultural practices (Baishya *et al.* 2009).

Old-growth broad-leaved forests are reported to have net addition to standing biomass leading to carbon storage if most trees are yet to be matured. The matured forests do not add up any further biomass as most part of the gross primary productivity is either used up in respiration or returned to soil as litter with no net addition to the aboveground biomass (Brown *et al.* 1989). Such old-growth broad-leaved forests thus do not significantly contribute towards carbon uptake, though they are important for regeneration and sustaining biodiversity. However, plantation forests with higher annual productivity were reported to be ideal for carbon storage and sequestration (Lal and Singh, 2000). Thus, creation of new plantation on degraded lands is a better option for carbon storage when these are planted and harvested periodically for silviculture purpose.

Pattern of net primary productivity across the tropical and montane sub-tropical forest ecosystems

The net primary productivity (NPP) of tropical forests is one of the most important and least quantified components of the global carbon cycle. Most relevant studies have focused particularly on the quantification of the above-ground productivity, and little is known about the carbon fluxes involved in other components of NPP, the partitioning of total NPP between above- and below-ground components and the main environmental drivers of these patterns (Aragao *et al.* 2009). In this study the allocation pattern of above and below-ground NPP of nine forest types of tropical and montane sub-tropical landscapes of Meghalaya has been quantified. Tropical forests alone have a major impact on global carbon cycling, accounting for about a third of overall terrestrial NPP (Del Grosso *et al.* 2008; Field *et al.* 1998; Grace 2004; Malhi and Grace, 2000).

The total aboveground NPP in the different forest types of tropical landscape was highest in regenerating broad-leaved forest (16.1 Mg ha^{-1} yr^{-1}) and lowest in teak plantation forest (8.4 Mg ha^{-1} yr^{-1}). In the montane sub-tropical landscape, the aboveground NPP was highest in the old-growth pine forest (20.6 Mg ha^{-1} yr^{-1}) and lowest in regenerating pine forest (3.2 Mg ha^{-1} yr^{-1}). The contribution of belowground NPP to the total ecosystem NPP in all the tropical and montane sub-tropical forest types were low (6.4 – 12.5%), except in the regenerating pine and regenerating broad-leaved forests of montane sub-tropical landscape where it was high (28.5 and 33.9%). These values were consistent with the findings of Cuevas *et al.* (1991) whereas the secondary forests allocate upto 50% NPP in the belowground compartment.

The high aboveground NPP and low belowground NPP in these forests may be due to decreasing soil nutrient availability and increasing stomatal limitation, leading to reduced photosynthetic rates (Gower *et al.* 1996; Karizumi 1974; Vogt *et al.* 1996). The total ecosystem NPP of the regenerating broadleaved forest (18.4 Mg ha^{-1} yr^{-1}) of tropical landscape and old-growth broad-leaved forest (16.6 Mg ha^{-1} yr^{-1}) and old-growth pine forest (22 Mg ha^{-1} yr^{-1}) of montane sub-tropical landscape were high which may be attributed to relatively higher soil NPK content and moisture regime in these forests. Gower *et al.* (1994) and Haynes and Gower (1995) reported that N fertilization increases NPP of forest, and carbon allocation to belowground components decrease with increase in soil N availability. The regenerating pine and regenerating broad-leaved forests of montane sub-tropical landscape with high belowground NPP may be due to high clay content. It has been reported that forests with higher clay content tend to allocate a lower proportion of their NPP to below-ground (Aragao *et al.* 2009).

SUMMARY

Data on carbon sequestration potential for different forest types in tropical and montane sub-tropical regions of north-eastern India are non-existent. Carbon data for different compartments are essential to understand the carbon sequestration potential of different forest types. In order to bridge this data gap, the present study was undertaken to assess the carbon sequestration potential in soil and vegetation in nine different forest types of Meghalaya, of which five were in tropical landscape and four were in montane sub-tropical landscape. The five tropical forest types were: (i) old-growth broad-leaved forest (TOBF), (ii) regenerating broad-leaved forest (TRBF), (iii) teak (*Tectona grandis*) plantation forest (TTPF), (iv) sal plantation forest (TSPF), and (v) mixed bamboo forest (TMBF). All these forests were located in Nongkhyllem wildlife sanctuary and Nongkhyllem reserve forest. The four montane sub-tropical forest types included were: (i) old-growth broad-leaved forest (SOBF), (ii) old-growth pine (*Pinus kesiya*) forest (SOPF), (iii) regenerating pine (*Pinus kesiya*) forest (SRPF), and (iv) regenerating broad-leaved forest (SRBF). All these forest types were located in Upper Shillong area. The elevation range of the tropical and montane sub-tropical forest types ranged from 205 - 297 and 1672 – 1920 m. a.s.l., respectively. All the forest types receive very high rainfall and rainy season commences with the onset of southwest monsoon in May and continues up to September. Three fourth of the total annual rainfall is received during this season.

Carbon was estimated in the above mentioned forest types in tropical and subtropical landscapes in three important pools viz., vegetation biomass carbon pool, soil organic carbon pool, and microbial biomass carbon pool. Four important fluxes of carbon viz., Net primary production (NPP), litter fall, litter decomposition, and soil respiration were also estimated.

Community characteristics, tree population structure and soil physic-chemical properties of each forest type were analyzed to relate and explain the observed patterns of the carbon pools and fluxes.

Soil organic carbon (SOC) was estimated colorimetrically. Carbon percentage value thus obtained was converted into Mg C ha^{-1} value by using the formula: SOC (Mg C ha^{-1}) = %C x bulk density x soil depth. Microbial biomass carbon values were obtained following chloroform fumigation method. The plant carbon content was estimated by igniting the samples at 550°C for 6 hours in muffle furnace and carbon content was calculated as 50% of ash free mass. The biomass carbon for different forest types was obtained by multiplying biomass values with the carbon percentage obtained. Tree above ground and below ground biomass was estimated by developing/adopting appropriate allometric models. Litter flux was estimated at seasonal interval over a period of three years by using litter traps of 1m x 1 m x 0.15 m size, randomly placed in each permanent plot. Leaf litter decomposition in each forest type was studied using litterbag technique. Soil respiration (CO_2 efflux) rate was determined following alkali absorption method. Net primary productivity was calculated following 'net positive increment in biomass' approach and was derived from the data collected over a period of four years.

Vegetation characteristics of the tropical and montane sub-tropical landscape

The dominant tree species in the tropical landscape were *Schima wallichii*, *Actinodaphne obovata*, *Artocarpus chaplasha*, *Tectona grandis* and *Shorea robusta* depending on the forest type. *Dendrocalamus hamiltonii* and *Teinostachyum dullooa* were the two dominant bamboo species in the tropical mixed bamboo forest. The dominant tree species in the montane sub-tropical landscape were *Lindera latifolia*, *Pinus kesiya* and *Rhododendron arboretum*.

Tree density in the tropical landscape was maximum in the regenerating broad-leaved forest (1728 individuals ha^{-1}) and minimum in the mixed bamboo forest (794 individuals ha^{-1}). *Dendrocalamus hamiltonii* had a density of 3420 culms ha^{-1} whereas *Teinostachyum dullooa* had 1524 culm ha^{-1}. In the montane sub-tropical landscape, tree density was maximum in the regenerating broad-leaved forest (2106 individuals ha^{-1}) and minimum in the old-growth pine forest (628 individuals ha^{-1}). About 91.2 – 94.5% tree species in the tropical landscape exhibited clumped distribution pattern, and the rest 5.5 – 8.8% exhibited random distribution pattern. No tree species exhibited regular dispersion pattern in tropical landscape. The figures for clumped, random and regular distribution patterns in montane sub-tropical landscape were 48.6 – 83.3%, 5.7 – 33.3% and 11.1 – 45.7%, respectively.

Species richness in the tropical landscape was highest in the old-growth broad-leaved forest with 94 species and lowest in teak plantation forest with 33 species. In the montane sub-tropical landscape, the regenerating broad-leaved forest had higher species richness with 35 species while the regenerating pine forest had lowest species richness with 6 species.

Physico-chemical properties of soil and soil organic carbon

The soils of both the landscapes were sandy in texture. The clay content in the surface and sub-surface soil layers were higher in the tropical landscape than the montane sub-tropical landscape. The clay percentage ranged between 9.2 – 13.3% in the surface layer and 11.2 – 17.3% in the sub-surface layer in the tropical landscape. In the montane sub-tropical landscape, the clay percentage ranged from 7.2 – 9.4% in the surface and 9.2 – 11.4% in the sub-surface layers. Water holding capacity (WHC) was higher in the surface soil layer than the sub-surface layer in both the landscapes. Since the surface layer had greater accumulation of organic matter, it is argued that SOC influenced WHC more than the clay particles. A strong negative correlation between bulk density and SOC was observed in the tropical ($R^2 = 0.84$; $p = 0.002$) and montane sub-tropical forest types ($R^2 = 0.96$; $p = 0.003$). The soil moisture content (SMC) ranged from 15.3 – 21.4% in the tropical landscape, and 24.6 – 36.4% in the montane sub-tropical landscape. SMC was lower in the surface layer of all the tropical forest types than in the surface layer of all the montane sub-tropical landscape. The soils of both the tropical and montane sub-tropical forest types were acidic in reaction (pH = 4.2 - 6.3 and 4.2 – 5.9, respectively).

Total kjeldahl nitrogen (TKN) was high in the montane sub-tropical landscape (0.3 - 0.5%), and it was low in the tropical landscape (0.18 – 0.22%). SOC was positively correlated with TKN only in the montane sub-tropical landscape ($R^2 = 0.834$; $p < 0.01$). Available phosphorus (45.5 $\mu g\ g^{-1}$) was relatively high in the regenerating broad-leaved forest of montane sub-tropical landscape and sal plantation forest (19.9 $\mu g\ g^{-1}$) of tropical landscape, and low in the teak plantation forest (2.3 $\mu g\ g^{-1}$) of tropical landscape and old-growth broad-leaved forest (2.6 $\mu g\ g^{-1}$) of montane sub-tropical landscape. Available phosphorus did not show any significant correlation with soil organic carbon in the tropical landscape, but showed positive correlation in the montane sub-tropical landscape ($R^2=0.91$; $p<0.0001$). The exchangeable K was highest in the sal plantation forest (454.0 $\mu g\ g^{-1}$) and lowest in the teak plantation forest (117.5 $\mu g\ g^{-1}$) of tropical landscape. In the montane sub-tropical landscape, it was higher in the old-growth broad-leaved forest (239.5 $\mu g\ g^{-1}$) and lower in the old-growth pine forest (74 $\mu g\ g^{-1}$). Exchangeable Potassium showed a positive correlation with SOC in the tropical ($R^2 = 0.751$; $p < 0.01$) and montane sub-tropical ($R^2 = 0.415$; $p < 0.05$) landscapes.

Soil carbon pool in tropical and montane sub-tropical forest ecosystems

The soil organic carbon (SOC) pool in the tropical landscape upto 1 m depth was largest in the old-growth broad-leaved forest (83.2 Mg C ha^{-1}), followed by mixed bamboo forest (81.2 Mg C ha^{-1}), regenerating broad-leaved forest (55.9 Mg C ha^{-1}), sal plantation forest (48.2 Mg C ha^{-1}) and teak plantation forest (41.7 Mg C ha^{-1}). In the montane sub-tropical landscape, SOC pool was largest in the regenerating broad-leaved forest (102.4 Mg C ha^{-1}), followed by old-growth broad-leaved forest (81.8 Mg C ha^{-1}), old-growth pine forest (58.7 Mg C ha^{-1}) and it was lowest in the regenerating pine forest (35.2 Mg C ha^{-1}). The SOC pool was highest during summer and lowest during winter season in all the forest types of both the landscapes. SOC declined with depth ($p<0.01$). The SOC showed a negative correlation with bulk density in all the forest types of both the landscapes.

The soil C/N ratio increased with depth in all the tropical and montane sub-tropical forest types, except for the sal plantation forest where it showed a reverse trend due to less SOC in the sub-surface layer. The C/N ratios in the tropical and montane sub-tropical landscapes ranged from 7.1 – 10.3 and 4.8 – 10.6, respectively.

Microbial biomass and soil organic carbon

The microbial biomass carbon (MBC) pool in the surface and sub-surface soil layers of tropical landscape was highest in the sal plantation forest (420.4 & 293.9 µg g^{-1}, respectively) and lowest in the regenerating broad-leaved forest (278.4 & 169.4 µg g^{-1}, respectively). The corresponding values in the montane sub-tropical landscape were highest in the regenerating broad-leaved forest (359.7 & 240.7 µg g^{-1}, respectively), and lowest in the regenerating pine forest (189.9 & 123.5 µg g^{-1}, respectively). The contribution of MBC to the soil organic carbon in the surface soil layer of tropical landscape was 1.2% - 4.9%, and the proportion in the montane sub-tropical landscape was 0.9 - 2.0%.

The MBC showed a strong positive correlation with SMC in the tropical (R^2=0.9, p<0.001) and montane sub-tropical (R^2=0.68, p<0.005) landscape. A strong positive correlation was also observed between SOC and MBC in the tropical (R^2=0.9, p<0.001) and montane sub-tropical (R^2=0.91, p<0.001) landscape.

Carbon flux through litterfall in tropical and montane sub-tropical landscapes

The litter production in the tropical and montane sub-tropical forest types ranged from 6.5 – 10.8 Mg ha^{-1} yr^{-1} and 4.2 – 10.1 Mg ha^{-1}yr^{-1}, respectively. High litter production was observed in sal plantation forest (10.8 Mg ha^{-1} yr^{-1}) of tropical landscape and old-growth broad-leaved forest (10.1 Mg ha^{-1}yr^{-1}) of montane sub-tropical landscape. Similarly, the litter carbon pool in the tropical and montane sub-tropical forest types ranged from 3.3 – 5.1 Mg C ha^{-1} yr^{-1} and 2.1 -4.6 Mg C ha^{-1} yr^{-1}, respectively. The litter accumulation in the tropical and montane sub-tropical landscape ranged from 2,172 – 5,865 kg ha^{-1} and 3,867 – 6,015 kg ha^{-1}, respectively. The decomposition rate was fastest in the mixed bamboo forest (k=0.06) of tropical landscape and the old-growth broad-leaved forest of montane sub-tropical landscape (k=0.3). The decomposition rate in the tropical landscape followed the trend: TMBF>TOBF>TSPF>TTPF>TRBF and the montane sub-tropical landscape followed the trend: SOBF>SRBF>SRPF>SOPF.

CO_2 efflux in the tropical and montane sub-tropical forest ecosystems

CO_2 efflux rate in the tropical landscape was higher than the montane sub-tropical landscape. CO_2 efflux rate in the tropical landscape was highest in the mixed bamboo forest (1.7 mg CO_2 kg^{-1} day^{-1}) and followed the order: TMBF>TSPF>TOBF>TTPF>TRBF. In the montane sub-tropical landscape, the value was highest in the regenerating broad-leaved forest (1.3 mg CO_2 kg^{-1} day^{-1}) and followed the order: SRBF>SOBF>SOPF>SRPF. Soil CO_2 efflux showed positive correlation with microbial biomass carbon in the tropical ($R^2 = 0.65$; $P<0.01$) and montane sub-tropical landscape ($R^2 = 0.81$; $p<0.01$) indicating a strong influence of microbial biomass on CO_2 efflux.

Ecosystem level biomass and carbon sequestration in the tropical and montane sub-tropical forest ecosystems

Total aboveground biomass (TAGB) in the tropical forest types was highest in the sal plantation forest (472 Mg ha^{-1}) and followed the trend: TSPF>TMBF>TOBF>TTPF>TRBF. The TAGB in the montane sub-tropical forest types was highest in the old-growth pine forest (425.9 Mg ha^{-1}) and followed the trend: SOPF>SOBF>SRPF>SRBF. The contribution of belowground biomass (BGB) to the total ecosystem biomass in the tropical landscape ranged from 11.0 – 16.0%, with highest being in the regenerating broad-leaved forest (16.0%). The corresponding figures in the montane sub-tropical landscape ranged from 8.7 – 43.4% with highest contribution in the regenerating broad-leaved forest (43.4%). The herb and shrub biomass contributed a negligible amount to the total ecosystem biomass in both the tropical and montane sub-tropical landscapes. The proportion of AGB in the higher diameter classes i.e. >60 cm in the tropical landscape was high in the mixed bamboo forest (52%) followed by old-growth broad-leaved (43%) and sal plantation forest (32%). The proportion of AGB in the higher diameter classes i.e. >60 cm in the sub-tropical landscape was only 16% and it was only in the old-growth pine forests.

The ecosystem level biomass in the tropical landscape was highest in the sal plantation forest (545.2 Mg ha^{-1}) and followed the trend: TSPF>TMBF>TOBF>TTPF>TRBF. The highest figure in the montane sub-tropical landscape was in the old-growth pine forest (466.6 Mg ha^{-1}) and followed the trend: SOPF>SOBF>SRPF>SRBF.

The ecosystem level carbon sequestration value in the tropical landscape was higher than the montane sub-tropical landscape. In tropical landscape, it was highest in the sal plantation forest (314.4 Mg C ha^{-1}) and followed the trend: TSPF>TMBF>TOBF>TTPF>TRBF. The highest figure in the montane sub-tropical landscape was for the old-growth pine forest (286.1 Mg C ha^{-1}) and followed the trend: SOPF>SOBF>SRBF>SRPF.

Net primary productivity in the tropical and montane sub-tropical forest ecosystems

The ecosystem level NPP in the tropical landscape was highest in regenerating broad-leaved forest (18.4 Mg ha^{-1} yr^{-1}) and followed the trend: TRBF>TSPF>TOBF>TMBF>TTPF. In the montane sub-tropical landscape, the figure was highest in the old-growth pine forest (22 Mg ha^{-1} yr^{-1}) and followed the trend: SOPF>SOBF>SRBF>SRPF. The contribution of belowground NPP to the total ecosystem NPP in all the tropical and montane sub-tropical forest types were low (6.4 – 12.5%), except in the regenerating pine and regenerating broad-leaved forests of montane sub-tropical landscape where it was relatively high (28.5 and 33.9%).

Very few studies quantifying ecosystem level C are available. In the present study, the ecosystem level biomass carbon in the sal plantation forest was greater than the old-growth broad-leaved forest and mixed bamboo forest in tropical landscape. Similarly, old-growth pine forests had greater biomass C than the old-growth broad-leaved forest in montane sub-tropical landscape. The study revealed high soil carbon sequestration potential of the regenerating forests than the old-growth forests. NPP was greater in the old-growth pine forest and the regenerating broad-leaved forests than the other forest types. The future carbon sequestration potential of the forest with lower diameter classes was argued to be high, while the sal and the old-growth forests had lower carbon sequestration potential. Based on the NPP values, ecosystem level biomass C, tree diameter distribution, species composition and the age of the forest, it was concluded that the sal plantation forest and the old-growth pine forest had greater carbon stock than the other forest types, and also have sequestered more C than the others. The study concludes that large variation in carbon content exists among different forest types. Overall, the carbon sequestration potential of natural forests of northeastern India is high, even with a past disturbance history. The forests of both the landscapes are yet to be fully matured and therefore, have the potential to store additional carbon in the future and substantially contribute to the reduction of atmospheric CO_2.

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ORIGINAL PAPER

Estimation of tree biomass, carbon pool and net primary production of an old-growth *Pinus kesiya* Royle ex. Gordon forest in north-eastern India

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Abstract

- ***Background*** The data on carbon pool and biomass distribution pattern of old-growth *Pinus kesiya* Royle ex. Gordon forests are not available.
- ***Methods*** The forest carbon pool and annual net primary production (NPP) were assessed in three old-growth *P. kesiya* forest stands in north-eastern India, using biomass equations developed from 40 harvested trees between 9 and 63 cm in diameter at breast height (DBH) range.
- ***Results*** Regression models of the form $\mathrm{Log}(Y)=a + b\ \mathrm{log}D+c\ (\mathrm{log}D)^2+d\ (\mathrm{log}D)^3$ were the best fits for biomass estimation of total tree and its various components. The total forest biomass (which includes live and dead compartments of trees, shrubs, and herbs) was 460.5 Mg ha^{-1}, of which 91.2% was in the aboveground and 8.8% in the belowground compartment. *P. kesiya* contributed 77%, broad-leaved tree species 13.5%, shrubs 0.12%, herbs 0.03% and litter 0.5% to the total forest biomass. The total ecosystem carbon content of the forest including soil organic carbon pool was 283.1 Mg C ha^{-1}. The annual net primary production (NPP) of the forest was 17.5 Mg ha^{-1} yr^{-1}.
- ***Conclusion*** The estimated total forest biomass and carbon pool of the *P. kesiya* forest were greater than for the other pine forests studied world-wide.

Keywords Old-growth *Pinus kesiya* forest · Tree biomass estimation models · Total forest carbon pool · Net primary production

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1 Introduction

The carbon pool of a forest ecosystem varies with age (Clark et al. 2004; Kurz and Apps 1995). While young and middle-aged forest stands act as active carbon sinks (Valentini et al. 2000), old stands are moderate to small C sinks or even C sources, depending on the forest type and species composition (Desai et al. 2005; Knohl et al. 2003; Law et al. 2003; Malhi et al. 1999). However, most NPP studies world-wide have been carried out in relatively younger stands, and data on carbon content and NPP in old-growth pine forests are limited (Delrio et al. 2008).

Determination of carbon sequestration potential in terrestrial ecosystems through biomass estimation has been the most widely followed and appropriate approach (Brown 1997; Brown et al. 1989; Chambers et al. 2001). Regression models are used for biomass estimation because of their relative simplicity and ease for converting inventory data into a biomass estimate. Although it is difficult and tedious at the initial stage to develop the best-fit models, tree dimension values as the input data requirement for subsequent estimations have made the regression-based biomass estimation method extremely popular (Brown 1997). Several regression models have been developed to estimate biomass or biomass-related parameters (Brown et al. 1989; Schmidt et al. 2009), which are being used to prepare volume tables for several forestry species (Li and Weiskittel 2010) and to estimate carbon in tropical, temperate, boreal, and semi-arid forest ecosystems (Schroeder et al. 1997). The total biomass data obtained from such models are then converted into carbon content for estimating carbon pools in different compartments, by multiplying by a conversion factor of 0.5 on the assumption that the tree biomass contains 50% carbon (Ravindranath et al. 1997; Richter et al. 1995).

Although several workers have used tree height, trunk diameter [i.e., diameter at breast height (DBH)] and wood density as independent variables for estimating tree above-ground biomass (AGB), the allometric relationship between AGB and DBH has been proved to be the best fit for tree biomass estimation in several forests (Brown 1997; Brown et al. 1989). The carbon present in other compartments of the ecosystem such as shrub, herb, litter, woody debris, root, and soil is added to the tree carbon data to obtain the size of the total carbon pool in a forest ecosystem. Since AGB of trees contains a large fraction of the total forest carbon stock, most studies on forest carbon budget have focused only on tree AGB estimation. Although these studies do provide empirical data on the major carbon pool of the forest, total ecosystem level carbon data for most ecosystems is lacking.

Net primary productivity (NPP), the balance between the light energy fixed through photosynthesis and respiratory loss and mortality, represents the net carbon input from the atmosphere to terrestrial vegetation (Melillo et al. 1993). It is the net organic matter produced by live plants at the end of a specific time interval (Clark et al. 2001). It is an important index for estimating carbon budget and evaluating the patterns, processes and dynamics of carbon cycling in forest ecosystems at local, regional and global scales (Luo et al. 2002). Most studies on NPP estimation consider only the increment in AGB and litterfall, and completely ignore the belowground component. Cairns et al. (1997) argued that the approach of allometric modeling should be more realistic than root/shoot ratio for estimating tree belowground biomass (BGB). For estimating ecosystem level NPP of a forest, time-series biomass data for tree, shrub, herb, and litter components are pre-requisites. However, such data for different components other than tree are not available easily, and therefore studies on total ecosystem level NPP estimation are limited.

The natural forests of *Pinus kesiya* Royle ex. Gordon are found throughout north-eastern India at an elevation range of 800–2,000 m a.s.l., and extend up to the Philippines through Myanmar and Vietnam (Changala and Gibson 1984). Plantation forests of the species have been reported from such far-off places as Kenya, Zimbabwe, and Tanzania, indicating its global importance. *P. kesiya* has been invading the montane subtropical broad-leaved forest areas of north-eastern India once the primary broad-leaved forest species are cleared (Banik et al. 1996). Therefore, *P. kesiya* forest is very important for north-eastern India as a carbon sink. Although the AGB and NPP of a *P. kesiya* plantation forest were studied by Das and Ramakrishnan (1987) along an age series of 1–22 years through developing an allometric model, the model did not fit well to the old-growth forests. As such, the total as well as compartment-wise carbon pool, and NPP of natural old-growth *P. kesiya* forests have not been studied. Therefore, the present study was undertaken (i) to develop regression models for biomass estimation of *Pinus kesiya* and broad-leaved trees, and (ii) to estimate carbon pools in different compartments, and (iii) net primary productivity of an old-growth *Pinus kesiya* forest ecosystem.

2 Materials and methods

2.1 Study site and climate

The study was conducted in Riat Laban reserved forest (latitude 25°55′N, longitude 91°88′E, elevation 1,643 m a.s.l., area 2.0 sq. km), and its adjoining Laitkor (latitude 25°56′N, longitude 91°89′E, elevation 1,660 m a.s.l., area 3.2 sq. km) and Upper Shillong (latitude 25°56′N, longitude 91°85′E, elevation 1,655 m a.s.l., area 7.9 sq. km) community forests in Meghalaya during 2005–2007. The forest stands are natural, continuous, well-protected, 65–80 years old and are dominated by *Pinus kesiya*. The forest is classified as Assam sub-tropical pine forest (Champion and Seth 1968). The study site received an average annual rainfall of 2466.2 mm during the study period. The average monthly temperature varied from a maximum of 22.9°C in the month of July to a minimum of 1.8°C in December.

The density of *P. kesiya* contributed to 71–73% of the total tree density in the forest. *Lyonia ovalifolia* Hort., with density ranging between 6 and 159 trees ha^{-1}, was the dominant sub-canopy tree species in all the three stands. The other associated tree species were *Alnus nepalensis* D. Don, *Lithocarpus dealbatus* Rehder, *Lyonia ovalifolia* Hort., *Myrica esculenta* Buch.-Ham. ex D. Don, *Rhododendron arboretum* Sm., and *Schima wallichii* Choisy. The shrub layer was dominated by *Eupatorium adenophorum* Spreng. and *Lantana camara* L. Some of the herbaceous species were *Arundinella benghalensis* Druce, *Duchesnea indica* Focke, *Eupatorium riparium* Regel, *Gnaphalium luteo-album* L., *Imperata cylindrica* Beauv., *Paspalam dilatatum* Poir, *Plantago major* L., *Pouzolzia hirta* Hassk, *Potentilla fulgens* Wall. ex Hook., and *Rananculus scleratoides* Perf. ex Ovczinn. *Aeginetia indica* L. was the dominant herb on decomposing litter. *Smilax aspera* DC. and *S. ovalifolia* A. DC. were the dominant climbers in the forest.

2.2 Determination of forest age and analysis of tree population structure

Six permanent plots of 250 m × 20 m size were laid in the three stands. In each plot, all trees with ≥5 cm DBH were tagged, measured, and identified The girth of each individual tree was measured The tree species other than *P kesiya* were identified with the help of regional flora (Haridasan and Rao 1985–1987) The ASSAM herbarium at Botanical Survey of India, Shillong was consulted for confirmation The density and basal area were calculated following Misra (1968) For depicting tree population structure of the forest, all trees including broad-leaved species were grouped into eight diameter classes i e >5–9 9, 10–19 9, 20–29 9, 30–39 9, 40–49 9, 50–59 9, 60–69 9 and 70–79 9 cm The diameter–density distribution of *P kesiya* was presented

2 3 Estimation of total biomass and carbon of the forest

The total forest biomass was estimated by adding the biomass of the following components (i) *P kesiya* trees, (ii) broad-leaved trees, (iii) litter, and (iv) shrubs and herbs Since biomass models for *P kesiya*, particularly in old-growth forests, were not available, allometric biomass equations were developed for AGB and BGB estimation The existing biomass models for broad-leaved species (Cairns et al 1997, Chambers et al 2001) were used to estimate the BGB and AGB of broad-leaved trees in the forest The biomass of shrubs and herbs were directly estimated through a harvest method following Misra (1968) The mean biomass values calculated from the six permanent plots in the three sites were presented Carbon content of each component was calculated as 50% of the ash-free mass Ash content was determined by igniting the oven-dried plant materials at 550°C for 6 hours in a muffle furnace

2 4 Estimation of *P kesiya* tree biomass for model development

Forty trees of *P kesiya* were randomly selected for felling from the three forest stands The trees selected for the Riat Laban reserved forest stand were from the adjacent community forest area, which is continuous with the reserve, since felling of tree is banned inside the reserve The trees selected in the two community forest stands were from the peripheral areas of the stands Five to six trees were selected from each of the seven diameter classes of *P kesiya* i e >5–9 9, 10–19 9, 20–29 9, 30–39 9, 40–49 9, 50–59 9 and 60–69 9 cm, which represented the minimum and maximum diameter range of the species in the forest The DBH of the felled trees was measured The age of the pine forest was determined by counting the annual growth rings in circular sections taken from the above mentioned 40 sample trees The counting of the rings was done in sections taken at 30 cm from the base of the tree The mean value represented the age of the forest stand The trees were separated into stem, branch, twig, needle, reproductive part, and root components, and the fresh weight of each component was taken Three replicate samples of 2 kg each for each component were oven-dried at 80°C till constant weight was achieved For estimation of BGB, the roots of each cut tree were excavated as completely as possible, and separated into fine roots (<2 mm diameter) and coarse roots (>2 mm diameter) Both the coarse and fine roots of each cut tree were weighed in the field The portion of the tree stump that remains underground was treated as a part of the coarse root The root samples in triplicate were brought to the laboratory and oven-dried at 80°C till constant weight was achieved

2 5 Development and evaluation of allometric models

Regression models were developed considering tree DBH as independent variable, and stem, branch, twig, needle, reproductive part, root, total aboveground, and total tree biomass as dependent variables The DBH and dry weight values were log-transformed, and nonlinear regression models were fitted for different tree components, as well as for total tree biomass For selecting the best-fit models, the coefficient of determination (R^2), standard deviation (SD), sum of square error (SSE), mean square error (MSE) and root mean square error (RMSE) of the allometric equations were compared with those of existing models developed by earlier workers (Brown 1997, Delrio et al 2008, Ter-Mikaelian and Korzukhin 1997) for *Pinus* spp

The models developed by Chambers et al (2001) for AGB and Cairns et al (1997) for BGB estimation were used for determining broad-leaved tree biomass The allometric model for the aboveground component (Y_1) is $\ln(Y_1) = -0\ 37 + 0\ 333\ \ln D + 0\ 933\ [\ln(D)]^2 - 0\ 122\ [\ln(D)]^3$, and that for belowground component (Y_2) is $Y_2 = \mathrm{Exp}\ [-1\ 085 + 0\ 9256\ (\ln AGB)]$ These two models were selected based on R^2, SD, SSE, MSE, and RMSE values

2 6 Litter

Litterfall was estimated at monthly interval over a period of 2 years from September, 2005 to August, 2007 Five traps of 1 m × 1 m × 0 15 m (length × breadth × height) were placed within each permanent plot (*n*=120) The litter components were segregated into five fractions, viz, leaf, twig, branch, cone, and reproductive parts The biomass of each component was determined after oven-drying the samples at 80°C till constant weight was achieved

2 7 Analysis of soil organic carbon

Composite soil samples were collected from each of the six permanent plots up to 1 m depth from the surface Soil samples were collected during each of the four seasons of the year 2005, and analysis for determination of soil organic carbon content was done after air-drying and sieving the soil samples through a 2-mm mesh sieve. Soil organic carbon was estimated following the colorimetric method described by Anderson and Ingram (1993).

2.8 Estimation of NPP

The NPP of the forest was determined from the NPP estimates for each component, i.e. tree, shrub, herb, and litter in the six permanent sample plots. The NPP was estimated for all these components for 2 consecutive years (i.e., 2006 and 2007), and the mean values were presented. The biomass for *P. kesiya* was estimated by applying the allometric equations developed in this study, and for other broad-leaved tree species it was estimated using the equations of Chambers et al. (2001) for AGB and Cairns et al. (1997) for BGB. The standing tree biomass component of NPP was estimated by subtracting biomass estimated for September 2005 from that of August 2006, and biomass for September 2006 from that of August 2007, for the years 2006 and 2007 respectively. The aboveground NPP was determined by summing the tree biomass component of NPP and annual litter production measured at the same time interval (Kira and Shidei 1967). The annual root production was measured by sampling roots using a soil augur in four seasons each year. The roots were washed and segregated into fine and coarse roots and the biomass was determined for each component after oven-drying the samples in 80°C till constant weight was achieved. The annual root production was measured by summing up the positive increments in live root biomass and concurrent positive increment in the dead root biomass during the successive samplings (Persson 1978). The NPP for shrubs and herbs was estimated using the biomass data for the same time interval as standing tree biomass component.

3 Results

3.1 Allometric biomass models for *P. kesiya*

The aboveground biomass data for 40 trees (Table 1) were regressed against the DBH using the regression models developed by the earlier workers for pine species. None of these models yielded a satisfactory coefficient of determination (R^2), MSE and RMSE. Hence, the following form of model was developed by log transforming the data of each tree component, which yielded greater R^2 and lower MSE, SSE and RMSE than the earlier models —

$$Log(Y) = a + b\log D + c(\log D)^2 + d(\log D)^3 \qquad (1)$$

where Y=AGB (kg/tree), a, b, c, and d are regression coefficients, and D is the stem diameter at breast height. All the measures of coefficients of the models for tree components were statistically significant ($P<0.001$) (Fig. 1).

3.2 Density of trees, shrubs and herbs

The density of trees in the forest ranged between 628 and 947 trees ha^{-1}. The density of *P. kesiya* ranged between 454 and 677 trees ha^{-1}. The total tree basal areas of the three forest stands were 70.6, 66 and 126 m^2 ha^{-1} respectively. *P. kesiya* contributed to 97.4, 96.5 and 74% of the total basal area in the three stands. The total shrub density of the forest was 4,400 plants ha^{-1}, and that for herb was 385,000 plants ha^{-1}.

3.3 Estimation of total forest biomass and NPP

The total biomass of the forest was 460.5 Mg ha^{-1}, of which 91.2% was in the aboveground compartment and 8.8% in the belowground compartment. *P. kesiya* contributed 77%, broad-leaved tree species 13.5%, shrubs 0.12%, herbs 0.03% and litter 0.5% to the total forest biomass (Table 2). The total AGB of pine including litter, herb and shrub components was 357.6 Mg ha^{-1}. The tree AGB and BGB were 419.7 and 40.8 Mg ha^{-1} respectively. While *P. kesiya* had 354.6 Mg ha^{-1} as AGB and 31.8 Mg ha^{-1} as BGB, the corresponding figures for the broad-leaved species were only 62.2 and 9.0 Mg ha^{-1} respectively (Table 2). The needles, twigs, branches, and reproductive parts accounted for 44.9, 31.3, 4.8, and 18.9% respectively to the total litterfall. The fine and coarse root biomass of *P. kesiya* were 0.5 and 6.4% of the total forest biomass and constituted 6% and 72% of the total BGB respectively. The BGB of broad-leaved species was only 1.9% of the total forest ecosystem biomass and 22% of the total BGB. The total ecosystem NPP of the forest was 17.5 Mg ha^{-1} yr^{-1}. The leaf litter, twig, branch, and reproductive parts contributed to 42.6, 14.9, 10.6, and 31.9% of the total litter production. The stem (55%), branches (8.6%), and twigs, needles, and reproductive parts (8.6%) of *P. kesiya* were the major contributors to the total ecosystem NPP. The total ecosystem carbon content of the forest was 283.1 Mg C ha^{-1} (Table 2). The soil organic carbon was 58.7 Mg ha^{-1}, contributing 20.7% to the total ecosystem carbon. The DBH class 50–59.9 cm had the highest tree density among all the DBH classes, and hence contributed maximum biomass of 115.9 Mg ha^{-1}, which accounted for 27.8 % of the total AGB of the forest (Fig. 2).

4 Discussion

In most allometric models that relate biomass with DBH, the variability in untransformed data increases with increase

Table 1 Dry weight (kg) of different components of *Pinus kesiya* tree used for developing regression models

DBH (cm)	Stem	Branch	Twig	Needle	Cone	Fine root	Coarse root	Total root	Total AGB	Total tree
9 01	15 18	4 99	3 27	1 53	0 42	0 31	2 22	2 53	25 38	27 90
9 20	15 57	6 28	3 83	1 97	0 44	0 98	2 54	3 52	28 08	31 60
11 28	19 61	8 13	3 96	1 88	0 90	0 26	3 18	3 44	34 46	37 90
11 41	20 12	8 51	3 40	2 00	0 97	0 29	3 61	3 90	35 01	38 91
12 37	22 19	9 72	3 53	2 13	0 82	0 55	6 76	7 31	38 38	45 69
13 94	32 51	12 99	3 27	4 97	2 22	0 95	11 65	12 61	55 96	68 56
14 33	48 25	14 02	3 40	4 66	2 38	1 05	12 81	13 85	72 70	86 55
15 96	41 11	19 61	4 56	5 49	2 33	1 43	17 50	18 93	73 09	92 02
16 12	63 08	23 91	5 42	4 76	2 60	1 46	17 94	19 41	99 76	119 17
17 12	81 36	28 34	6 71	5 34	2 51	1 68	20 62	22 31	124 24	146 55
18 11	88 84	35 52	8 15	5 01	3 25	1 89	23 19	25 08	140 77	165 85
19 29	72 93	22 49	9 72	6 67	2 58	2 13	26 10	28 23	114 39	142 62
22 60	121 52	37 20	8 28	5 40	3 27	2 73	33 40	36 13	175 67	211 80
23 15	133 86	48 29	9 61	6 12	3 07	2 82	34 52	37 33	200 94	238 28
27 76	287 37	83 95	10 99	6 48	2 66	3 47	42 60	46 08	391 44	437 52
28 59	287 37	83 91	10 99	6 48	3 37	3 58	43 85	47 43	392 12	439 55
32 76	354 54	86 06	11 72	6 41	3 27	4 01	49 22	53 23	462 01	515 24
35 03	376 52	80 57	11 52	6 22	5 83	4 20	51 56	55 76	480 66	536 42
37 12	425 53	82 26	11 87	6 55	7 20	4 35	53 38	57 73	533 40	591 13
38 11	478 43	84 99	11 33	7 29	4 77	4 41	54 14	58 55	586 80	645 35
39 29	502 51	90 58	14 06	8 15	5 16	4 48	54 96	59 44	620 46	679 89
40 61	578 48	88 92	14 02	7 38	3 95	4 54	55 77	60 31	692 75	753 07
43 47	640 42	93 03	13 91	7 52	4 65	4 66	57 19	61 84	759 54	821 39
47 53	722 25	105 93	15 20	8 38	9 88	4 76	58 48	63 24	861 65	924 89
48 27	730 40	101 70	15 57	7 09	5 15	4 77	58 63	63 40	859 90	923 30
49 23	739 84	106 12	14 04	8 16	8 38	4 78	58 80	63 58	876 53	940 11
49 74	750 93	102 00	13 14	7 92	6 39	4 79	58 87	63 66	880 38	944 04
51 41	769 36	110 38	15 70	8 43	4 18	4 80	59 05	63 86	908 04	971 89
52 95	799 52	114 25	15 10	8 37	6 39	4 81	59 14	63 94	943 63	1,007 57
53 59	815 11	105 93	17 01	9 12	7 13	4 81	59 15	63 96	954 31	1,018 27
54 58	836 39	124 40	16 94	7 57	7 88	4 81	59 15	63 95	993 18	1,057 14
56 70	961 72	115 52	13 81	7 71	5 62	4 80	59 06	63 86	1,104 37	1,168 23
57 44	1,000 33	114 43	15 63	8 46	6 65	4 79	59 01	63 81	1,145 50	1,209 31
59 49	1,046 02	93 03	14 04	8 16	8 77	4 78	58 82	63 59	1,170 02	1,233 61
59 90	1,056 21	101 31	13 14	8 38	9 88	4 77	58 77	63 54	1,188 93	1,252 47
60 10	1,080 29	105 93	15 70	7 09	5 15	4 77	58 75	63 52	1,214 16	1,277 68
61 28	1,085 15	101 70	15 10	8 16	8 38	4 76	58 60	63 36	1,218 48	1,281 84
61 28	1,104 67	106 12	17 01	7 92	6 39	4 76	58 60	63 36	1,242 10	1,305 45
62 37	1,142 38	102 00	16 94	8 43	4 18	4 74	58 45	63 20	1,273 93	1,337 13
62 69	1 184 05	110 38	13 81	8 37	6 39	4 74	58 41	63 15	1,323 00	1,386 14

in diameter (Beauchamp and Olson 1973) The log transformation brings the variance down to uniformity by stretching the smaller values and compressing the larger values. The actual biomass values are obtained from the model-derived values through antilog transformation (Ovington and Olson 1970) The log-transformed DBH data of the harvested trees used for the model development yielded lower variance (0 074), standard deviation (0 272) and standard error (0 043) than the untransformed DBH The variance (0 056–0 076), standard deviation (0.238–0 275) and standard error (0 008–0 009) of the log-transformed DBH data in the three pine forest stands, used for estimating total forest biomass, were also low Several authors (Baskerville 1972, Beauchamp and Olson 1973)

Fig. 1 Regression analyses between tree diameter at breast height (cm) and biomass of different tree components above ground biomass and total tree biomass of *Pinus kesiya*

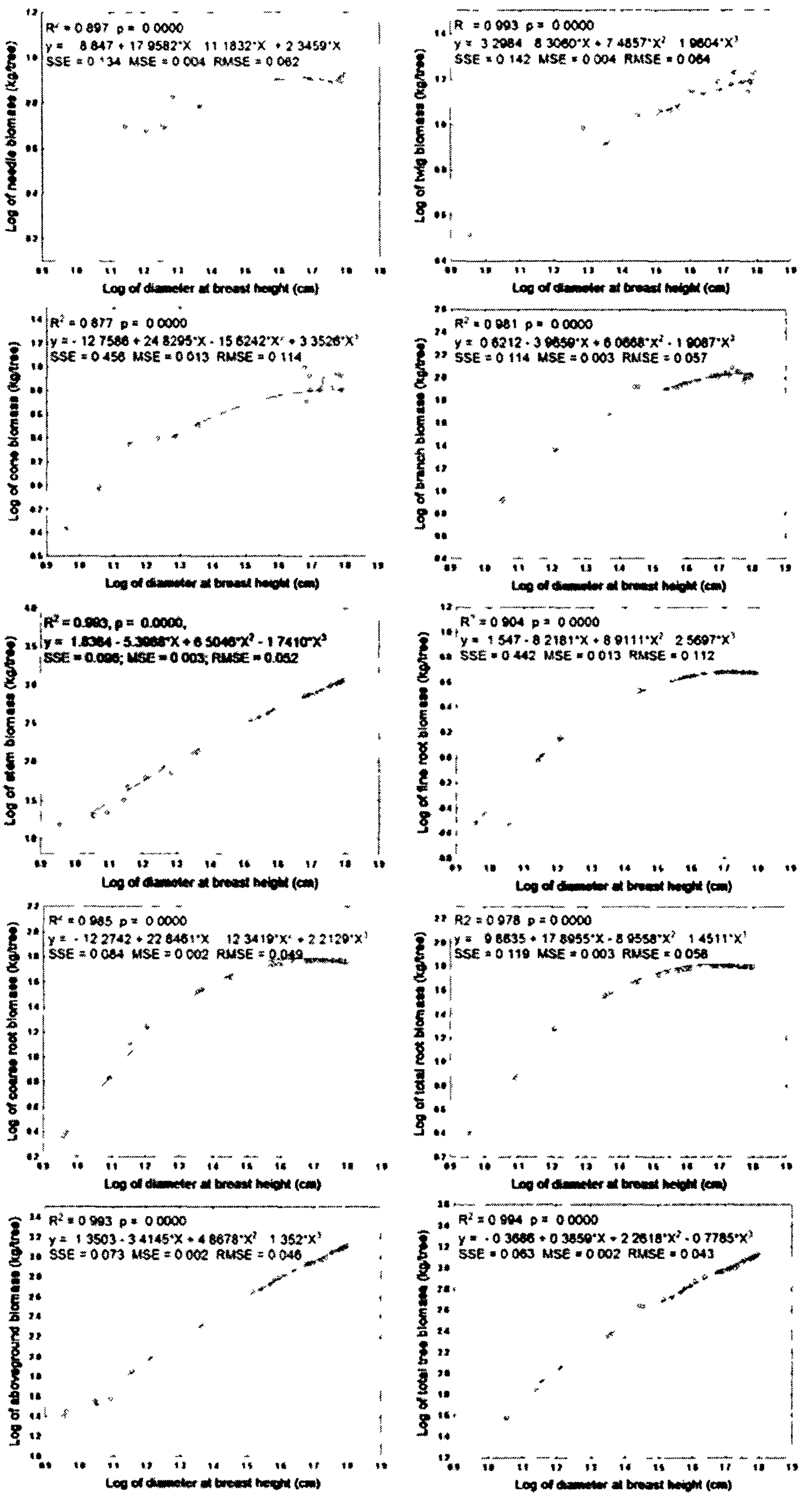

Table 2 Total ecosystem, above- and belowground biomass, carbon content and net primary production of an old-growth *Pinus kesiya* forest in north-eastern India

Component	Biomass ($Mg\ ha^{-1}$)	Carbon ($Mg\ C\ ha^{-1}$)	% contribution	Net primary production ($Mg\ ha^{-1} yr^{-1}$)	% contribution
Aboveground biomass of *Pinus kesiya*					
- Stem	291 8±33 0	143 0	63 4	9 8	56 0
- Branch	48 2+2 0	23 6	10 5	1 5	8 6
- Twig	7 3±0 5	3 6	1 6	0 7	4 0
- Needle	4 3±0 4	2 1	0 9	0 5	2 9
- Reproductive parts	3 1+0 2	1 5	0 7	0 3	1 7
Aboveground biomass of broad-leaved species	62 2±40 3	30 5	13 5	3 2	18 3
Herbs	0 1+2 5	0 1	0 03	0 0	0 0
Shrubs	0 6+4 6	0 3	0 12	0 0	0 0
Detrital biomass					
- Needle	1 0+1 6	0 5	0 2	0 0	0 2
- Twig	0 7±2 4	0 3	0 2	0 0	0 1
- Branch	0 1±1 4	0 1	0 0	0 0	0 1
- Reproductive parts	0 4±2 2	0 2	0 1	0 0	0 2
- Total detrital biomass	2 3±7 6	1 1	0 5	0 1	0 5
Total aboveground biomass	419 7	205 7	91 2	16 1	92 0
Belowground biomass of *Pinus kesiya*					
- Fine roots (<2 mm)	2 4+4 7	1 1	0 5	0 3	1 7
- Coarse roots (>2 mm)	29 4±7 2	13 5	6 4	0 7	4 1
Belowground biomass of others species	9.0±1.7	4 1	1 9	0 4	2 2
- Total forest	460 5	224 4	100 0		
- Total soil organic carbon		58 7			
Total ecosystem	460 5	283 1	100 0	17 5	100 0
BNPP				0 1	8 0
ANPP				0 9	92 0

have argued that log-transformed nonlinear regression models produce up to 20% error for certain components if they are back-transformed into linear form. The back transformation is acceptable if MSE is not too large (Baskerville 1972). In this study, the MSE and RMSE values for different tree components are extremely low.

The AGB in this old-growth *Pinus kesiya* forest (419.7 Mg ha^{-1}) was about 36% greater than that of the 22-year-old young forest studied by Das and Ramakrishnan (1987) (308.7 Mg ha^{-1}). Increase in AGB with age was also depicted through the studies conducted by Ovington and Madgwick (1959), and Delrio et al. (2008) for *Pinus sylvestris* L. for young- and old-growth forests respectively. A similar increase was also noted from the studies of Karizumi (1974) and Tanabe et al. (2003) for *Pinus densiflora* Sieb et Zucc. The AGB obtained in this study is comparable with a 41–80-year-old *P. sylvestris* forest in Spain (359.7–456.9 Mg ha^{-1}) (Delrio et al. 2008). However, it is much greater than that of a 71–80-year-old *Pinus koraiensis* Sieb. et Zucc. forest of Japan (317.9 Mg ha^{-1}) (Son et al. 2001). The AGB obtained in the present study was also greater than that of most tropical forests studied. Muller (1982) obtained an AGB of 330 Mg ha^{-1} for the tropical broad-leaved forests of the eastern hardwood region of USA, and Brown et al (1989) reported 238–341 Mg ha^{-1} for Cameroon and 153–221 Mg ha^{-1} for Sri Lanka. However, the AGB of the present study is comparable with the findings of Brown et al. (1989) for the tropical rain forests of Malaysia (225–446 Mg ha^{-1}).

The 50–60 cm tree diameter class contributed 27.8% to the total tree AGB, indicating the important role of this diameter class in carbon storage The larger trees (>60 cm DBH) contributed 15.8% to the total aboveground biomass. Thus, the large trees together accounted for more than 43% of the total carbon in the tree component. The greater contribution of large trees to AGB is in conformity with the findings of earlier workers (Baishya et al 2009; Brown 1996; Brown and Lugo 1992), who had found that large

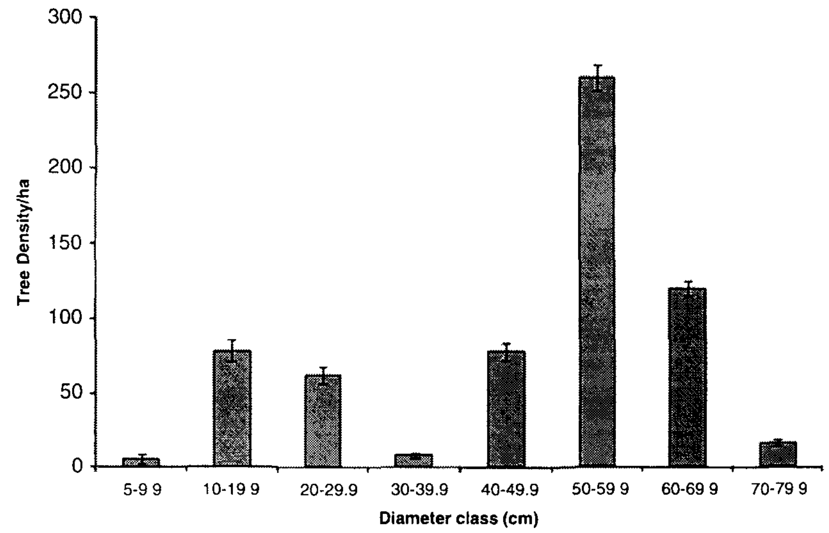

Fig. 2 Diameter–density distribution of *Pinus kesiya* in an old-growth pine forest of north-eastern India. *Bars* show standard error

trees contributed up to 50% of the aboveground biomass. The smaller diameter trees thus had about 57% of the total tree carbon in this forest, indicating their importance in carbon storage.

The annual litterfall in this old-growth *P. kesiya* forest (2.3 Mg ha^{-1}) was greater than that in the 22-year-old forest (1.5 Mg ha^{-1}) reported by Arunachalam et al. (1996). The amount of annual litter fall is within the reported range of 2.2–22.6 Mg ha^{-1} for various tropical and subtropical forests (Vogt et al. 1986). The proportion of leaf litter in the total litter was 42.6%, and the twigs, branches, and reproductive parts accounted for 14.9%, 10.6%, and 31.9% respectively. Arunachalam et al. (1996) reported that leaf litter constituted as high as 75% of the total litter in the young 22-year-old *P. kesiya* forest stand. Therefore, it is evident that with increasing age of a *P. kesiya* forest, the relative proportion of leaf litter in the litter decreases.

Fine and coarse root biomass of *P. kesiya* contributed 6% and 72% to the total root biomass respectively. The fine root biomass was lower (2.4 Mg ha^{-1}) than the values reported by Arunachalam et al. (1996) (3.4–5.1 Mg ha^{-1}) and by John et al. (2001) (4.6 Mg ha^{-1}) in a 22-year-old and 23-year-old *P. kesiya* plantation forest, indicating the reduction of fine root production in old-growth forest. The fine root biomass obtained in the present study was also lower than the global range of 1.0–17.7 Mg ha^{-1} for various ecosystems (Vogt et al. 1986). Low availability of soil nutrients and water has been reported to promote high production and accumulation of fine roots (Vogt et al. 1986).

The BGB in this forest was 8.9% of the total tree biomass, which is within the range reported by earlier workers. Cairns et al. (1997) concluded that the BGB of an ecosystem can reach up to 25% of the total tree biomass. The contribution of shrub and herb components to the total forest biomass was negligible (0.15%). Brown (1997) concluded that shrubs and herbs can contribute up to 3% of the total forest AGB.

The total NPP of the forest was high, which may be attributed to relatively higher soil NPK content and moisture regime in this pine forest. Gower et al. (1994) reported that N fertilization increases NPP of forest, and carbon allocation to belowground components decreases with increase in soil N availability (Haynes and Gower 1995). The extremely low BGB (40.8 Mg ha^{-1}) obtained in this study may be attributed to this reason. Vogt et al. (1996) reported that root NPP does not depend only on nutrient availability for many pine species. Relatively low root NPP as observed in this study (1.4 Mg ha^{-1} yr^{-1}) in comparison to other studies (Karizumi 1974) supports this argument.

The total aboveground NPP in this old-growth forest (16.1 Mg ha^{-1} yr^{-1}) was lower than the younger (5–22-year-old) *P. kesiya* forest (30.1–20.1 Mg ha^{-1} yr^{-1}) reported by Das and Ramakrishnan (1987). The total aboveground NPP was higher than most pine forests around the world. For example, Chaturvedi and Singh (1982, 1987) and Rana et al. (1989) calculated the aboveground NPP for *Pinus roxburghii* forest (6.1–15.6 Mg ha^{-1} yr^{-1}) in the central Himalayas of India. Ma (1988) reported NPP of 3.5–17.5 Mg ha^{-1} yr^{-1} for 33–70-year-old *Pinus tabulaeformis* forest from China. The high productivity of *P. kesiya* may be attributed to high net assimilation rate due to prolonged photosynthetic activity, and higher uptake of nutrients due to rapid turnover of nutrients (Das and Ramakrishnan 1987). The potential of a forest to sequester carbon depends on the forest type, age of forest and size class of trees (Terakunpisut et al. 2007). Considering the NPP level, tree diameter distribution, species composition and the age of the forest, it can be concluded that the forest is yet to fully mature and has the potential to store additional carbon in the future.

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Distribution pattern of aboveground biomass in natural and plantation forests of humid tropics in northeast India

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Abstract: Tree aboveground biomass (AGB) distribution and carbon storage in different DBH (*diameter at breast height*) *classes were compared between natural semi-evergreen forest* and sal plantation forest in the humid tropical region of northeast India. The natural forest had lower AGB (323.9 Mg ha^{-1}) than the plantation forest (406.4 Mg ha^{-1}). About 49% of the AGB was present in > 60 cm dbh trees in the natural forest against 24% in the plantation forest. The carbon storage was highest in 60-80 cm and 40-60 cm dbh classes in the natural forest and plantation forest, respectively. The differential AGB and carbon distribution pattern has been related to past disturbance history and age of the forests. Although both the forests had potential for carbon sequestration due to presence of large number of trees belonging to small dbh classes, the plantation forest had an edge over the natural forest because of better silvicultural practices.

Resumen: La distribución de la biomasa arbórea aérea (BAA) y los almacenes de carbono en diferentes categorías de DAP (diámetro a la altura del pecho) fueron comparados entre un bosque natural subperennifolio y una plantación de sal en la región tropical húmeda del nordeste de la India. El bosque natural tuvo una BAA menor (323.9 Mg ha^{-1}) que la plantación (406.4 Mg ha^{-1}). Alrededor de 49% de la BAA se concentró en árboles con DAP > 60 cm en el bosque natural, contra 24% en la plantación. Los almacenes de carbono fueron más grandes en las categorías diamétricas de 60-80 cm y de 40-60 en el bosque natural y la plantación, respectivamente. El patrón diferencial en la distribución de BAA y de carbono estuvo relacionado con la historia de disturbio y la edad de los bosques. Si bien ambos bosques tienen el potencial de secuestrar carbono debido a la presencia de un gran número de árboles en las categorías diamétricas pequeñas, la plantación está en ventaja sobre el bosque natural debido a la aplicación de mejores prácticas silvícolas.

Resumo: A distribuição da biomassa arbórea aérea (AGB) e fixação de carbono para diferentes classes de DAP (diâmetro à altura do peito) foram comparadas entre a floresta natural semi-sempreverde e plantações florestais de "meranti" na região tropical húmida do nordeste da Índia. A floresta natural apresenta mais baixo AGB (323,9 Mg ha^{-1}) do que a floresta plantada (406, 4 Mg ha^{-1}). Cerca de 49% do AGB encontrava-se em árvores com DAP > 60 cm na floresta natural contra 24% na floresta plantada. A fixação de carbono era a mais alta nas árvores das classes de DAP de 60-80 cm e 40-60 cm na floresta natural e plantada, *respectivamente. O padrão diferencial de distribuição do AGB e carbono foi relacionado com a* história de perturbação e a idade das florestas. Embora ambas as florestas tenham o potencial para o sequestro de carbono devido à presença de um grande número de árvores pertencente às pequenas classes de DAP, a floresta plantada tem uma vantagem sobre a natural por causa das melhores práticas silvícolas.

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Key words Aboveground biomass, carbon storage, northeast India, sal plantation forest, silvicultural practices, tropical semi-evergreen forest

Introduction

The tropical forests spread over 13 76 million sq km area worldwide account for 60% of the global forests (FAO 1988, 2005) and play a key role in global C cycle both in terms of C flux and the volume of C stored The significant influence of tropical forests on carbon cycle is attributed to the high rate of primary production besides the large pool and flux sizes (Brown & Lugo 1982, 1984) Because of higher net productivity, the tropical forests are more effective in carbon sequestration than any other forests (Brown *et al* 1989, Soni 2003) The tropical forests store large quantities of carbon in vegetation and soil, exchange carbon with the atmosphere through photosynthesis and respiration These forests account for 37% of the total 90% of the world's terrestrial C that is stored in forests (Houghton 1996) Very few tropical forests are at their maximum potential level of biomass density because of prevailing or past cultural disturbances, and therefore, have a larger additional carbon sequestration capacity and have the potential to increase the global carbon store beyond the present value (Iverson *et al* 1993) Consequently tropical forests have attracted a great deal of experimental and theoretical attention in recent years (Malhi *et al* 1999, Malhi & Grace 2000)

The Amazonian tropical forests have the highest carbon sequestration as well as release among all the ecosystems, because of large area and high productivity associated with large scale deforestation, burning, and fast rate of decomposition and landuse changes (Fearnside 1997) The tropical forests act as sources of atmospheric carbon if disturbed by anthropogenic activities or natural calamities However, they become atmospheric carbon sinks during land abandonment, forest regrowth after disturbance and due to afforestation, reforestation and forest conservation Therefore, the role of tropical forests as overall CO_2 sink or source is scale-dependent and site-specific The need for generating data at high spatial resolution for both above- and belowground phytomass, soil and other pools and fluxes of C has been emphasized by several workers for improving quantification of global C pools and fluxes (Chhabra & Dadhwal 2004)

As a mitigation measure to global climate change due to greenhouse gas emission, it is required to cut the rate of emission either through reducing tropical deforestation or to enhance the natural carbon sequestration potential of degraded forests through forest regeneration and afforestation The degraded areas have a large potential to sequester carbon in the soil, storage in vegetation is preferable due to their longer residence time and less risk of rapid release to the atmosphere (Lal 2001) This can only be achieved through afforestation or reforestation of such areas The protection of existing forests, regeneration of degraded forests and raising of forest plantations in India have been contributing to enhanced carbon stock (Ravindranath *et al* 2008)

The major C pools such as phytomass, soil, litter, and fluxes of C due to litterfall and landuse changes have been estimated for India based on very coarse resolution data and extrapolation, as the primary data for many regions of the country are either non-existent or over-estimated (Dadhwal & Nayak 1993) The data available on carbon sequestration i e net woody biomass accumulation in trees for long term storage in tropical forests are extremely limited and incomplete Because of the lack of reliable data on standing biomass and rates of forest degradation, the net annual carbon emission estimates for India have also been highly variable (Ravindranath *et al* 1997) Thus, the improved quantification of C pools and fluxes in tropical forest ecosystems is important for understanding the contribution of these forests to net C emissions and their potential for carbon sequestration (Chhabra & Dadhwal 2004)

The potential of tropical forests for increased carbon sequestration capability can be assessed either through the amount of carbon stored or estimating the annual carbon sequestration rate (Iverson *et al* 1993) The studies on carbon sequestration have been focusing on and expressing the sequestration in terms of biomass and carbon stock. The design and evaluation of global scale carbon models require field estimates of forest biomass. Among the phytomass components i.e. aboveground, belowground and dead wood, the live aboveground wood biomass is the most important because it is involved in the regulation of atmospheric carbon concentration and constitutes about 60% of total phytomass. Therefore, estimation of aboveground biomass (AGB) is the most important aspect of studies of carbon sequestration (Ketterings *et al.* 2001). Estimation of AGB is also a useful measure for comparing structural and functional attributes of forest ecosystems across a wide range of environmental conditions (Brown *et al.* 1999).

The most reliable technique for estimating forest carbon stock is through forest inventories followed by developing allometric relationships between the aboveground biomass (AGB) of a tree and its trunk diameter (Brown 1997, Brown *et al.* 1989, Clark *et al.* 2001). The potential changes in the other carbon pools of the ecosystem such as litter, coarse woody debris, root biomass and soil organic matter are subsequently added to the estimations to obtain the size of the total tree carbon pool. Because AGB represents a large fraction of total forest carbon stock, its estimation offers a practical and reliable way of evaluating the carbon balance of tropical forest.

Several studies have concluded that matured tropical forests with high AGB, contain a large proportion of their aboveground biomass in large trees (Brown *et al.* 1995, Brown & Lugo 1992, Clark & Clark 1996). In contrast, several other workers have argued that old growth forests have less potential for carbon sequestration as the constituent older trees cease to grow (Terakunpisut *et al.* 2007). Similarly, in absence of precise data, it is believed that the potential for carbon sequestration of natural forests is manifold greater than the plantation forests. Since most tropical forests are now affected by one or the other form of human interventions, the density-diameter distribution of trees would be an important determinant of the carbon stock in these forests. It is not clear whether (i) carbon sequestration potential is greater in natural forests than the plantation forests, and (ii) young trees have greater carbon sequestration potential than the old trees in a natural forest.

The north eastern region of India with 99,260 km^2 of tropical forests (Roy & Joshi 2002) that spread up to an elevation of 900 m asl in the Eastern Himalayas and sub-Himalayan areas offers appropriate situation for examining the above two questions. The forests of the region include undisturbed evergreen and semi evergreen forests, secondary forests developed following shifting cultivation and forest degradation, and plantation forests raised by various agencies. Baring a few pockets of undisturbed forests, most tropical forests of the region were affected by one or the other form of cultural disturbances. Therefore, the present work aims to quantify carbon sequestration through estimation of aboveground carbon storage and AGB distribution pattern in different diameter classes in a natural and a plantation forest. The study would not only provide the distribution pattern of carbon in different tree size classes but would also fill the AGB data gap for the tropical forests of northeast India.

Materials and methods

Site description

The study was conducted in two forests *viz.*, tropical semi-evergreen forest and sal (*Shorea robusta*) plantation forest. These two forest types were selected due to the fact that tropical semi-evergreen forests constitute the majority of natural tropical forests of northeast India and sal plantation forests dominate the planted forest in the region. The two forests are part of humid tropical forest of Nongkhyllem wildlife sanctuary in Meghalaya, northeast India. The experimental plots lie between 25°55.578' - 25°56.062' N latitude and 091°46.453' - 91°46.546'E longitude. The area was declared as a wildlife sanctuary in 1981 prior to which it was part of the reserve forest. It covers 29 km^2 area on a steep hill slope (20° to > 65°) with an elevation ranging from 208 to 295 m and is 79 km from Shillong, the state capital of Meghalaya. The sal plantation was created during the year 1972 and is well-protected. On the other hand, the natural forest is an old growth forest, where the past disturbance in the form of selective logging took place at least 80 years ago prior to its declaration as a wildlife sanctuary. The forest is currently undisturbed and is characterized by dense canopy. Such forests constitute about 21% of the total sanctuary area.

Climate

The climate of the area is monsoonic with distinct warm-wet (May-October) and cold-dry (December-February) periods. The rainy season starts from May and extends up to October. About 90% of the total annual rainfall occurs during this period. The meteorological data were collected from the meteorological station of District Horticulture office, Government of Meghalaya located at Nongpoh, about 10 km from the study site. The total annual rainfall was 1355 mm during the year 2006-2007. The mean maximum temperature of 35°C was recorded in August and mean minimum temperature of 14°C was recorded in January.

Forest inventory

The different types of forests were mapped using IRS LISS III imageries through supervised classification and a forest type map within the wildlife sanctuary was prepared (Fig.1a). Sampling of vegetation in the two forests was carried out by belt transect method. Since the natural and plantation forests were continuous and each forest type was represented by a clearly demarcable single patch, an area of 10 ha was demarcated in each forest and was divided into four quarters of 250 m x 100 m each. In each quarter, a transect of 250 m x 20 m was laid laterally (Fig.1b) (Adapted from Alves *et al.* 1997). Thus four transects with a total 2 ha area were laid in each forest that was considered adequate for sampling, given the structure and size of each forest type. The transects were further divided into 10 x 10 m plots. In each plot, all individuals > 5 cm dbh (diameter at breast height) were tagged, measured and identified. The girth and height of each individual were measured. The plant specimens were identified with the help of regional floras (Balakrishnan 1981-83; Haridasan & Rao 1985-87; Joseph 1982; Kanjilal *et al.* 1934-40). The herbarium at Botanical Survey of India, North-Eastern Circle, Shillong and North-Eastern Hill University, Shillong, were consulted for confirmation. Frequency, density, basal area and importance value index (IVI) were calculated following Misra (1968) and Muller-Dombois & Ellenberg (1974). Trees were grouped into six dbh classes i.e. >5-20 cm, 20-40 cm, 40-60 cm, 60-80 cm, 80-100 cm and > 100 cm, and the density and AGB distribution under each dbh class were analyzed.

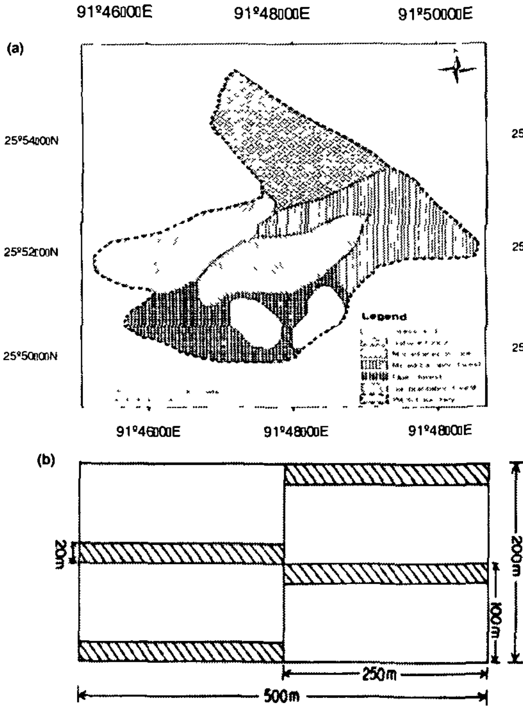

Fig. 1. (a) Map showing different forest types in Nongkhyllem wildlife sanctuary, Meghalaya, northeast India prepared based on IRS LISS III 2006 imagery and (b) diagrammatic presentation of the sampling procedure.

Biomass regression equations

Because of high species richness in tropical forests, it is difficult to use species-specific regression models, as used in the temperate zone (Brown & Schroeder 1999; Shepashenko *et al.* 1998; Ter-Mikaelian & Korzukhin 1997). Therefore, mixed species tree biomass regression models were used for AGB estimation of natural and plantation forests. The regression models for tree biomass used in the studies of Brown (1997), Brown *et al.* (1989), Chambers *et al.* (2001), Chave *et al.* (2001) and Brown & Iverson (1992) were evaluated based on prediction errors, residual analysis, and logical behavior of the models, R^2, and simplicity of the models (Table 1). Diameter at breast height i.e. at 1.37 m above the ground level, and individual tree height were measured during the study using diameter tape and clinometer These two parameters were used as inputs to the ten models and the best fit model was selected The model Y= exp [-0 37 + 0 33 ln(D) + 0 933 ln(D)^2 0 122 ln(D)^3] developed by Chambers *et al* (2001) was used to estimate the aboveground biomass in the two forests The best R^2 values of 0 93 and 0 91 were obtained for the natural forest and plantation forest, respectively

Estimation of AGB

The AGB in different tree diameter classes in each forest was estimated using the above mentioned model, where the variable biomass was AGB The aboveground biomass for all diameter classes was summed to arrive at the total aboveground biomass in each of the two forests The values for the natural and the plantation forests were compared using t test

Estimation of carbon

The aboveground biomass carbon stock was calculated by assuming that the carbon content is 50% of the total aboveground biomass (Brown & Lugo 1982, Cannel *et al* 1995, Dixon 1994, Ravindranath *et al* 1997, Richter *et al* 1995, Schroeder 1992)

Results

Stand characteristics

Ninety four species were recorded from the natural forest, and 67 species were recorded from the plantation forest The density of woody species (>5cm dbh) was greater in plantation forest (1028 trees ha ^{1}) than the natural forest (996 trees ha ^{1}) Based on density, *Antidesma acuminatum* (120 trees ha ^{1}) and *Polyalthia jenkinsii* (108 trees ha ^{1}) were the dominant species in the natural forest and these two species accounted for 21% of the total stand density In plantation forest, *Shorea robusta* (386 trees ha ^{1}) and *Litsea monopetala* (102 trees ha ^{1}) were the dominant and co-dominant species, respectively The basal area was greater in the plantation forest (89 92 m^{2}ha ^{1}) than the natural forest (73 41 m^{2}ha ^{1}) (Table 2)

Aboveground biomass distribution

Although the young individuals belonging to 5-20 cm dbh class dominated both forests in terms of density (Fig 2), the AGB accumulation was greater in the 40-60 cm diameter class in plantation forest and in 60-80 cm dbh class in natural forest (Fig 3) The AGB contribution by subsequent higher diameter classes got reduced in both the forests The contribution of trees having >60 cm diameter to AGB was greater in natural forest (6 6%) than the plantation forest (5 6%)

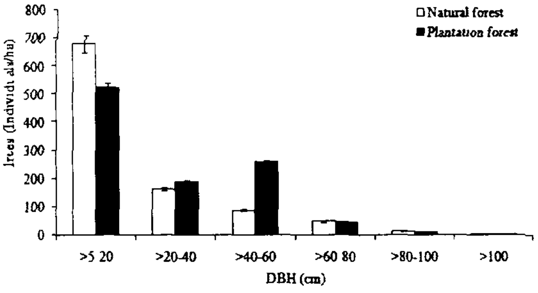

Fig. 2. Tree density in different diameter classes in natural and plantation forests of northeast India

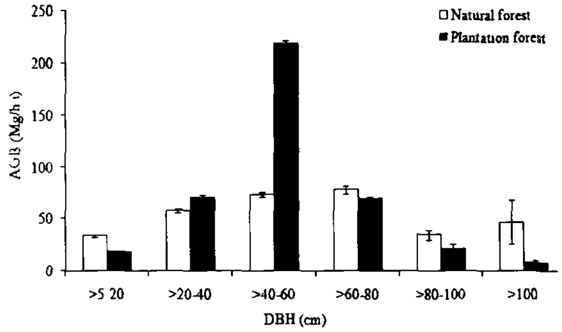

Fig. 3. Aboveground biomass in different tree diameter classes in natural and plantation forests of northeast India

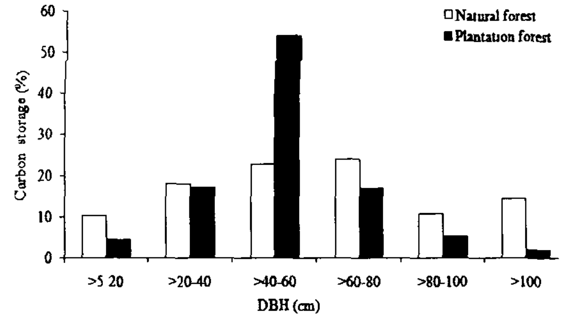

Fig. 4. Carbon storage in different tree diameter classes in natural and plantation forests of northeast India

Table 1. Regression models run to obtain best fit for estimation of biomass in natural and plantation forests of northeast India

Model	Regression equation	R^2	
		Natural forest	Plantation forest
FAO 3 2 3 (1997)	Y= 42 69-12 800(D) + 1 242(D^2)	0 87	0 84
FAO 3 2 4 (1997)	Y= exp {-2 134+2 530*ln (D)}	0 80	0 73
FAO 3 2 5 (1997)	Y= 21 297- 6 953 (D) + 0 740(D^2)	0 87	0 84
Brown *et al* (1989)	Y= exp [-3 114+0 972*ln (D^{2}H)]	0 87	0 65
Brown *et al* (1989)	Y= exp [-2 409+0 952*ln (D^{2}HS)]	0 88	0 67
Chave *et al* (2001)	Y= exp (-2 00+2 42) ln (D)	0 82	0 76
Chambers *et al* (2001)	Y= exp [-0 37+0 33 ln (D) + 0 933 ln(D)^2-0 122 ln(D)^3]	0 93	0 91
Brown & Iverson (1992)	Y= 1 276+0 034 (D^2*H)	0 86	0 63
Brown *et al* (1989)	Y= 38 4908-11 7883 (D) + 1 1926 D^2	0 88	0 85

Table 2 Stand characteristics of natural and plantation forests of northeast India

Variables	Forest types	
	Natural	Plantation
Species richness (No of species)	94	67
Stand density (trees ha ^{1})	996	1028
Density of dominant tree species (trees ha ^{1})		
Antidesma acuminatum Wall ex Wt	120	34
Dysoxylum gobara (Buch -Ham) Merr	36	-
Elaeocarpus tectorius (Lour) Poir	58	44
Leea alata Edgew	90	-
Litsea monopetala (Roxb) Pers	-	102
Mesua ferrea Linn	60	24
Polyalthia jenkinsii Benth & Hk f	108	40
Schima wallichii (DC) Korth	38	46
Shorea robusta Gaertn	-	386
Basal area (m^2 ha ^{1})	73 4	89 9

Total aboveground biomass and carbon stock

The total AGB was significantly greater (p<0 001) in the plantation forest (406 Mg ha ^{1}) than the natural forest (324 Mg ha ^{1}) The aboveground carbon stored by natural and plantation forests was 161 97 and 203 18 Mg C ha ^{1}, respectively The maximum carbon was stored in 60-80 cm dbh class in natural forest (22 5%) and in 40-60 cm dbh class in plantation forest (53 9%) The younger (5-20 cm dbh class) trees, which had highest density in both the forests, stored only 4 4% of total carbon in the natural forest and 8 4% in the plantation forest (Fig 4)

Discussion

The potential of forests to sequester carbon depends on the forest type, age of forest and size class of trees (Terakunpisut *et al* 2007) The observed AGB value of 324 Mg ha ^{1} in the natural forest was comparable with the findings of Ramachandran *et al* (2007) who reported a value of 307 Mg ha ^{1} for the tropical evergreen forests of eastern coast of Tamil Nadu, India However, the present value is less than the AGB values of 607 7 Mg ha ^{1} and 468 Mg ha ^{1} reported for tropical wet evergreen forest and tropical semi evergreen forest of Western ghats of India by Rai (1981) and Swamy (1989), respectively The AGB value was also close to those reported by Muller (1982) for the tropical forests of eastern hardwood region of USA (330 Mg ha ^{1}), and by Brown & Lugo (1982) for the tropical rain forests in Malaysia (225-446 Mg ha ^{1}) and Cameroon (238-341 Mg ha ^{1}) The value range of 153-221 Mg ha ^{1} reported from Sri Lankan tropical rain forests (Brown & Lugo 1982) is lower than the values found in the present study Terakunpisut *et al* (2007) reported an AGB value of 275 Mg ha ^{1} for the tropical rain forests of Thailand which is also less than the present study The lower AGB values of 170 Mg ha ^{1} for the broadleaved forests of tropical America, 260 Mg ha ^{1} for tropical Africa, 215 Mg ha ^{1} for tropical Asia and 150 Mg ha ^{1} for total tropics were reported by Brown & Lugo (1984) However, the AGB *value obtained for this northeast Indian* tropical forest is much lower than the highest AGB value reported so far i e 500-600 Mg ha ^{1} for the undisturbed deciduous forest in the southern Appalachain Mountains (Whittaker 1996)

The AGB of 406 Mg ha ^{1} obtained for the sal plantation forest in the present study was within the earlier reported ranges of 337 - 698 Mg ha ^{1} estimated for sal-dominated forest in Central

Nepal (Shrestha *et al* 2000) However, it is much higher than the estimates of 304 Mg ha^{-1} for the forests of Uttar Pradesh (Singh *et al* 1992) and 261 Mg ha^{-1} for a 10-year old recovering tropical sal forest of eastern ghats, India (Behera & Misra 2006)

The amount of carbon stored in the natural forest of the present study (162 Mg C ha^{-1}) was greater than the disturbed tropical forests of Sri Lanka (77 Mg C ha^{-1}), but lower than the relatively undisturbed matured tropical rain forest of Malaysia (223 Mg C ha^{-1}) reported by Brown & Lugo (1982) Ogawa *et al* (1965) reported a carbon stock of 60 to 179 Mg C ha^{-1} in different tropical forest types of Thailand Flint & Richards (1996) estimated carbon sequestration in Southeast Asia including India, Thailand, Cambodia, Malaysia and Indonesia, and reported the value range of 17 Mg C ha^{-1} in severely degraded tropical dry forest to 350 Mg C ha^{-1} in the undisturbed matured tropical rain forests

The large trees contributed 49% to the total AGB in natural forest In contrast, the contribution of the smaller trees to total AGB in the plantation forest was significantly higher (76%) than the larger trees The greater contribution of large trees to AGB in natural forest was in conformity with the findings of earlier workers (Brown 1996, Brown *et al* 1995, Brown & Lugo 1992, Clark & Clark 1996) who reported up to 50% contribution to AGB by the large trees (> 70 cm dbh) On the other hand, Brown *et al* (1997) reported that smaller trees contribute to most AGB in forests with < 300 Mg ha^{-1} aboveground biomass Analyses have shown that forests with reduced biomass either had their large trees removed by past human disturbance or represent regenerating secondary forests which do not yet have large trees The distribution of biomass in large trees, therefore, could be an indicator of the presence or absence of past anthropogenic disturbance (Brown 1996)

A higher proportion of AGB in the higher diameter classes in natural forest does indicate the important role of large trees in carbon storage, but does not undermine the role of small trees (<60 cm dbh) which would enhance the future carbon stock because of their high carbon sequestration potential It is well established that forest plantations sequester carbon till maturity that varies from 25 to 75 years depending upon the forest type Beyond the maturity, the trees generally have marginal carbon sequestration capability (Lal & Singh 2000) The higher AGB in the plantation forest than natural forest may be attributed to the more or less uniform stand structure that results from a combination of site factors and adopted management practices On the other hand, wide variation in stand structure and tree growth in the natural forest resulted in lower above ground biomass Other factors responsible for such low total AGB are, different stages of forest growth cycle, habitat and species variabilities, and varying tree density (Terakunpisut *et al* 2007)

Many workers have reported that in natural forests, there is a net addition to standing biomass leading to carbon storage if most trees are yet to be matured Such scenarios are applicable to the forests where disturbance events are sporadic and concurrent On the other hand, the matured forests do not add up any further biomass because most part of the gross primary productivity is either used up in respiration or returned to soil as litter with no net addition to the above ground biomass density Such matured natural forests thus do not significantly contribute towards carbon uptake, though they are important for regeneration and sustaining biodiversity However, plantation forests with higher annual productivity were reported to be ideal for carbon storage and sequestration (Lal & Singh 2000) Thus, creation of new plantation on degraded lands is a better option for carbon storage when these are planted and harvested periodically and used as a long-term source of timber

In agreement with the findings of earlier workers, the sal plantation forest with less AGB in higher diameter class had the greater potential to accumulate significant quantities of biomass, and thus sequestering more atmospheric C than the natural forest However, given the fact that both the natural and plantation forests had large tree populations under smaller dbh classes (Fig 2), it can be argued that both the forests have high potential to sequester carbon The present study concludes that the carbon sequestration potential of natural forests of northeast India is high, even with a past disturbance history The carbon values are comparable with most other tropical forests of the region Considering the dbh-AGB distribution pattern, the sal plantation forest had, however, greater carbon stock as well as higher sequestration potential than the natural forest because of ongoing scientific management practices, uniform age and stand structure

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