Forests in Asian tropics are the part of cultural landscape which includes traditional societies as its integral components, meeting their livelihood needs from forest-based resources particularly for fuel wood and timber extraction (Ramakrishnan, 2001). In many terrestrial ecosystems, woody trees are the living constituents of terrestrial ecosystems and landscape modulators as they create resource niches and patches for a whole suite of other organisms dependent on the development, structural support, decay and renewal of trees (Lonsdale et al., 2008; Shachak et al., 2008). The spatial variation in tree genetic diversity, in turn, determines the adaptability of tree populations to environmental change and is thus, essential for the long-term sustainability of forest ecosystems (Savolainen et al., 2007).

Currently 3% of the world’s forests are plantations comprised of 60 million hectare in developed nations and 55 million hectare in developing nations (WRI, 1998; FAO, 1992, 1999). Though, tropical forest cover is declining throughout the world, tropical plantation area has increased dramatically from about 10 million hectare in 1980 to about 44 million hectare in 1990 (Evans, 1982; Lugo, 1997) and to 70 M ha in 2000 (Brown, 2000). Demand for wood products has increased for decades and the plantation forests are most likely to contribute positively to such human needs and to the biodiversity conservations when used to reforest degraded or deforested areas (Moore and Allen, 1999) that can play a key
role in reducing global warming by acting as carbon sinks (Trexler, 1995).

In many regions, older plantations more closely resemble the native forest communities than do younger plantations (Norton, 1998) and the abundance in plant diversity often increases with plantation age (Donald et al., 1998). Tribal communities of North Eastern (NE) Himalayan region heavily rely on forest resources for their subsistence and 90% population of the region use forest biomass as an important source of energy (Bhatt and Sachan, 2004). Shifting cultivation, also known as Jhum or slash and burn agriculture, coupled with excessive deforestation for timber and fuel wood extraction has brought more than 50% area of NE states under wasteland desertification (Anonymous, 2000; Kushwaha, 2006) which in due course of time are typically characterized by impoverished or eroded soils, hydrological instability, reduced primary productivity and diminished biological properties (Bhatt and Sachan, 2004; Ramakrishnan et al., 2006).

At another level, it has been observed that the Jhum farmers in hilly areas of NE region have differential values attached with certain selected tree species and wherever the different ethnic groups come across species like Himalayan or Nepalese Alder (Alnus nepalensis), they leave them in situ and do not slash and burn. Some of them have also gone for sedentary agro-forestry systems with Nepalese alder which is a nitrogen-fixing tree species and can enrich the soil up to 125 kg N ha\(^{-1}\) year\(^{-1}\) (Sharma and Ambasht, 1988; Ramakrishnan, 1992). To meet out growing demand of timber and fire wood in these areas efforts had also been made for mass afforestation through planting suitable woody tree
species like *Alnus nepalensis*, *Castanopsis hystrix*, *Schima wallichii*, *Quercus* spp. and *Dipterocarpus* spp. etc. which have exhibited ideal properties of wood density and biomass ratio (Bhatt and Tomar, 2002) besides the fact that they are grown on hill slopes under highly leached, thin soil cover. Therefore, the fragility of such plantation forests are exacerbated due to tight nutrient cycling into the forest biomass even before the nutrients get into the soil profile (Keiwtam and Ramakrishnan, 1993).

Himalayan alder (*A. nepalensis*), a member of the family Betulaceae and the order Fagales, is common native tree species that occurs in subtropical belts of the Eastern Himalayas. It is predominant in both natural and managed ecosystems in the hilly areas of Manipur and other NE states as a successful colonizer of denuded habitats, nutrient-deficient soils, degraded sites, land-slide affected or freshly exposed rocky eroded hill tops and their slopes (Pradhan, 2000; Chauhan and Misra, 2002). Although it is grown in forestry, agroforestry and shifting cultivation areas of North Eastern India at altitudes between 1000 and 2500 m a.s.l., best growth in this belt usually occurs between 1500 to 2000 m (Sharma, 1988) and is generally planted in perennial landslides to check erosion and to control land-slips. *A. nepalensis* is used as a pioneer plant in forest regeneration with rapid juvenile growth that has the capability to fix atmospheric nitrogen through root nodules having symbiotic association with *Frankia*, an actinomycete. Alder is considered as important tree species in alleviating soil N and other nutrients status and accelerating the N and P cycle thus, increasing the stand productivity (Sharma *et al.*, 1994). It is tolerant to wet soil conditions with high water
use efficiency and is considered as a useful species for social forestry in this region. *A. nepalensis* provides fuel wood for domestic use and the trees attaining more than 25-30 years age provide timber of high value.

*Castanopsis hystrix*, commonly known as chinkapins, is another evergreen tree species of family Fagaceae and originated primarily in the South China and Indochina borders (Li *et al.*, 2007). It is one of the dominant species of broad-leaved subtropical forests located between 800 m to 1200 m a.s.l. in North Eastern India (Haridasan and Rao, 1987). *C. hystrix* serves as a precious timber tree and is mainly used for planking, boat building, door and window frames, furniture making and house construction (Sharma *et al.*, 2011). It is also used for fuel wood and charcoal making in NEH regions. Due to extensive use of *C. hystrix* wood for many purposes, fragmentation occurred in their natural distribution and thus, resulted in population shrinkage and possible genetic erosion (Li *et al.*, 2007).

Inventories on nutrient dynamics and soil quality are major parameters for predicting the site degradation and site amelioration, consequent to raising the indigenous and/or exotic tree plantations and logging of trees in the tropics (Parrota, 1989). Nutrient dynamics in relation to plantation forest age or development may be quite important because the ecosystem structure and other functions do not remain constant as stands mature (Sharma and Ambasht, 1991). Forest soil properties in a small region with similar soil texture are mainly influenced by the vegetation through their litter, root activity and associated microclimate (Bernhard-Reversat, 1988; Kara and Bolat,
In recent years, the interaction between above ground and below ground components of terrestrial ecosystems are receiving much research attention in light of their important driving ecosystem processes that govern primary productivity and carbon balance (Wardle et al., 2004).

Fallen litter, which consists of a set of dead leaves, barks, twigs, reproductive materials and course woody fractions, serves three main functions in the ecosystem such as energy input to soil microflora and fauna, nutrient input for plant growth and material input for soil organic matter build up (Meentemeyer et al., 1982). First two functions are completed through decomposition and mineralization while, the third one through decomposition and humification. Annual litter production varies widely between the ecosystems as well as within each habitat and is controlled by endogenous and environmental factors prevailed thereon. Also litterfall is not constant year around, particularly in ecosystem experiencing climate variability and the different plant materials follow different patterns of fall with highly variable litter compositions. The nutrient input of non-leaf plant parts is generally much lower than that of the leaf litter and therefore, the speed of recycling of mineral nutrients largely depends upon the rate at which leaf litter decomposes at forest floor (Hayes, 1979). Temporal distribution of litterfall and decomposition in different age series of A. nepalensis plantations have been reported by Sharma and Ambasht (1987).

Litterfall and decay processes serve as an index of primary production and as an indicator of the efficiency of nutrient cycles (Vitousek, 1982, 1984). Litter production is closely related to plant
species composition, age structure and growth rate (Faceli and Pickett, 1991; Mohan Kumar and Deepu, 1992; Arunachalam et al., 1998; Jamaludheen and Kumar, 1999; Sundarapandian and Swamy, 1999; Scherer-Lornzen et al., 2007). Species composition is the most important factor influencing litter production within a climate zone (Paoli and Curran, 2007). Numerous studies have reported that litterfall productivity is higher in diverse mixed stands than in monoculture stands (Binkley et al., 1992; Parrotta, 1999; Wang et al., 2007). It has also been reported that total litterfall is similar in primary and secondary forests, but lower in plantations (Barlow et al., 2007). However, the majority of studies related to litterfall in tropical regions have mainly focused on mature and pristine forests and only a few papers have concentrated on tropical plantation forests (reviewed by Proctor, 1984; Vitousek, 1984; Lugo, 1992; Clark et al., 2001; Barlow et al., 2007). Our understanding is less complete with reference to litter production and decomposition in disturbed and regenerating habitats such as subtropical forests and plantations of North Eastern India (Deka and Mishra, 1982; Ram and Ramakrishnan, 1988; Singh and Yadava, 1991; Pandey et al., 2007).

Plant species assemblages and species traits have the potential to influence decomposition process through altering plant-species interactions, plant-decomposer interactions and the microclimate (Berg and McClougherty, 2008; Vivanco and Austin, 2008). Litter decay process has been related to litter substrate quality, population of the decomposers, environmental factors (temperature, rainfall) and the soil properties (Swift et al., 1979; McClougherty et al., 1985). It is reported that these factors played varying role in litter decomposition due to the changing
microbial activity at time and space scale. There are three main levels of litter decay control which operate in the following order: climate > resource quality > soil organisms (Aerts, 1997). Substrate quality, which is defined as chemical composition of the decomposing material, is the most important factor in determining the decay rate, chemical indices of resource quality including initial element concentrations and the concentrations of organic compounds of various classes (Melillo et al., 1982). In general, soil nutrient availability is another factor that influences the rate of litter decomposition because soil can provide the necessary nutrients to decomposers to maintain its life activities (Gartner and Cardon, 2006). Litter substrate quality changes with soil nutrient availability (Xiaogai et al., 2013). Species that grow in low nutrient environment likely produce poor litter substrate quality and therefore, the soil and resource quality could directly affect the decomposer community, abundance and their activity. Early litter decay rate was found to be influenced by nutrient contents and water soluble and structural carbon compounds whereas, the late decomposition rate was dominated by the lignin and cellulose ratio (Zeng et al., 2010).

After analyzing the different nutrient proportions found in more than 90 plant species on a large geographical scale and for nitrogen (N) particularly on a continental level by using 204 litter samples, Liu et al. (2006) have concluded that broad-leaved foliar litters generally have higher N concentrations. Likewise, there were large differences in phosphorus (P) levels of deciduous plant litters averaging three times as much P as in coniferous litters. Loskowski et al. (1995) found significant differences in average initial potassium (K) concentrations between
coniferous and deciduous litters. Cellulose may constitute between 10 to 50% of the deciduous litter mass (Berg and Tamm, 1991). Lignin often makes up between 15 to 40% of the litter mass (Eriksson et al., 1990). Initial composition of lignin varies with the plant species and this variation is sufficient to make differences in litter decomposition period.

Mass loss rate in leaf litters of different plants have been correlated with initial N concentration (Bocock, 1964; Anderson, 1973 a, b; Flanagan and Van Cleve, 1983), initial N and P concentrations (Meentemeyer and Berg, 1986; Blair, 1988) and initial lignin and N ratio (Melillo et al., 1982; Laishram and Yadava, 1988; Fioretto, et al., 2005).

Study of biological parameters in soil and litter, which consists of the microbial population, their diversity and metabolic activities, has received much attention in recent years because the nutrient transformation processes make the soil a dynamic part of the biosphere with the vital role of microbes and invertebrates (De Vos et al., 1994; Kayang, 2001). Link between microbial biodiversity and soil functioning in tropical ecosystem are still poorly understood (Nannipieri et al., 2003). Moreover, one particularly intriguing question is the degree to which the nature of the plant community influences the composition and activities of the soil microbial composition. Several evidences indicate that changes in the nature of plant community as a consequence of deforestation or species selection for forestry could initiate profound changes in ecosystem functioning. As the principal drivers of soil nutrient cycling processes, soil microorganisms are the critical link between shifts in the composition of dominant vegetation and habitat characteristics (Hobbie, 1996; Mitchell et al., 2010) because different habitats exhibit variation in
plant systems, environment and the edaphic factors which greatly influence the growth and development of microbes (Gentry, 1988; Behera et al., 1991; Satish et al., 2007; Shivkumar et al., 2012).

Recent studies on the biogeography of soil microorganisms have demonstrated strong relationship between the soil microbial community and the soil properties such as pH, texture, organic matter content and C:N ratio (Fierer and Jackson, 2006; Fierer et al., 2009; Brockett et al., 2012). These site factors could quite conceivably overwhelm the influences of tree species in structuring microbial population. Trees influence the mineral soil of forest floor through several mechanisms including leaching of dissolved organic materials and nutrients, permeation by roots which may alter soil physical structure and water flow, input of organic matter in the form of root litter and exudation of ions and organic compounds (Grayston et al., 1997; Grayston and Prescott, 2005). Lejon et al. (2005) have found distinct species compositions of bacteria and fungi in surface soil of pure stands of spruce, douglas-fir, oak and beach species. Weand et al. (2010) have also reported the distinct microbial communities associated with conifers which had lower abundance of bacteria and greater population of fungi and actinomycetes.

Seasonal variations in climate regimes, vegetation type and management practices are other factors which affect the diversity of microorganisms in soil-litter systems (Behra and Mukherji, 1985; Maithani and Tripathi, 1996; Zeller et al., 2001). Jha et al. (1992) have reported the microbial population in relation to altitude variation and forest degradation in North East India. Arunachalam et al. (1997) have
recorded maximum bacterial counts during rainy season and minimum in winter while, fungal population was higher during autumn in four forest regrowths in North-East India. However, in comparison to fungal and bacterial components relatively less attention has been paid towards actinomycetes population of decaying litter and the underlying soil. Moreover, limited evidences available suggest that actinomycetes play a minor yet important role in cycling of organic materials as they form a balanced biological community characterized by an extremely diverse flora (Issac and Nair, 2005).

Among different groups of microorganisms, fungi commonly rank as the most abundant in terms of biomass and physiological activity (Kjøller and Struwe, 1982; Schnürer et al., 1985) and comprise approximately 78-90% of the total decomposer biomass in forest ecosystem. In addition to their functioning as primary, secondary and tertiary decomposers in the process of organic matter degradation (Hayes, 1979; Swift, 1984), they are significant elements in food webs involving the soil fauna and microbes. Given the key roles that fungi play in both carbon and nutrient cycling in tropical forests (Lodge, 1993) there is a clear imperative to better understand the tropical forest soil fungal communities (Satish et al., 2007; Panda et al., 2010; Bhattacharyya and Jha, 2011).

Investigation of soil fungal assemblages in tropical forests are limited and mainly rely on dilution plate approaches (Mohanty and Panda, 1994; Persiani et al., 1998) in which soil suspensions are dispersed on the culture media. Methods for study of soil-litter fungi and the limitations
of those methods have been reviewed extensively during the past four
decades (Parkinson and Waid, 1960; Garrett, 1963; Griffin, 1972;
Frankland et al., 1990; Gams, 1992; Parkinson, 1994; Bridge and
Spooner, 2001). Accounts on historically important studies of soil
mycotas are found in other reviews (Griffin, 1972; Christensen, 1981,
1989; Christensen and Tuthill, 1985; Bills et al., 2004). Several guides
for identification of soil fungal genera and species have been compiled by
researchers from temperate regions (Gilman, 1957; Barron, 1968; O’
Donnell, 1979; Domsch et al., 1980; Watanabe, 1994) which also includes
many common tropical fungal species as well.

The reports on specific biotic and abiotic factors that influence
species occurrence and the diversity of soil fungal communities are
numerous and interactive (Christensen, 1989; Beare et al., 1995). In fact,
biotic interactions may exceed abiotic factors as effectors of fungal
composition and diversity in soil (Swift, 1984; Christensen, 1989;
Ingram, 1992). The primary factors promoting high fungal species
diversity in soil are probably the microhabitat heterogeneity,
intermingling of primary, secondary and tertiary decomposers, faunal
grazing, persistence of organic matter that promotes niches partitioning,
temporal changes in climate and on-site vegetation and the remarkable
capacity of fungal cells and mycelia to behave individualistically
(Christensen, 1969, 1989; Rayner, 1994; Zolan, 1995) and thereby
allowing different fungal species to coexist together in the ecosystems
(Christensen, 1981, 1989; Ogawa et al., 1996).
Less is known about the characteristic composition of fungal species in soils of humid tropical forests except that *Aspergillus*, *Penicillium* and *Trichoderma* species usually are abundant (Bettucci and Roquebert, 1995; Saravanakumar and Kaviyarasan, 2010). Tropical soils also contain many rare fungal genera that appear to be primarily tropical in distribution and are commonly associated with overlying litter (Rambelli *et al.*, 1983, 1984, 2004). The litter and humus layers that overlie mineral horizons harbour distinct mycotas. The litter is a transition area between the living plant community and the soil, thus the composition of litter mycota is tied closely to that of plant community and contains distinctive assemblage of fungal species which are largely restricted to decaying plant materials (Bills *et al.*, 2004). Moreover, the relative abundances of the common fungal species in forests and grassland soils vary seasonally (Clarke and Christensen, 1981; Widden, 1986).

Determining the degree to which decomposing litter of a particular tree influence the microbial communities is more challenging for several reasons. First, because microbial population in litter are affected by temperature and moisture conditions and the pure effect of species can only be distinguished by comparing litter of different species decomposing on the same site or the nearby sites. Secondly, a successional pattern of fungal and bacterial communities associated with changing nature of decomposing litter types be compared at the same decay stage (Frankland, 1998; Baldrian *et al.*, 2012). Early intensive studies on exploration of fungal composition in a single decaying litter type were made (Kendrick and Burges, 1962; Visser and Parkinson, 1975;
Sharma et al., 2011) but a few compared fungal communities amongst different leaf litter types in a mixed broad-leaved forest including Quercus, Betula and Corylus trees (Hayes, 1966; Frankland, 1975; 1998). Dominant species of fungi identified were similar in all litter types although the timing of their peak occurrence varied with the state of decay of each litter type. Wallenstein et al. (2010) have reported that the property of underlying soil of forest floor also influences the microbial community in decomposing litter. Therefore, it appears that leaf litters of different tree species develop distinct microbial communities during their decomposition, and that these result from the influences of the nature of litter itself and also from differences in microbial reservoir in forest floor soils under different tree species.

Actual figure of fungal diversity in soil or litter layer still remains unclear (Bills et al., 2004). The diversity of soil fungi largely depends on the isolation method employed and the number of isolates obtained. The well-known estimate of 1.5 million fungal species (more accurately 1.62 M) by Hawksworth (1991), later revised to 2.27 M (Hawksworth, 2001) assumes certain ratios of fungi to vascular plant species and is an indication of generalized number of fungi that exist in nature. If only 10% of these are the components of soil ecosystem, this would still give a figure at least six times the number of species so far, reported from soil. Soil fungi spread easily and most of them are regarded as having a cosmopolitan distribution (Christensen, 1989). Even in tropical forests many taxa are similar to those found in temperate latitudes, and the numbers of species for a particular tropical soil are normally the same or even lower than those observed for soils in temperate regions (Pfenning,
In conclusion, it can be assumed that the fungal species that we currently know is only about 30% of the total diversity in the soil and litter system and further studies on fungal biodiversity from various ecosystems may result in many novel species in the coming years over the globe (Gams, 2007; Mueller and Schmit, 2007).

Activities of microorganisms and other biotic and abiotic factors bring about change in physical structure and chemical compositions of organic matter which in turn determine the species composition of the successive colonizers, and thus, the microbial succession continues until all organic substrates are mineralized (Holland and Coleman, 1987; Kjøller and Struwe, 1987; Singh et al., 1989; Shukla et al., 1990; Rosenbrock et al., 1995). Attempts have been made to explain the underlying principle by studying the species composition of fungal communities appearing at different stages of decomposition of various plant substrates (Garrett, 1963; Hudson, 1968; Frankland, 1981, 1998; Fryar, 2002). However so far, a unanimous theory explaining fungal succession on decaying plant parts could not be established. It appears that each decomposition system has its own characteristic patterns of species composition and succession depending on several factors such as substrate quality, fungal species reservoir of the site and environmental parameters (Swift, 1976). It is also suggested that fungal succession are autogenic process in which the sequential appearance of taxa is determined by the depletion of available carbon sources (Frankland, 1981). Gourbière and Gourbière (2002) studied the persistence and extinction of a fungal species colonizing the natural resource units and
showed that the particular biological traits of individuals/species alone are not fully responsible for fungal successions, but instead that biological patterns and processes play a decisive role in the way in which species are assembled in both the time and space, a characteristic shared by plant communities which has been recognized for a long time (Watt, 1947).

Fungal succession on leaf litter have been well documented in temperate regions over several decades (Webster, 1956; Hudson, 1962; Kendrick and Burges, 1962; Meredith, 1962; Hogg and Hudson, 1968; Wildman and Parkinson, 1979; Kuter, 1986; Gamundi et al., 1987), but there have been only a few such study in forests and/or plantations of tropical or subtropical regions (Promputtha et al., 2002; Tokumasu and Aoiki, 2002; Yanna and Hyde, 2002; Tang et al., 2005; Paulus et al., 2006 a, b; Sharma et al., 2011). A similar conclusion can also be made with regard to the assessment of fungal diversity and their succession in soil or plant litter in the tropics or subtropics as documented by Bills and Polishook (1994) and Paulus et al. (2006 a, b) which have taken a synecological approach regarding the species assemblages at different stages of decay process.

Fungal species composition of leaf litter of several forest tree species has been investigated (Saito, 1966; Hering, 1967; Hudson, 1968; Remacle, 1971; Visser and Parkinson, 1975; Swift, 1976; Kjøller and Struwe, 1980, 1987; Kuter, 1986; Singh et al., 1990; Bills and Polishook, 1994; Kshattriya et al., 1994; Vardavakis, 1998; Sadaka and Ponge, 2003; Tempesta et al., 2003, 2005; Paulus et al., 2006b; Sharma et al., 2011). However, the actual mechanisms responsible for the apparent species
replacements are poorly understood. It has been suggested that fungal successions are autogenic process in which the sequential appearance of taxa is determined by the depletion of available carbon sources. Frankland (1981) have concluded that a variety of physical and chemical changes occurring as decay proceeds may influence the composition of the fungal community. Nevertheless, we still know little about the functional roles of fungi in the decomposition of organic compounds in tropical leaf litter (Osono, 2007), despite the fact that the decay process of leaf litter in soils is characterized by more rapid removal of lignin fraction (Muskoto et al., 2000; Hirobe et al., 2004), which leads to faster decomposition of litter and lower accumulation of soil organic carbon than in temperate soils (Takeda, 1998). Hence, the time for complete decay of leaf litter varies enormously in different regions. It is generally rapid in tropics e.g. 2 months for Magnolia liliifera (Promputtha et al., 2002), 12 months for the leaves of Phoenix hanceana (Yanna and Hyde, 2002), 14 months for leaves of Saccharum officinarum (Hudson, 1962), and 2 years for litter of Ananas comosus (Tiwari et al., 1994), while in temperate regions, 50% the original litter mass remained following a 3 years study of leaf decomposition of Fagus crenata (Osono and Takeda, 2001b), and 95% degradation of Pteridium aquilinum was estimated to take 11-23 years (Frankland, 1976).

The proportion of mass loss and substrate utilization by different groups of fungi may vary among forest stands depending on the reservoir of fungal community within each ecosystem. Decomposition is usually initiated by generalist primary colonizers involving a diverse community of fungi (and bacteria) which utilize simple sugars, oligosaccharides and
other low molecular weight compounds (Rosenbrock et al., 1995). After this initial flush of microbial activity, specialist secondary colonizers mainly the basidiomycetes degrade the composition of more recalcitrant plant polymers complexes (Frankland, 1981).

Therefore, litter decomposition in forest ecosystems is mainly driven by fungal activity among which basidiomycetes are the most ecologically significant group of fungi involved in the breakdown of litter components (Hättenschwiler et al., 2005) by producing a wide variety of extracellular oxidoreductases and hydrolases enzymes (Colpaert and van Laere, 1998; Hofrichter, 2002; Baldrian, 2006). They constitute a major fraction of the living biomass responsible for efficient degradation of many recalcitrant organic compounds in litter and the humic layer (Dix and Webster, 1995). There are three major macromolecular components of plant cell walls which are found in litter namely the polysaccharides i.e. cellulose and hemicellulose, and the aromatic polymer lignin which is being considered as the most recalcitrant compound (Steffen et al., 2002). All three components form a complex referred to as lignocelluloses. Saprotrophic fungi along with basidiomycetes ones are able to degrade all these three principal litter components (Steffen et al., 2000) and are further more involved in the formation, stabilization and degradation of humic substances (humus turnover).

Cellulose and lignin are the major structural components of woody tree leaf litters that constitute 70-80% of fresh organic material and are the important sources in plant tissues that are available to decomposer
fungi (Swift et al., 1979). Cellulases, in particular the enzyme complex consisting of endoglucanase, cellobiose hydrolase and β-glucosidase, hydrolyze the long chains of cellulose resulting in the liberation of cellobiose and finally to glucose. Most of these enzymes have been reported in filamentous fungi such as *Trichoderma* spp. and the basidiomycetous fungi like *Agaricus* spp., *Mycena* spp., *Marasmius* spp. etc. (Cai et al., 1994).

Lignin that alone account for 50-500 mg/g leaf material (Osono, 2007), on the other hand, is degraded with the help of different non-specific oxidoreductases which specifically attack the aromatic moieties, preferably phenolic structures. Most important enzymes in this group are laccase, lignin peroxidase, manganese peroxidase etc. These enzymes also contribute to the formation of humic and fulvic acids in the course of lignin degradation as well as to their subsequent cleavage and partial mineralization (Steffen et al., 2002).

Litter decaying ability of fungi has been examined in pure culture tests which revealed that the functional role of various fungi differ greatly in preferential degradation of structural components in litter (Mikola, 1956; Saito, 1960; De-Boois, 1976; Dix and Simpson, 1984; Miyamoto et al., 2000; Osono and Takeda, 1999, 2002). Pure culture decomposition test is important measure to assess the decomposing abilities and substrate utilization patterns of various fungi (Kuyper and Bokeloh, 1994). Because decomposition of lignin and cellulose are the key factors controlling the mass loss rate it is imperative to evaluate the ability of dominant fungi occurring on decaying litters in order to
understand their roles in decay process. Fungal decomposition of lignin often results in the production of bleached portions in leaf litter which has been often found on forest floor soils of many tropical and temperate ecosystems (Osono and Takeda, 2001a). However, limited researches have been carried out in relation to the comparative abilities of diverse fungi in decomposition of broad-leaved litters and the function of individual fungi in substrate utilization patterns (Osono, 2007).

North East India, blessed with a wide range of physiography and ecoclimatic conditions, is regarded amongst one of the biodiversity hot spots of the globe (Myers et al., 2000). It represents the transition zone between the Indian, Indo-Malayan and Indo-Chinese biogeographic regions and a meeting place of the Himalayan Mountains and Peninsular India. This region is unique in providing a profusion of habitats that features diverse biota with a high level of endemism. Highly undulating topography leads to marked variation in altitude, due to which it is rich in diverse groups of flora and fauna that have been docum 30 (Ramakantha et al., 2003) but the microbial groups have not attracted much attention as yet (Devi et al., 2012). Besides this, studies covering different aspects on litter production, litter decomposition and the quantitative and qualitative nature of associated microbes in general, and the diversity of filamentous fungi in particular, in subtropical plantation forests of NE India are yet to be complemented thoroughly (Pandey et al., 2007; Sharma et al., 2011).

Keeping these points in view, a systematic and comparative investigation was carried out to study the pattern of litterfall, leaf litter
decomposition, nutrient status of soil-litter systems, changes in organic-chemical compositions of litter, quantitative nature of microbial populations and diversity of fungi colonizing the surface soils and leaves of *Alnus nepalensis* and *Castanopsis hystrix* during decomposition processes in two adjacently located subtropical mixed plantation forest stands in Senapati district of Manipur. Assessment of decomposing ability of dominant fungal species with reference to their role in cellulose and lignin degradation was also evaluated in pure culture decomposition tests.