The significance of fire as a shaper of vegetation composition and structure is well known. Studies on the effects of fire on ecosystems have traditionally focused predominantly on fire effects on flora, fauna and more recently also on the soil properties. During the last few decades, however, there has been an increasing trend of wildfires and prescribed fires. Both have profound effects on the functioning of soil system and the overall changes to the ecosystem encountered in a post fire situation can lead to short, medium and long term changes in the soil. These relate to soil functioning in the physical, chemical and biological sense and can include, for example, moisture content, changes in C:N ratios, pH and nutrient availability; and further microbial biomass composition. These effects have typically been studied in isolation by researchers and progress towards a comprehensive understanding of wildfires effects on soils has been held back. Fire is a natural component of many ecosystems, which include plants and animals that interact with one another and with their physical environment. In fact to examine the role of fire in ecosystems in detail, a separate branch in ecology originated known as fire ecology. Fire ecologists study the origins of fire, what influences spread and intensity, fire’s relationship with ecosystems, and how controlled fires can be used to maintain ecosystem health.

Fire is a primary agent of natural disturbance in many ecosystems and has the potential to reshape landscapes and reset biological clocks. Fire is unique as a disturbance in that it causes not only an alteration of the physicochemical and biological environment through heating and oxidation, but also creates new sources of physicochemical and biological inputs into the soil system in the form of charcoal, distillates, metal oxides, and plant litter. Therefore, fire both directly and indirectly influences the forest ecosystem. In most cases, the biotic effects are initiated via changes in the physicochemical (abiotic) characteristics of the soil following fire. The major features of forest disruption and renovation depends upon factors such as intensity and duration of the fire, soil type, soil
moisture content at the time of the fire (Pyne, 1984). Numerous findings on the effects of fire on soil properties are available in the literature. The extent and duration of these effects depend firstly upon fire severity, which, in turn, is controlled by several environmental factors that affect the combustion process, such as amount, nature, and moisture, air temperature and humidity, wind speed, and topography of the site.

Fire severity consists of two components: intensity and duration. Intensity is the rate at which a fire produces thermal energy. Although heat in moist soil is transported faster and penetrates deeper, latent heat of vaporization prevents soil temperature from exceeding 95°C until water completely vaporizes (Campbell et al., 1994); the temperature then typically rises to 200–300°C (Franklin et al., 1997). In the presence of heavy fuels, 500–700°C is reached at the soil surface (DeBano et al., 1998), but instantaneous values up to 850°C can occasionally be recorded (DeBano, 2000). The combination of combustion and heat transfer produces steep temperature gradients in soil. Temperatures at 5 cm in the mineral soil rarely exceed 150°C and often no heating occurs below 20–30 cm (DeBano, 2000). The depth and trend of temperatures depends overall on thickness, and moisture of the fuel on forest floor (Campbell et al., 1995). Duration is perhaps the component of fire severity that results in the greatest belowground damage. After fires, soil temperatures can remain high for from a few minutes to several days.

Basically, there are two types of forest fires: prescribed (controlled) fires and wildfires. Prescribed burning of naturally accumulated forest floor or slash following tree harvest is a standard practice to reduce fuel levels, with the intention of minimizing the extent and severity of wildfires or facilitating germination and growth of desired forest species. They are primed when soil is moderately moist, and consequently they show a low severity (Walstad et al., 1990). In contrast, wildfires generally occur in the presence of an abundant and dry fuel load and, thus, are very severe. Fuel accumulation under aggregated trees, especially of Pines which produce resinous fuels, may result in more intense fire behaviour (Fule and Covington, 1998; Rebertus et al., 1989). However, due to the inhomogeneous spatial distribution of severity, naturally burnt soils often appear as chaotic mosaics of areas little affected by the fire alternating with others seriously impacted (Rab, 1996).
2.1 Effect of fire on physical properties of soil

Soil properties can experience short-term, long-term, or permanent fire-induced changes, depending chiefly on type of property, severity and frequency of fires, and post-fire climatic conditions. A plethora of recent works has been given here as what type of modification selected properties of forest soils undergo following fire. Literature cited here examined wide differences among the various ecosystems and fire severities.

A direct effect of moderate fires on physical properties is the creation of a discreet and continuous water-repellent front parallel to the surface that decreases soil permeability (Imeson et al., 1992). The depth of the water-repellent front is mainly a function of heating but also of soil characteristics, such as moisture and particle-size distribution (Huffman et al., 2001). However, whatever the fire severity and soil features, it rarely exceeds 6–8 cm (Henderson and Golding, 1983; Huffman et al., 2001). Often, due to the irregular pattern of fire severities, portions of water-repellent soil alternate horizontally with portions of permeable soil (Imeson et al., 1992; Martin and Moody, 2001). Structure stability can be increased by low to moderate fires because of the formation of the hydrophobic film on the external surface of aggregates (Mataix-Solera et al., 2004), whilst stability decreases dramatically when, at high temperatures, organic cementing particles are disrupted (Badi’a and Marti’, 2003). In the latter case, however, the surviving aggregates can show a higher stability than the original ones due to formation of cementing oxides (Giovannini and Lucchesi, 1997; Ketterings et al., 2000).

Bulk density increases as a result of the collapse of the organo-mineral aggregates (Giovannini et al., 1988) and the sealing due to the clogging of soil pores by the ash or the freed clay minerals (Durgin and Vogelsang, 1984). It implies a decrease in the water holding capacity of soil (Boyer and Miller, 1994; Boix Fayos, 1997) and a consequent accentuation of runoff and surface erosion (Martin and Moody, 2001). Particle-size distribution is not directly affected by fires (Oswald et al., 1999). In terms of erosion, the combustion of vegetation and litter layer, which mitigate the impact of raindrops on soil and break runoff, often, is more detrimental than water repellence (Sevink et al., 1989; Scott and Van Wyk, 1990; Marcos et al., 2000). Burning and heating can increase dry bulk density and decrease organic matter. It also considerably decrease soil moisture
content at most tensions. All these changes only occur for soils heated to 300°C and above, but the effects did not always increase with increasing temperature (Stoof et al., 2010). Wohlgemuth and Hubbert (2008) reported increase in bulk density increased by 27 percent after the fire, leading to a similar decrease in calculated porosity. This suggests that the combustion of low-density organic material was not confined to the litter and duff layer, but extended into the mineral soil as well. Not surprisingly, soil moisture was reduced by 69 percent after the burn, attesting to the heat and residence time of the ground fire (Wohlgemuth and Hubbert, 2008). Ulery et al. (1996) detected mineral alterations in only 1–2% of a burnt forest, just where concentrated fuel such as logs or stumps had burnt for a long time at high temperatures. Partially decomposed organic matter can hold five times its weight in water and, as a top-dressing; it effectively slows evaporative water loss and moderates the temperature of forest soil (Cohen, 2003). As insects and microbes further digest this organic matter, they release nutrients and substances that glue together individual mineral soil particles. These aggregated particles enhance soil structure by increasing pore space, which, in turn, increases air and water availability. Bulk density was found to be significantly lower at higher elevations (Griffiths et al., 2009). Li et al. (2007) have reported that changes in bulk density were relatively slower than the other measured soil physical properties during the 50 years after revegetation; however, bulk density at the 50-year-old site was significantly different from that found at the reference site. During a 50-year period, they found that recovery for most soil characteristics was more rapid in the early stages than in the later stages.

### 2.2 Effect of fire on soil pH

Forest fires normally cause increase in soil pH because of release of some basic cations. Nardoto and Bustamante (2003) observed that the pH values were significantly higher in the burned site while soil moisture content was significantly higher in the unburned site (p<0.05). In the unburned site, no significant differences in the pH values were found between the rainy and dry season. Soil pH is inexorably increased by the soil heating as a result of organic acids denaturation. However, significant increase occur only at high temperatures (>450– 500°C). Khanna et al. (1994) assessed that the capacity of
ash to neutralize soil acidity is well correlated with the sum of the concentrations of K, Ca, and Mg in the ash itself. Ulery et al. (1993) found that the topsoil pH could increase as much as three units immediately after burning; this rise was essentially due to the production of K and Na oxides, hydroxides, and carbonates, which did not persist through the wet season. In contrast, the neo-formed calcite was still present three years after burning and maintained moderately alkaline soil pH. Fire-induced increase in pH is negligible in soils buffered by carbonates. Increase in soil pH following fire may increase the activity of at least some soil microorganisms.

2.3 Effect of fire on soil nutrients (N, P, K)

Repercussions of fires on the biogeochemical cycle of nutrients have been investigated especially for N and P. The immediate response of soil organic N to heating is a decrement because of some loss through volatilization (Fisher and Binkley, 2000). However, a substantial portion of soil organic N survives low intensity fires, may be by changing the form of N. Moderate to high intensity fires convert most soil organic nitrogen to inorganic forms. In the topsoil of a forest dominated by Eucalyptus, Weston and Attiwill (1990) quantified the fire-induced inorganic N increase into three times the original concentration over the first 205 days, however, after 485 days, a return to the pre-fire level was observed. Ammonium (NH$_4^+$) and nitrate (NO$_3^-$) are the inorganic forms of nitrogen that originate during the burning. Ammonium is a direct product of the combustion, while nitrate forms from ammonium some weeks or months after fire as a result of biochemical reactions called nitrification (Covington and Sackett, 1992). Ammonium is adsorbed onto the negatively charged surfaces of minerals and organics and, thus, is held by the soil (Mroz et al., 1980). However, unless it is fixed in the interlayer of clay minerals, ammonium tends to transform into nitrate. Covington et al. (1991) found that slash pile burning caused an immediate strong increase (approximately 50-fold) in soil ammonium. The importance of a prompt plant recolonisation for conservation of soil N in burnt areas is clearly demonstrated by Weston and Attiwill (1996). Often, soil organic N at the new steady state can even exceed the pre-fire level (Johnson and Curtis, 2000). Soils under frequently burned prairie exhibit lower N
availability, evidenced by reduced concentrations of inorganic N and 33–66% lower rates of soil net N mineralization, relative to unburned prairie. Soils of frequently burned prairie also exhibit less spatial heterogeneity in available soil N, with potential consequences for plant community dynamics (Blair, 2007). Forest fires have not necessarily the same impact on soil P as on N, because losses of P through volatilization or leaching are small. Nevertheless, the combustion of vegetation and litter causes impressive modifications on biogeochemical cycle of P. Burning converts the organic pool of soil P to orthophosphate (Cade-Menun et al., 2000), the sole form of P available to biota. Furthermore, the peak of P bioavailability being around pH 6.5 (Sharpley, 2000), any fire-induced change in soil pH toward neutrality has a positive effect in this regard.

Fires result in an enrichment of available P (Serrasolsas and Khanna, 1995) but this enrichment is destined to decline soon. In fact, in acid soils orthophosphate binds to Al, Fe, and Mn oxides through chemisorption, while in neutral or alkaline soils it binds to Ca-minerals or precipitates as discrete Ca-phosphate. The time in which the positive effect of fire in terms of P availability runs short is highly variable, depending on numerous factors. Fire-induced changes to cycles of soil nutrients other than N and P generally are slighter and more ephemeral. The availability of these nutrients generally increased by the combustion of soil organic matter and the increase is strictly dependent upon type of nutrient, burnt tree species, soil properties, and pathway of leaching processes (Kutiel and Shaviv, 1992). Months after a wildfire, available Ca, Mg, and K in the soil of a Quercus forest were significantly higher than pre-fire levels, but after further three months the increases were almost gone (Adams and Boyle, 1980). Concentrations of cations, such as Ca$^{2+}$, Mg$^{2+}$, and K$^+$, and the anion SO$_4^{2-}$ increase considerably in the soil solution immediately following burning (Khanna and Raison, 1986). The behaviour of micronutrients, such as Fe, Mn, Cu, Zn, B, and Mo, with respect to fire is not well known because specific studies are lacking. Gonzalez Parra et al. (1996) found that both total content and easily reducible forms of Mn increase significantly following fire, thanks to Mn supplied by the ash in the form of amorphous and crystalline oxides, while the exchangeable Mn does not show any variation. Presumably Fe, Cu, and Zn behave similarly to Mn and move downwards very little. Heating can affect nutrient availability indirectly, by modifying the soil microbial community (Perry et al., 1984). Horizons not
subjected to thermal shock can retain most of the elements leached from above horizons (Soto and Diaz-Fierros, 1993). As well as being leached out of soil, nutrients can be removed off-site in particulate form by convection in smoke columns during fire or by surface wind transport. Particulate contributions to elemental transfers is higher where combustion is complete, resulting in formation of a highly nutrient enriched, fine, low-density grey ash, rather than where a coarse-sized black ash forms as a result of an incomplete combustion (Raison et al., 1985). In bared steep surfaces, post-fire loss of nutrients is controlled by water erosion (Thomas et al., 1999).

Li et al. (2007) documented the recovery in soil properties and processes after sand burial at five different-aged revegetated sites in the northern China and found that Soil organic C, total N, total P, and exchangeable K tended to increase with an increasing site age. Soil pH was lower at the younger sites than at the older sites. Recovery of topsoil after sand burial through a revegetation approach needs between 23 and 245 years. The recovery rate is more rapid in the early successional stages than in the later stages, with chemical properties generally recovering more rapidly than physical properties. An increase in available nutrients like N and P and in microbial biomass found after burning (Singh et al., 1991).

Some studies have indicated that soil properties are related to topographic positions in different forest ecosystems. Like, soil moisture content is affected by the slope and aspect in the landscape (Daniels et al., 1987). Soil properties on different slope positions were significantly affected by the degree of soil development and the leaching processes. Soils can significantly accumulate these soluble ions such as Ca, Mg, K, and Na from the highest elevation position and deposit on the foot-slope position where leaching is weaker and soil enrichment is stronger. Significant differences among slope positions were observed for most soil properties. Soil pH, available P, exchangeable K usually increased in a down-slope direction, while organic carbon, available K, tended to decrease. Soils on the foot-slope had a significantly higher pH than those on other slope positions due to the accumulation of soluble cations on the foot-slope. The available N and available K contents were highest in the soils at highest elevation as well as organic C; however, the trend was less regular for the available K and no difference for the available N among other slope positions. In case of available P, the most marked
difference is between the foot-slope sites. The contents of available P and pH value were highest on the foot-slope position. However, the contents of organic carbon, available N and K, were generally higher on the top than those of back-slope and foot-slope (Tsui et al., 2004).

Ubeda et al. (2005) examined a prescribed fire-break in grassland in north-east Spain in relation to its impact on soil quality. All parameters measured showed a significant increase immediately after the fire. But one year later, pH and carbon had declined and returned to pre-fire levels, nitrogen and phosphorus were above, whereas potassium levels had decreased to below pre-fire levels. Overall, the fire did not appear to adversely affect the soil, may be because fire intensity was low and temperature did not exceed 200°C. Higher microbial growth utilizes phosphorous, potassium and causes mineralization of nitrogen hence, amount of phosphorus, potassium decreased and of available nitrogen increased during post monsoon compared to pre monsoon season (Shilpkar et al., 2010). Miesel et al. (2007) observed in their results that burning influences nitrification and available N in the soil, and that those effects differed between burn-only and thin burn treatments. Based on prior studies in coniferous forests, the increases observed in available N are likely to be transitory, and further study is required to determine if there will be longer-term effects of these treatments on the supply of N to trees. The difference in magnitude of increase observed for total inorganic N may be explained by difference in fire severity as lower-intensity fires transfer larger amounts of NH$_4^+$ to the soil than more severe fires, during which a greater amount of soil N is volatized (DeBano, 1991). Mineral soil N may increase after fire due to transport of N from the forest floor (Wells et al., 1979), as well as from the release of NH$_4^+$ from soil minerals and clay-organic complexes during combustion, and both may be followed by conversion of NH$_4^+$ to NO$_3^-$ due to nitrification (Raison, 1979). However, such increases in available N are typically transitory and are likely to dissipate by the end of the second post-fire growing season (Wan et al., 2001).

2.4 Effect of fire on soil organic matter and organic carbon

Fire can cause soil damage, especially through combustion in the litter layer and organic material in the soil. This organic material helps to protect the soil from erosion.
When organic material is removed by an essentially intense fire, erosion can occur. Heat from intense fires can also cause soil particles to become hydrophobic. Rains then tends to run off the soil rather than to infiltrate through the soil. This can also contribute to erosion. In reality, the negative effects of fires on soils are often exaggerated, sometimes many fairly intense fires causes only little soil damage. The effect of fire on the total soil organic matter content is highly variable, and depends on several factors including the type and intensity of the fire, soil moisture, soil type, nature of the burned materials and even slope (Gonzalez-Perez et al., 2004). These effects may range from the almost total destruction of the soil organic matter to increases that may reach 30% in the surface layers as a consequence of external inputs, mainly from dry leaves and partially burnt plant materials in fires affecting the tree canopy (Chandler et al., 1983). Soon after a fire, a sharp decrease in organic matter content may be observed in some soils, this could be accelerated by changes in soil physico-chemical properties i.e., water repellency (DeBano, 2000) and the temporal removal of the herbaceous layer, with effective erosion-controlling belowground root structures.

For Indian tropical ecosystems, Singh et al. (1989) have reported that there is a reciprocal relationship between the plant growth rates (which are highest during the wet period) and microbial biomass (which is highest in the dry period). In the months of March-April (dry season) the average mean value of organic matter was 1.71% in top soil and 1.36% in sub soil layer. In the months of September-October (wet season) the average mean value of organic matter content was 1.94% in top soil and 1.51% in sub soil layer. As a whole it is seen that the organic matter as well as organic carbon content was highest in wet season and was lowest in dry season however, the concentration of K showed a decreasing tendency towards wet season (Hoque et al., 2008). If we compared the value of nutrients at higher elevations with low elevation sites, soil moisture, mean annual precipitation, SOM, labile C, mineralizable N, increased availability of N, were significantly greater at high elevations, pH was significantly lower at higher elevations. Soil organic matter (SOM) accumulation at higher elevations is likely driven by a reduction in decomposition rates rather that an increase in primary productivity (Griffiths et al., 2009). A general increasing trend was observed in soil OC and total N along the slope from summit to toeslope (Nourbakhsh and Rad, 2011).
Soil organic matter is important to a productive forest ecosystem because of its role in stabilizing soil, maintaining soil conditions suitable for seedling establishment and growth, and supplying both nutrient storage and water-holding capacity. Thus, managers should be aware not only of the effect that management activities may have on soil organic matter, but also of the strong interactions that exist among stand density, fuel loading, fire behavior, and post-fire soil organic matter (Miesel et al., 2007). When compared with pre-burn conditions, fire reduced soil organic matter quantity in the burn-only treatment, but not in the treatment units that had been mechanically thinned prior to burning. The significant loss of organic matter in the burn-only treatment could have been due either to loss via combustion during the fire or to increased microbial mineralization of organic matter after the fire (Johnson and Curtis, 2000). The loss of soil organic matter is largely dependent upon the temperature and intensity of a fire (Ahlgren and Ahlgren, 1960); prescribed burning generally results in increased soil C near the soil surface, but higher-severity fires have been shown to decrease soil C. Carbon can be added to the soil when charcoal from burned forest floor material is incorporated into the soil (Johnson, 1992) and this can result in overly conservative estimates of the loss of soil organic C during a fire.

2.5 Effect of fire on biological properties of soil

Biological properties of soil are more sensitive to soil heating than chemical and physical soil characteristics, with fatal temperatures for most living organisms occurring below 100°C. Through soil heating, fire can directly alter the size, activity, and composition of the microbial biomass. The immediate effect of fire on soil microorganisms is a reduction of their biomass. However, almost no information exists regarding how fire affects soil microorganisms over the long-term, and whether any of the changes in the composition or activity of these organisms feedback to impact the forest plant community. Long-term responses of the soil microflora to fire may be primarily due to alterations in plant community composition and production because of the strong interrelationships between plants and soil microorganisms. In fact, the peak temperatures often considerably exceed those required for killing most living beings (DeBano et al.,
In extreme cases, the topsoil can undergo complete sterilization. Adverse effects on soil biota can be due to some organic pollutants produced by the combustion processes. Heat also indirectly affects survival and recolonisation of soil organisms through reduction and modification of organic substrates, removal of sources of organic residues, buffering and every other eventual change to soil properties (Bissett and Parkinson, 1980; Monleon and Cromack, 1996). On the other hand, as demonstrated by Wardle et al. (1997) for boreal forests of *P. sylvestris*, continued fire suppression may lead to late secondary succession under which microbial activity declines. This fact can be explained by limitations in microbial activity and organic matter decomposition imposed by excessive concentrations on phenols, not adsorbed and inactivated by freshly charred materials.

### 2.6 Effect of fire on soil microbial biomass

In a soil under *Pinus* spp. Prieto-Fernaández et al. (1998) assessed that immediately after the occurrence of a wildfire, microbial biomass had almost disappeared in the surface layer (0–5 cm) and reduced by 50% in the immediate subsurface zone (5–10 cm). After 4 years, reductions with respect to pre-fire levels of 60 and 40% for the ratio microbial C/organic C and 70 and 30% for the ratio microbial N/total N were recorded in the top and the subsurface layer, respectively. The addition of cellulose to the burnt soil increased microbial C, but the negative effect of burning was not counteracted completely. In a coniferous stand, up to 12 years were necessary for microbial biomass to return to pre-fire levels (Fritze et al., 1993). In a *P. abies* forest, Pietikainen and Fritze (1995) found that soil basal respiration diminishes after a low-intensity prescribed fire but not proportionally with the reduction in microbial biomass C, evidently because the specific respiration rate is greater in burnt areas than in the control. The impact of fire on biological properties of soil depends strictly on soil moisture. Soils at different levels of moisture experienced different fire-induced declines of microbial biomass C; the highest decline was observed at the moistest condition, may be as a result of faster heat transmission than in drier soils, being water a better conductor than air. Water content is a major factor affecting heat transfer in soils and the effectiveness of the heat for lysing...
microbial cells. As water content increases, the thermal diffusivity of the soil also increases, which increases the depth at which fire-generated heat affects the soil. Temperatures in moist soils do not rise above 95°C until all the water in a given layer is driven off; hence, physicochemical properties of soils are not altered substantially until the soil becomes dry. However, moist heat is more effective at killing soil microorganisms than dry heat. Consequently, for a given fire severity, moist soils will likely lead to greater mortality of the soil microflora than dry soils (Choromanska and DeLuca, 2002).

The loss of shade from forest vegetation, the loss of insulating organic matter, and the accumulation of charred and blackened residues can all influence the temperature of forest soil long after fire has passed. Both the tree canopy and the blanket of organic matter at the forest floor help to prevent heat loss from forest soil. Removal of a substantial amount of these shading and insulating materials by fire will invariably heighten daily and seasonal soil temperature extremes. Changes in soil temperature can have important repercussions for post-fire forest development. For example, elevated soil temperatures tend to heighten the activity of soil microbes, further enhancing decomposition and nutrient release from burned sites (Borchers and Perry, 1990). Burning also alters the specific composition of soil microbial community. Choromanska and DeLuca (2001) demonstrated that applying preventive low-intensity fires may predispose the microbial community to the impact of wildfire and most often there is an immediate net increase in available nutrients after fire, sometimes leading to a short-term increase in microbial activity.

Soil microbial biomass is a living pool containing 1-5% of the soil organic matter. Microbial biomass determinations may indicate changes in the soil organic matter before they can be detected by measuring total soil organic carbon (Jenkinson and Ladd, 1981) making possible its use as an indicator of early changes in soil organic matter content (Cosentino et al., 1998). Soil microbial biomass, both a source and sink of available nutrients for plants, plays a critical role in nutrient transformation in terrestrial ecosystems (Singh et al., 1989). Any changes in the microbial biomass may affect the cycling of soil organic matter. Thus, the soil microbial activity has a direct influence on ecosystem stability and fertility (Smith et al., 1993). Generally, microbial biomass can offer a means
in assessing the soil quality in different vegetation types (Groffman et al., 2001). The results indicated that the MBC, MBN and MBP were significantly lower in plantation stands than in natural secondary forests. The percentage ratios of MBC, MBN, and MBP to soil organic C, total N or total P were significantly reduced in plantation stands; and they varied with time during the growing season significantly. MBC concentration was higher in summer, indicating the more immobilization of nutrients by the microbial biomass from the decomposing litters.

Furthermore, the soil temperature and moisture were favorable for the microbial growth during the summer (Zhu et al., 2010). MBC as measured by chloroform fumigation extraction method was significantly lower in summer than in winter was observed by Lipson et al. (1999). Generally, soil microbial biomass has been considered as the major indicator in the evaluation of soil restoration (Ross et al., 1982). Some researchers have found that soil microbial biomass decreased in the plantations in comparison with the natural forests in tropical and subtropical forest ecosystems (Behera and Sahani, 2003). Yang et al. (2010) found that soil MBC and MBN were significantly lower in the plantation than in the natural secondary forest. Moreover, the values of MBC, MBC/SOC, and MBN/TN significantly varied during the seasons, all with an apparent peak in summer. Soil MBC concentration was higher in summer than in spring and autumn. Their results indicate that natural secondary forest is better in sustaining soil microbial biomass and nutrients than plantation. It is well known that soil microbial biomass greatly depends on soil organic matter as substrate; a decrease in SOC causes reduction in soil microbial biomass (Chen et al., 2005). Thus, the higher MBC and MBN in the natural secondary forest stands than that in larch plantation stands are mainly attributable to the greater availability of organic matter in natural secondary forest stands. This is evident from the significant positive correlations between soil microbial biomass and soil organic matter.

The importance of the soil microbial biomass as a source of potentially available nutrients, and the essential role of its component microorganisms in nutrient cycling, are now generally accepted (Jenkinson, 1988). Microbial biomass can also respond relatively rapidly to changes in land management (Powlson et al., 1987). The effects of fire on soil microbial biomass in other grassland systems have consequently been recently examined.
Variable effects of burning on soil microbial C at grassland sites have been reported, with no change found by Groffman et al. (2001) and an increase by Singh et al. (1991) and Ojima et al. (1994). Results were, to some extent, dependent on burning frequency and sampling time in relation to plant growth recovery. A marked decline in microbial C appears to have occurred in our soil, with values identical 1.5 and 2.5 years after the fire, and about 23% lower than at the unburned site. However, the microbial N value after 2.5 years was about 10% higher than at the unburned site. Microbial immobilization of potentially available N released by burning is likely to have resulted in microbial N enrichment and the lower microbial C: microbial N ratios at the burned site. Ratios of microbial C to total C were non-significantly lower at the burned site. The greater availability of N and P at the burned site than at the unburned site is strongly suggested by the N and P concentrations in the young *Chionochloa rigida* leaves sampled 2 years after the fire (Ross et al., 1997). Increases in microbial N, as well as microbial C, have also been found after burning and attributed to increased plant growth (Singh et al., 1991). Several studies have noted the dramatic increases in available N following fire, most significant increases in inorganic N (Choromanska and DeLuca, 2001).

Some researchers have suggested that microbial biomass is a good indicator of changes in soil fertility, since it responds more rapidly and sensitively than chemical nutrition indexes to changes in fertility. For example, short-term measurements of SMBC can reflect long-term trends in total soil carbon (Powlson et al., 1987). A decrease in soil microbial biomass could result in the mineralization of soil nutrients, whereas an increase in microbial biomass may lead to nutrient immobilization (McGill et al., 1986). Zhang et al. (2011) showed that MBC and MBN differed significantly among all six soils, indicating that microbial biomass seems to be strongly influenced by the nature of the plants, in agreement with the results of Garcia et al. (2005). Garcia also reported that a positive correlation between SOC and MBC is usually found in soils where C is a limiting factor and the soils have reached equilibrium. Differences in MBC in the studied soils were higher than the differences in SOC, confirming that MBC is a more sensitive index of changes of the SOC (Powlson et al., 1987).

The role of the soil microbial biomass in nutrient cycling after forest fires remains uncertain. There have been few studies of the direct effects of forest fires on soil
microorganisms, however there are some notable exceptions which include Vilarino and Arines (1991) and Rashid et al. (1997). The heat during a fire may kill soil microbial biomass in the surface soil. After a fire the nutrients released from the microbial biomass are likely to be removed from the ecosystem by leaching and runoff, but an increase in microbial biomass after fire could limit the losses of nutrients such as N and P. After the fire large amounts of incompletely burnt organic matter were available for rapid microbial decomposition. There were clear differences in microbial biomass between the sites (burnt and reference site) before the burning experiment. Just after the fire, microbial biomass had increased slightly but few days after burning; site showed a decrease of microbial biomass, this was followed by a further decrease in the microbial biomass on the burnt site, whereas the reference site showed a distinct rise (Wüthrich et al., 2002).

Changes in the microbial biomass C/soil organic C ratio reflect the input of organic matter to soils, the efficiency of microbial incorporation, C losses from the soils, and the stabilization of organic C by the soil mineral fractions. The ratio of microbial biomass C to soil organic C may, therefore, be a more sensitive parameter to monitor organic matter dynamics than either microbial biomass C or soil organic C considered alone (Sparling, 1992). Seasonal differences in the ratio of microbial biomass C to soil organic C contents were correlated with elevation and temperature. Ratio of microbial biomass C to soil organic C was correlated negatively with elevation and positively with mean annual temperature. The decrease in the contents of soil organic C with increasing mean annual temperature in the 12 sites studied, this decline in microbial biomass C during summer is thought to be due to increased decomposition rates. In the ecosystems (the mountainous areas of southwest China) studied, the relationship between the differences in microbial biomass C between winter and summer and soil organic C content demonstrates that temperature influenced the decomposition of organic C in soils mainly through its effects on microbial biomass C (Piao et al., 2001).

Nardoto and Bustamante (2003) during their study in savannas of Central Brazil observed that during the rainy season, the organic C at the 0 to 5 cm depth was significantly higher in the unburned site (p<0.05) In 1998, the unburned site showed a reduction of approximately 24% in soil microbial biomass C from October to December while in the burned site (after the fire in September) an increase of 3.4 times in soil
microbial biomass C was observed from October to November (maximum value of 850 mg kg\(^{-1}\) in November) with a subsequent decrease of 55.2% from November to December. They also suggested that the peak in microbial activities occurred with the first rain events, with an initial net immobilization followed by net mineralization. A decrease of soil microbial biomass from October to December was also observed in 1999 (one year after the fire) by them. In accordance with other studies (Bauhus et al., 1993), soils from burned sites had initially higher soil microbial activities than unburned soils. Microbial biomass has been reported to vary seasonally in European soils (Patra et al., 1990). Microbial C and N were highest in summer and lowest in the rainy season. Observations of greater soil microbial biomass in the lower than upper slope are consistent with the results reported in the study (Liu et al., 2010) and previously by others (Liu et al., 2009). These findings could be largely attributable to the differences in soil water availability induced by topography. Positive dependence of soil microbial biomass upon soil water availability demonstrated, in addition to, its direct effects on soil microbes, soil water availability can also indirectly influence soil microorganisms via increasing decomposition of litter and SOM. There were observations of declining positive responses of microbial biomass to annual burning which may also imply the negative responses of microbial biomass to annual burning in the long-term. The magnitudes of increase in both soil moisture and aboveground biomass caused by burning declined gradually (Liu et al., 2010). The MBN values increased towards the lowest slope positions. The values of MBN and inorganic N were significantly higher in toe slopes (Nourbakhsh and Rad, 2011).

Johnson and Williamson (1994) also working on opencast sites, found very small ratios of biomass C: Organic C on sites 3 years after re-instatement. Harris et al. (1989) also used measurements of microbial biomass to determine the impacts of storing topsoil during opencast mining, the results of which indicated the serious additional degradation caused by the stockpiling. Ruzek et al. (2001) also demonstrated that there were clear relationships between time since restoration and increases in soil microbial biomass. Malik and Scullion (1998) showed that increases in microbial biomass during restoration are not related simply to the passage of time. They found that although soil organic matter did increase over time in soils re-instated after opencast mining, there were not
proportional increases in soil microbial biomass, or aggregate stability. This indicates that there were restrictions to successional processes on these sites, possibly related to management.

2.7 Effect of fire on vegetation

Fire ecology refers to the response of the biotic and abiotic components of an ecosystem to a fire regime. This would include type and intensity of fire and season and frequency of burning (Trollope, 2007). Fires at different frequencies affect many aspects of ecosystem structure and functioning, including plant species composition, plant productivity, plant tissue, and the soil microclimate, all of which can affect soil processes and soil biota (Blair, 2007). Heratha et al. (2009) compared the response of vegetation to experimental fires on mature post-mine restored and nearby natural shrubland communities in a Mediterranean-climate region of Australia and observed that the species richness fell by 22–41% after fire in restored sites, but increased by 4–29% in natural sites. Of the species present before fire, 44–60% persisted after fire is restored sites, and 88–96% in natural sites. Only 42–66% of resprouting species recovered in restored sites, whereas 96–100% recovered in natural sites.

Recent evidence (MacKenzie et al., 2004) suggests that woody shrub cover increases significantly with time since last fire. This shift to a woody shrub-dominated understory along with an increase in forest basal area may reduce nutrient availability with succession. In the northwestern forests of US, fire results in the retreat of the shrub community and a renewed dominance by and herbaceous species. In Great Plains ecosystems, fire has long been used to suppress shrub encroachment and augment herbaceous production. Accordingly, the effects of fire on vegetative physiognomy and herbaceous production have received much attention. However, the effect of fire on the temporal stability of herbaceous production has received relatively little attention. Potential effects of seasonal fires, or the resultant alteration in shrub physiognomy, on the temporal stability of herbaceous production are important information for land managers, especially in arid and semiarid environments where precipitation varies greatly between
years. Long-term, post-fire biomass data are needed to determine how fires affect processes such as temporal stability and drought resilience (Castellano, 2007).

There are studies which reported decline in concentration of available phosphorus and nitrogen with the deterioration of ecosystem whereas the concentration of salts of potassium increases in soil with degradation. Consequently, decrease in biomass of herbaceous plants can be attributed to the degradation of soil condition and ultimately lead to deterioration of ecosystem. Dominant plant species changed from trees at forest ecosystem to perennial grasses and herbs at forest grassland and open grassland ecosystem to annual plant species at degraded land ecosystem. Decreased plant density, the number of perennial plant species and herbaceous biomass can be attributed to the deterioration of ecosystem (Panchal et al., 2004). Fires may burn during the pre-growing dormant period, the active growing season, or the post-growing dormant season with variable results that may be related to differences in the fire’s energy release characteristics, plant susceptibility to injury, or physiological response to injury. Seasonality is important because of direct changes in fuel moisture that affect flammability (Johnson, 1992). Kumar et al. (2010) found that soil variables showed a high degree of correlation with tree species richness. However, tree density was clearly negatively correlated to variables like phosphorus and nitrogen and positively correlated with carbon.

It has been proposed that the microbiological and biochemical status of a soil can be used as an early and sensitive indicator of soil ecological stress or restoration processes in natural ecosystems. Plant communities affect soil properties through a range of mechanisms, including amounts and qualities of organic inputs to the soil from the plant community, the effects of plants on the physical soil environment, and the impact of nutrient availability on plant community interactions and plant population growth. Plant species also act as important drivers of soil microbial biomass and soil microbial activity by influencing net primary productivity and litter input to the soil. The dominating influence of plant species on soil microbial biomass has been found to be associated with litter quantity and quality as well as carbon inputs to soil (Wardle and Lavelle, 1997).
Under nutrient limiting conditions, species effects could be significant on soil microbial biomass. Plant species combinations result in greater microbial biomass and nitrogen mineralization rates in the soil. Nitrogen-mineralization of soil organic matter is dependent on organic substrates and the activity of soil microbial biomass (Hungate et al., 1996).