CHAPTER-2
REVIEW OF LITERATURE

Most soils harbour a diverse and rich community of organisms, including multitude of micro-organisms like bacteria, fungi and protozoans and meso-fauna such as mites and nematodes as well as macroinvertebrates namely earthworms, termites and myriapods (Petersen and Luxton, 1982; Hendrix et al., 1986; Crossley et al., 1992). These organisms have been recognized as significant components in the complex soil ecosystem (Hector and Bagchi, 2007; Wagg et al., 2014), and have played an important role in maintaining a healthy soil ecosystem (Jones et al., 1994, Beare et al., 1995, Lavelle et al., 1997, Matson et al., 1997, Lavelle et al., 2001). Soil macroinvertebrates have been involved in the functioning and sustainability of natural and man-made habitats (Gillison et al., 2003), but they have remained poorly understood component of terrestrial ecosystems (Ruiz et al., 2008).

The scientific studies on the ecology of soil organisms dates back to the last quarter of the nineteenth century, when Charles Darwin (1881) brought to light, for the first time, the role of earthworms in the mineralization of materials and transformation of dead organic matter into plant nutrients. Darwin was also the first to recognise the biological functions of soil (Feller et al., 2003). The modern studies on the role of macroinvertebrates in the soil ecosystem were, however, initiated sometime in the second half of the twentieth century.

Several biologists have investigated different ecological and biological aspects of soil macroinvertebrates in various parts of the world (Tsai et al., 1999, 2000; Blakemore 2003; Chang and Chen, 2004, 2005a and b; Blakemore et al., 2006; Sautter et al., 2006). Comprehensive ecological studies on the soil fauna, including macroinvertebrates, have been those of Macfadyen (1957), Kevan (1955, 1968), Farb (1959), Doeksen and van der Drift (1963), Burges and Raw (1967), Graff and Satchell (1967), Wallwork (1976), and Lebrun et al (1983).

In India, a number of workers, Blanchart and Julka (1997), Yadav (2001), Sinha et al (2003), Rossi and Blanchart (2005), Tripathi et al (2005) have studied the distribution and habitat preference of various soil macro fauna. Impact of change in land use on these organisms were assessed by Senapati et al. (2005) in south and southeast India and Bhadauria and Saxena, (2010) in Central Himalaya.
2.1 Synchrony between Aboveground and Belowground Biodiversity

The shaping and functioning of terrestrial ecosystems have been attributed to tremendous diversity of organisms that live in the soil (Bardgett and Putten, 2014). According to Giller et al. (1997), soil ecosystem was significantly impaired and degraded due to decline and loss of soil biodiversity. Though habitat quality may be the most important factor determining the presence of a species at a given site (Duelli, 1997), diversity within a patch additionally depended on the structure of the surrounding landscape.

Lavelle et al. (1997) linked belowground biodiversity, including macroinvertebrates, to long-term fertility of soil. Furthermore, Hooper et al. (2000) and Gilson et al. (2003) observed mutual dependence between aboveground and belowground biodiversity. Diversity of macroinvertebrates decreased sharply due to severe aboveground disturbance, such as replacement of forests with agriculture (Eggleton et al., 2002). However, Warren and Zou (2002) did not find differences in the population of belowground macroinvertebrates under different planted species. Belowground communities usually support higher diversity of organisms than do corresponding aboveground ones, while the factors that regulate their diversity are far less well understood (Lavelle, 1996).

Wardle (2006) reviewed biotic factors that influenced community diversity of soil organisms across a spectrum of spatial scales. At the local scale, soil biodiversity was affected by interactions within trophic levels or by direct trophic interactions. Within the soil, larger bodied invertebrates also influenced diversity of smaller sized organisms by promoting dispersal and through modification of the soil habitat (Lavelle et al., 1997). At larger scales, individual plant species, vegetation composition, plant species diversity, mixing of plant litter types, and aboveground trophic interactions, all impact on soil biodiversity. Further, at the landscape scale, soil diversity also responded to vegetation change and succession (Wardle, 2006).

Moghimian et al (2013) observed that soil organisms, including macroinvertebrates, played a significant role in reviving degraded forests and accelerated growth in natural forests. Several studies, mostly in temperate regions and even in some tropics, revealed that feeding activity of earthworms enhanced decomposition process of litter and influenced nutrient release rates, which affected above ground vegetation (Madge, 1965; Satchell, 1967; Sugi and Tanaka, 1978; Edwards and Bohlen, 1996). In North American forests, invasive
European lumbricid earthworms were more affective in organic matter turnover rate, particularly in areas where there were no earthworms (Hale et al., 2005; 2006; 2008). Rossi and Blanchart (2005) reported significant effect of earthworms on aboveground plant growth. Earthworms were among the most important components of soil biota in terms of soil fertility in Central Himalaya (Bhadauria and Saxena, 2010).

Other important macroinvertebrates like termites made significant contributions to ecosystem processes, including carbon fluxes, soil formation and soil conditioning (Lee and Wood, 1971; Bignell et al., 1997, Bignell and Eggleton, 2000). The aboveground post-logging forest succession, however, affected soil termite species assemblages (Eggleton et al., 1995, 1997). This also led to their reduced overall species richness (Collins, 1980). Changes in plant cover affected termite diversity in the natural vegetation and buffel savanna (Castellanos et al., 2013).

Grass tufts modified the properties of the underlying soil as their leaves acted as a protection against solar radiation, which was critical for foraging termites (Smith and Rust 1994). The density of termites and earthworms was higher under tree trunks than under grass tufts and under bare ground. Gaps between the soil and the tree trunks were also reported to provide shelter for species that had surface activity, such as anecic earthworms, diplopods and spiders (Hamazaki, 1996). The densities of groups like Diptera larvae, Coleoptera larvae, Lepidoptera, Hemiptera, Isopoda, Mollusca and earthworms changed markedly with time, which was attributed to environmental variations. Many soil organisms displayed strong seasonality in their life cycles (Fayolle et al., 1997; Dibog et al., 1998).

Another important group of macroinvertebrates, the millipedes have been considered as important components of belowground soil community (Lavelle, 1997). Positive correlation was observed between millipedes and growth of red oak trees by Galford et al. (1992).

### 2.2 Soil Macroinvertebrates and Land-use Systems

Traditionally, soil communities have been considered as saturated communities with intense biotic interactions between species and trophic groups (Setala et al., 2005). Their distributions were influenced by change in land use systems and agricultural practices (Robertson et al., 1993; Hedlund et al., 2004). Most land use practices resulted in decrease in
the activity and diversity of soil macrofauna (Lavelle et al., 1997). According to Dauber and Wolters (2005), alterations in land use systems created a variety of successional habitats and knowledge about successional changes in species richness and community composition was essential for understanding soil biodiversity dynamics.

Stable ecosystems have been known to harbour high species diversity as compared to unstable environments (May, 1979). Scheu (1992) concluded that individual earthworm species reacted only to the availability of preferred microhabitats in the different succession stages. Variation in abundance and diversity of earthworms was greater in deciduous broad-leaved forests and meadow than in coniferous forests (Wu et al., 2012). Moeed and Meads (1986) observed their higher species richness in lowland broadleaf forest as compared to the less diverse high-altitude silver beech site. Population and diversity of soil macroinvertebrates, including earthworms decreased under agriculture systems (Evans and Guild, 1948; Graff, 1953). Lavelle and Pashanasi (1989) also reported lower density of invertebrates in forest and pasture systems compared with fallow, agroforestry and annual cropping systems.

Bhadauria et al. (2012) investigated density and biomass of earthworm populations in the wide range of land use types viz., moderately degraded natural forests, highly degraded natural forests, rehabilitated forest land, traditional pure crop system, traditional agroforestry system, and abandoned agricultural land and rehabilitated agricultural land in a village landscape in Central Himalayan region of India. They found that a change from traditional pure crop system to traditional agroforestry system resulted in substantial increase in earthworm density/biomass but not in species richness. Further, conversion of traditional pure crop system to abandoned agricultural land and highly degraded natural forests caused drastic reduction in species richness. Smith et al. (2008) recorded the lowest population of earthworms in most intensively managed agricultural land-use and coniferous forest, intermediate population in no-till and tilled organic cropping systems and the highest population in old deciduous forest systems. They also concluded that both land-use intensity and land-use type are strong drivers of abundance and composition of earthworm communities in agricultural ecosystems.

Panigrahi (1993) reported higher macroinvertebrate density in native forests soil than in a south Indian tea garden soil. Senapati et al. (2001) observed that intensive tea plantations
were not conducive to propagation of high populations and activities of soil fauna. Various forest characteristics, including conifer species composition (Lattin, 1993), forest succession (Paquin and Corderre 1997), plant functional diversity (Siemann et al., 1998), coarse woody debris (Marra and Edmonds, 1998), understory vegetation and litter (Bird et al., 2000) were related to macroinvertebrate diversity, which increased with increasing composition of conifer vegetation. Kimmins (1997) reported higher population of macroinvertebrates were greater under herbaceous cover but with reduced diversity. Other studies also found invertebrates occurring in greater numbers under deciduous and herbaceous cover relative to conifer litter (Wallwork, 1983; Paquin and Coderre, 1997; Hammond 1997; Marra and Edmonds, 1998).

Tree plantations influenced earthworm abundance by changing physico-chemical properties of soil viz. temperature, moisture regime, pH, organic matter content and litter inputs (Gonzalez et al., 1996). In their studies on earthworm communities in rubber plantation of different age groups in West Tripura in India, Chaudhuri et al. (2013) observed that both density and biomass increased, while species diversity, species richness and species evenness decreased with increase in the age of plantation.

Several studies have indicated that earthworms and other macrofauna influenced plant production (Lavelle et al., 1992; Brown et al., 1999; Lavelle and Spain, 2001; Wardle, 2002). Panigrahi et al., 2014 found that tea quality improved significantly with inoculation of earthworms in soil in tea gardens. Compared to forest ecosystems, functional guild diversity was lower in agroecosystems (Sinha et al., 2003). Rubber agroecosystem was dominated by endogeic earthworm species, whereas epigeic species formed minor component (Chaudhuri et al., 2008).

2.3 Spatial and Temporal Distribution of Soil Macroinvertebrates

Macroinvertebrates showed three types of basic distributions in soil, namely regular, random and clumped, which were related to patchy distribution of food or water (Usher, 1976; Usher et al., 1982). Several studies have reported that soil macroinvertebrates were irregularly distributed and aggregated (Guild, 1955; Satchell, 1955; Svendsen, 1957; Jimenez et al., 2001) and showed high degree of spatial-temporal organization (Jimenez et al., 2006). Phillipson et al. (1976) studied spatial pattern of distribution in earthworms and found that
their horizontal distribution was complex (Poier and Richter, 1992). Perry et al. (1999) defined aggregated distributions of a species as regions of either relatively large or low density. Robertson (1994) observed that soil fauna was generally spatially structured at scales (<100m). Rossi et al. (1997), however, recorded uneven distribution of many soil macroinvertebrates.

Edaphic factors played a major role in the distribution and occurrence of various earthworm species (Ismail and Murthy, 1985; Bhadouria and Ramakrishnan, 1991; Chaudhuri and Bhattacharjee, 1999; Tripathi and Bharadwaj, 2004). The aboveground plant species richness, coverage and litter were important in determining spatial distribution of soil macroinvertebrates (Pengfei Wu et al., 2012), which was influenced by gradients of nutrient availability and microclimatic conditions (Lavelle, 1983), and micro topography or intrinsic population processes like dispersal and reproduction (Ettema & Wardle, 2002).

Land use systems and agricultural practices, showed direct affect of soil resource patchiness in the horizontal distribution of earthworms (Robertson et al., 1993). However, land-use practices in disturbed ecosystems did not show apparent impacts on the horizontal distribution of these organisms. Further, Jiménez et al., 2001 observed that spatial distribution was determined by factors acting within the population itself. Rossi et al., (1997) indicated that spatial dispersion of a given species was related to its life history and morphological traits.

Distribution of macroinvertebrates was linked to various environmental parameters like substrate, moisture, temperature and pH and resource availability (Robertson and Freckman, 1995). Fragoso and Lavelle (1992) related patchy vertical distribution of earthworms to different seasons and also recorded seasonal vertical movement of both adult and immature stages.

Soil macroinvertebrates showed variable distribution over a period of time (Doblas-Miranda et al., 2007). Earthworms attained high densities and biomass during wet season (Dash and Patra, 1977; Kale and Krishnamoorthy, 1978; Kale and Krishnamoorthy, 1982; Blanchart and Julka, 1997; Rossi and Blanchart, 2005; Joshi and Aga, 2009, Rahman et al., 2011), and they moved to deeper layers to avoid drought conditions during dry and hot summer months (Blanchart and Julka, 1997). Fragoso and Lavelle (1987, 1992) and Fragoso (1993), however, also reported vertical migration of earthworms during rainy and dry seasons.
in Mexican soils. Callaham and Hendrix (1997) were of the opinion that earthworms were most abundant during spring. During low autumn temperatures, a number of macroinvertebrates in the northern temperate forests were reported to undergo quiescence, which lasted till spring (Drift, 1951). In Denmark, Neilsen (1974) reported the seasonal activity of the weevil *Phyllobius argentatus* lasted for 9 weeks during summer months (May to July), and of the moth *Diurnea flagella* for 4 weeks (May and June). Timmerman et al. (2006) recorded minimum coleopteran population density during winter. Changes in the composition, abundance and diversity of beetle communities among the sampling periods were smaller than those among habitats (Pengfei Wu et al., 2012).

### 2.4 Abiotic and Biotic Factors Affecting the Density and Abundance of Soil Macroinvertebrates

Species diversity and abundance of soil macroinvertebrates were dependent on various climatic factors, like dry and/or cold periods, regional variation in vegetation, soil texture, and nutrient contents (Edwards and Lofty, 1977; Lee, 1927; Lavelle, 1987; Kamin, 2010). Johnson et al. (1995) found that high soil moisture, increased organic matter, and lower temperatures affected soil invertebrate communities in older forests. Soil types, vegetations and other abiotic and biotic factors influenced the population of soil macroinvertebrates (Dash and Patra, 1977). Soil macroinvertebrates, vegetation and soil processes were also affected by various soil properties (Lavelle et al., 1997; Lavelle and Spain, 2001).

Earthworm density and biomass fluctuated seasonally, and both physical and organic factors of soil influenced abundance and distribution of earthworms (White, 1975). Abundance and diversity of earthworm population was correlated to biotic factors such as soil moisture, temperature, altitude and agricultural management practices (Handricksen, 1990). Julka (1988) reported that earthworms doubled their population in one month under ideal conditions of temperature, moisture and food availability. Likewise, Tripathi and Bhardwaj (2004) reported higher earthworm species diversity indices and evenness in cultivated soils. In North-west and Central Kashmir, Najar and Khan (2011) correlated higher species diversity of earthworms with high moisture content of soil.
Haokip and Singh (2012) reported that various soil parameters showed more favorable conditions for earthworms in mixed reserved forest ecosystem as compared to disturbed oak dominated and managed oak plantation forest in northeastern state of Manipur. Fragoso and Lavelle (1987, 1992) and Fragoso (1993) observed a relationship between earthworm activity and rainfall; during the rainy season, the worms moved toward the upper layers of soil and in the dry season they migrated to lower layers. This was supported by the work of Callaham and Hendrix (1997) who reported that earthworms were most abundant during spring. Martinez (2006) also found abundance of earthworms in upper layers of soil during the rainy season and their migration to lower layers in dry season. However, Jimenez (1999) did not observe this type of movement for epigeic species. Joshi and Aga (2009) noted increased earthworm activity in high rainfall areas. Mariappan et al (2013) also reported that during rainy season the population density was higher as compared to low population density during summer and winter.

Soil moisture has been implicated in regulating diversity, distribution and abundance of macroinvertebrates (Abrahamsen, 1971; MacKay et al., 1986; Lavelle, 1988; Lavelle, 1995). Earthworms’ population density was the result of the interaction of a number of factors of which moisture is of greater importance (Valle et al., 1997). Ismail et al. (1990) and Ghosh (1993) reported positive relationship between soil moisture and overall macroinvertebrates population density. In their studies on centipedes, Corey and Stout (1992) found that the organisms preferred wet habitats than drier places. Termite diversity was positively affected by soil moisture (Bohlen et al., 1995; Edwards and Bohlen, 1996). Soil moisture played a major role in the distribution and occurrence of various earthworm species (Bohlen et al., 1995; Edwards and Bohlen, 1996; Auerswald et al., 1996). Earthworm population density was correlated with a number of factors of which moisture was of greater importance (Valle et al., 1997). Marra and Edmonds (1998) found that invertebrate biomass and species richness were influenced by soil moisture. Schmidt and Curry (2001) reported low levels of earthworm population in soils with low moisture, and Smetak et al (2007) opined that earthworms were sensitive to moisture stress. Effect of moisture content on soil fauna was also studied by Ganihar (1996) and Mariappan et al (2013).

Macroinvertebrate species structure varied with soil temperature (Abrahamsen, 1971; Dibog et al., 1998; Jimenez et al., 1998). Fecundity, reproductive pattern or competitive
ability of soil arthropods were affected by changes in soil temperature (Hopkin, 1997; Walter and Proctor, 1999). Several species of macroinvertebrates became inactive at low temperatures in northern temperate forests (Drift, 1951). Higher surface temperature was reported to be a limiting factor for temperate earthworms (Nordstrom and Rundgren, 1974).

Population of earthworms and other soil macroinvertebrates declined with decreasing soil temperatures (Dash and Patra, 1977; Seely and Louw, 1980; Kale and Krishnamoorthy, 1982; Ismail and Murphy, 1985; Ganihar, 1996; Timmerman et al., 2006; Karmegam and Daniel, 2007). Higher temperature resulted in higher decomposition rate of organic matter leading to reduced litter availability that negatively impacted litter dwelling epigeic and litter feeding anecic earthworm species and encouraged abundance of endogeic earthworms (Lavelle et al., 1999). The effects of changes in temperature on the populations and biomass of overall soil macroinvertebrates was also recorded by Sharon et al., (2001).

Most earthworm species have been reported to withstand a wide soil pH range between 6.0 and 7.8 (Bouche, 1972; Edwards and Bohlen, 1996; Kale and Krishnamoorthy, 1978; Julka and Senapati, 1987; Chaudhuri and Bhattacharjee, 1999). Reddy and Pasha (1993) reported positive correlation between pH and seasonal abundance of juveniles and sub-adults of an earthworm, Octochaena philloti, in semi-arid tropical grassland in southern Peninsular India.

Very few studies have been made on the effects of quantity and quality of organic resource on the diversity and abundance of soil fauna (Lavelle et al., 1997; Tian et al., 1997b). Higher abundance of earthworms was significantly correlated with soil organic matter by Hendrix et al., (1992). In agroecosystems in which organic residues form chief sources of nutrients, the composition and activity of soil fauna, including their effects on nutrient availability were regulated by litter quality (Tian et al., 1997b; Wardle and Lavelle, 1997).

Available carbon source encouraged population growth of earthworms (Tiunov and Scheu, 2004). Rajiv et al. (2002) found that earthworms changed the soil chemical dynamics as they secreted enzymes like proteases, lipases, amylases, cellulases and chitinases which brought rapid biochemical conversion of the cellulosic and the proteinaceous materials in a variety of organic wastes. Suthar (2009) linked higher organic carbon and nitrogen content of soil with higher macroinvertebrates density and biomass. Mariappan et al. (2013) observed
increase in organic carbon was associated with increased density and biomass of earthworms in agroecosystems in Tamil Nadu.

Application of plant residues (Newman, 1988; Ayuke et al., 2003; Fonte et al., 2009) increased the abundance of macroinvertebrates. Organic residues over short periods increased macrofauna populations but effect on its diversity was not significant (Mando, 1998; Ayuke et al., 2003; Ouédraogo et al., 2004). Boyd (1960) and Hutchinson and King (1980) observed that high quality of litter supported significant diversity and population of macroinvertebrates. These organisms were more abundant in forests with litter covered floor than without litter. Scheu (1992) found low species richness and diversity of soil macroinvertebrates in a 44-year old fallow land and attributed it to homogeneous herbaceous cover. Macroinvertebrate communities were regulated by quality and quantity of the litter (Zou, 1993; Zou and Bashkin, 1998). Mando (1998) reported that plant litter influenced the macroinvertebrates diversity and population in Sahelian crusted soil. In *Acacia mangium* plantation and mixed dipterocarp forest, litter quality and quantity regulated diversity and population of macroinvertebrate communities (Tsukamoto, 2005).

In recent years, biologists have been investigating impacts of macroinvertebrates, earthworms in particular, on nutrient cycling in soil. These studies focussed mostly on soil macrofauna of temperate region (Marinez et al., 2006). A few studies were, however, available for tropics; for example on Indian earthworms by Dash and Patra (1977) and Senapati (1980); earthworms of natural savannas in Ivory Coast by Lavelle (1978), Mexico by Lavelle et al. (1981) and Colombia by Jiménez et al. (1998).

The higher population density of earthworms in reserved forest as compared to disturbed forests was attributed to the higher organic C, N, P and K content in the reserved site which had a direct influence on the availability of food sources of earthworms. Various chemical properties of the soil, viz. organic C, N, P and K have been reported to be responsible for the distribution and abundance of earthworms (Phillipson et al., 1976; Baker et al., 1993). Bhatnagar (1975) attributed low C/N ratio in soils to high earthworm population because of stimulation of N-fixers in drilosphere. According to Edwards and Bohlen (1996), earthworms favoured high quality litter with C: N of 20:1. Aubert et al. (2003) pointed out that the difference in soil macrofauna at the stand level in a beech forest depended on Ca, C and N. Pospiech and Skalski (2006) reported that higher concentrations of macro-elements (N
and K) in soil benefitted macroinvertebrate species richness. Nitrogen limitation was implicated in low macrofaunal density in Congolese eucalyptus plantations (Sayad et al., 2012). Based on PCA studies, Moghimian et al. (2013) suggested that the macrofauna distribution was regulated by total nitrogen.

2.5 Macroinvertebrates and Soil Management Practices

Extensive studies have been made on the variations in soil macroinvertebrates along disturbance gradients, mainly through studies on the impact of cultivation (Wardle, 2002; Bardgett et al., 2005b). These organisms responded negatively to anthropogenic perturbations due to soil management practices (Wardle, 1995, 2002; Bardgett et al., 2005b). The affected macroinvertebrates included earthworms (Fragoso et al., 1997; Decaëns & Jiménez, 2002; Decaëns et al., 2003), termites (Gillison et al., 2003) or macrofauna as a whole (Lavelle and Pashanasi, 1989; Decaëns et al., 1994; Mathieu et al., 2005). Site-specific studies by Eggleton et al. (1996), Okwakol (2000) and Curry et al. (2002) also demonstrated that deforestation, soil disturbance and increased intensity of agriculture negatively affected soil macrofauna species richness and abundance.

Species richness of ants increased during secondary forest succession in Atlantic forest of southern Brazil (Bihn et al., 2008). Scheu (1992) and Decaëns et al. (1997) observed little change in earthworm species richness along secondary successional gradients under temperate climatic conditions. Ponge and Delhaye (1995) recorded steady species richness of earthworms in a beech forest in north-western France. In grazed systems, the diversity of soil fauna usually increased after sowing and then stabilized (Hedde, 2006; Chauvat et al., 2007), indicating positive response to increased resource availability. The use of organic inputs and crop rotation favoured macrofauna diversity due to improvements in the abiotic conditions and increased substrate supply (Tian et al., 1997; Ayuke et al., 2003; Curry, 2004; Osler et al., 2008; Fonte et al., 2009). Ayuke et al. (2011) investigated the effect of agricultural practices on earthworm and termite diversity across humid and semi-arid tropical of Eastern Africa. They found a decrease in earthworm species richness in high carbon treated soil than in fallow soil and high termite species richness in fallow soil than in the cropping systems.
Chapter-2

Review of Literature

2.6 Altitudinal and Latitudinal Impact on Soil Biodiversity

Few studies have been carried out on altitudinal variation in soil macroinvertebrates (Decaens, 2010). Species richness of earthworms decreased with high elevations in France (Bouché, 1972; Dahmouche, 2007), for ants in the Smoky Mountains, USA (Cole, 1940) and for termites in Indonesia (Collins, 1980). However, a humpbacked pattern with a peak of taxonomic richness at intermediate altitudes was reported for dipterans between 1300 and 1700 m (Collins, 1980). González et al. (2007) observed that earthworm species richness increased from 0 to 1000 m altitudes. Altitudinal variation in macroinvertebrate species richness was related to altitudinal variation in environmental factors such as rainfall, temperature, pH or quality of organic matter (Decaens, 2010).

Species richness in soil macroinvertebrates increased from northern latitudes towards the equator as exemplified in ants (Kusnezov, 1957) and earthworms (Lavelle, 1983). On the other hand, Abensperg-Traun and Steven (1997) observed that there was no consistent decline in species richness from the lowest to the highest latitudes.

2.7 Impact of Disturbance on Macroinvertebrates

Soil macroinvertebrates are very sensitive to disturbance and associated stress (Barros et al., 2002). Density of earthworms declined in Central Himalayan forest ecosystems with different degrees of disturbance which also lead to dominance of exotic species (Bhadauria et al., 2000). According to Decaens et al. (2006) and Mboukou-Kimbatsa et al. (2007), introduction of exotic plants affected soil faunal community. Studies of Sileshi and Mafongoya (2006), and Apigian et al (2006) reported lower species richness and population density of annelids, chilopods, arachnids and some insects under disturbed forest patches. Abensperg-Traun (1992) observed grazing had no effect on the population of termites because of their food plasticity. Reversal of farming and forestry practices helped in conservation of invertebrate diversity in the Australian soils (Greenslade, 1992). Decline in soil fauna was attributed to disturbances and stress due to impact of logging and fire (Jones et al., 2003; Donovan et al., 2007). Steinbauera and Peveling (2011) were of the view that input of chemicals in soil inhibited activity and diversity of macroinvertebrates. Dash (2012) reported proliferation of termites at the expense of earthworms in disturbed systems.