CHAPTER - II
REVIEW OF LITERATURE

Biodiversity has been well introduced and approached the status of a household word after the Rio de Janeiro Earth summit in the year 1992. Convention on biodiversity at the earth summit brought the subject into the mainstream of politics, and contemporary science on environmental studies. Much of the writing moved back and part across the boundaries of biology, the physical science, economics and forestry. The biodiversity research include the systematic analysis of hereditary variation at all levels of biological organization, from genes within population to species to ecosystem, together with the development of technologies to conserve and manage the diversity for the benefit of humanity.

In 1990’s the field of biodiversity become an interdisciplinary subject at global level. The rising interest in the practical side of biodiversity for the past three decades is mainly due to the habit destruction form the toxic pollution, ozone depletion, climatic warming and other changes in the physical environment. The species extinction cannot be reversed, the loss of species cannot be replaced by new one. Actually each and every individual on earth are created through a long-term evolutionary process. The present review on biodiversity is restricted and relevant to the main thematic area of research, which include the historical basis of biodiversity, characterizing biodiversity, indicators for biodiversity assessment, biodiversity monitoring through remote sensing, relationship among and between biodiversity and ecosystem functioning, successional diversity and forest ecosystem
functioning, biodiversity and ecological interactions, biodiversity and economical aspects and biodiversity act.

**Historical base for biodiversity**

The investigation of life through time, the main focus of biodiversity studies has been on the numbers of species - or their surrogate higher taxa - in systems ranging from local habitats to world - encompassing ecosystems. Beginning with the work of Phillips (1860), paleontologists have intuited that counts of fossil taxa through the geologic record provide a metronome of the pace of large - scale evolution. Simpson (1953) made some of paleontology’s most important contributions to the modern synthesis with ideas about how rapid expansion of diversity in fossil groups related to chance adaptations of a Darwinian nature that permitted progenitors to exploit new ecological regimes. Our knowledge of the history of biodiversity and of processes of diversification has progressed considerably since Simpson’s time, and the fossil record.

Vast majority of scientists interested in present day diversity believed that the biodiversity crisis is caused by species becoming extinct at an alarming rate. Some estimates place the rate of extinction at 74 species per day (mostly in the tropics), which is a “conservative” estimate; higher estimates range to 150 species extinctions per day. A few workers have described this current episode of extinction as the most severe ever suffered in the history of life (Briggs, 1994). However, the estimate of the daily extinction rate is certainly not precise. It is based on sketchy data on the degree of endemicity of tropical animals, numbers of species (mostly large animals) presently known to be endangered, measurement of the area of
tropical rainforest destroyed each year, and considerations of species-area relationships. More conservative estimates of extinction rate range from 5.5 to 41 species per day, assuming $10^7$ species (Pimm et al., 1995).

**Characterizing biodiversity**

Characterization of biodiversity depends critically on the work of three scientific disciplines *i.e.*, taxonomy, ecology and genetics. Organisms occur in an intricate spatial mosaic classified on a world scale into biogeographic zones, biomes, eco regions and oceanic realms, and at a variety of smaller scales within landscape into ecosystems and communities. The biodiversity at landscape level can be characterized by measures of species richness, species diversity, toxic diversity and functional diversity. Hence, in the assessment of characterization units and techniques levels of description, the remote predictors or surrogates often play very significant role in measure richness. The habitat surrogates including classification of vegetation, details on the physical environment, factors determining the biodiversity loss in a spatial context may be of practical information value and could reduce sampling intensity. This information base could also guide detailed sampling on the ground. These larger scale surrogates include entire functional system and are more likely to promote population viability in the ecosystem.

In conservation, this is likely to differ with earlier measures of ecological diversity formulated with the narrower aim of representing differences in abundance among species, exploring distribution of resources within community. If the value of biodiversity to a conservationist is associated with its use to people then this ought to be separated carefully from issues of rarity, viability and threat. If the
biodiversity value is associated with richness in a currency of characters of organism then the higher level of biological organization (or environmental factors affecting its distribution) will have to be used in surrogate measures. The taxonomic inventories in the past have only been able to reach partial level of understanding the richness.

**Indicators for biodiversity assessment:**

Indicators are key tools in biodiversity and natural resources assessment. No universal agreement on indicator choice or necessary properties exists. Some authors restrict indicators to “quantitative measures” (Convention on Biological Diversity, 1997) yet some important or interesting features of biodiversity and natural resources are of a qualitative nature. Some authors have proposed selection criteria to choose indicators (Riley, 2000). The SDI group proposed six general and four specific selection criteria. i) capacity to reflect changes in important endowments ii) reflects an issue that could have significant costs or benefits for current or future generations iii) reflects an issue that could be addressed for a period of time iv) reflects an issue that involves thresholds beyond which small changes could potentially lead to irreversible effects. Other desirable features of indicators may be: first, to quantify and simplify information in such a way that its importance is clear; second, to be able to detect changes in time and space; third, to have scientific credibility; fourth, to be able to be represented in diverse ways to address different audiences. And two final desirable features are to have the capacity to distinguish natural and man-made changes and to be easy to understand (Convention on Biological Diversity, 1999). Other valuable properties of indicators
are: universality, portability, sensitivity to change, be operationally simple and inexpensive, already in existence with historical data and be of wide international use (Riley, 2000).

As per the guideline proposed by Subsidiary Body on Scientific, Technical and Technological Advice (SBSTTA) in order to understand the indicator representation from all three levels. Ecosystem quality indicators are proposed at three levels: the ecosystem itself, the species level and the genetic level. At the first level, habitat fragmentation/conversion and species richness indicators are included. At the species level, indicators of change in abundance or distribution of selected species and threatened species are considered. At the genetic level, indicators are related to replacement of indigenous crops or animal races by alien ones. With respect to pressure indicators, they are grouped into six types: population density, harvesting/use-indicators, infrastructure, pollution, alien/invasive species and climatic change indicators. Additionally, two groups of pressure indicators named as indicative reserves are included. They are habitat management and special habitat indicators. Finally, ecosystem use-indicators are related to ecosystem goods and services (Camacho-Sandoval and Duque, 2001).

**Relationships among genetic, species and ecosystem levels of biodiversity**

The importance of the conservation of all three fundamental levels of biodiversity (ecosystem, species and genes) has been widely acknowledged, but only in recent years it has become technically feasible to consider intraspecific diversity, *i.e.*, the genetic component to biodiversity. In order to facilitate the assessment of biodiversity, considerable efforts have been made towards identifying
surrogates because the efficient evaluation of regional biodiversity would help in designating important areas for nature conservation at large spatial scales. However, known little about the fundamental relationships among the three levels of biodiversity, which impedes the formulation of a general, widely applicable concept of biodiversity conservation through surrogates (Gugerli et al., 2008).

Genetic diversity defines the evolutionary potential of species and is consequently of prime importance for the long-term preservation of biodiversity in changing environments (Forest et al., 2007). However, intraspecific diversity is often neglected in conservation strategies because of difficulties not only of rating its significance, but also of merely quantifying it.

The persistence of populations has been shown to be positively linked to genetic variability. Although Lande (1988) argued that demographic factors were more important than genetic ones in determining the short-term persistence of populations, it is now accepted that demographic and genetic processes often act synergistically. Moreover, genetic variability may interact with demographic effects to foster the “extinction vortex” of small populations.

Species richness represents the most widely applied measure in biodiversity assessment (Gaston, 1996) many attempts have been made to avoid the time consuming and expensive direct assessment of all species distribution to facilitate conservation planning (Moreno et al., 2007). A popular approach in conservation is to rely on focal species as surrogates for regional biota. However, recent investigations question this approach which may not perform better than if any randomly selected species were studied (Andelman and Fagan, 2000). In the same
way, the use of higher-taxon sets as surrogates is often not satisfactory. Alternatively, deducing overall species richness from a taxonomic subset of organisms, thought to be representative of other taxonomic groups, is widespread, but has yielded contradictory results (English et al., 2005). At the same time, there is very limited knowledge on the relationship between species richness and genetic variation. Currently, mainly theoretical considerations have been published, while empirical data are mostly restricted to single species studies (Vellend, 2004) or are limited in sample size or geographic and taxonomic representation (Vellend and Geber, 2005).

Integrative studies have demonstrated that intraspecific diversity may have a positive effect on the associated species richness (Crutsinger et al., 2006) and that genotypic diversity may enhance ecosystem resilience (Reusch et al., 2005). It is argued that locality characteristics, e.g. area, geographical isolation or environmental heterogeneity, may affect both diversity levels in parallel, i.e. via similar neutral processes such as drift and immigration, making it plausible that intra- and interspecific diversity are positively correlated. Such a correlation was illustrated in single-species studies in island situations, but not for mainland situations (Vellend and Geber, 2005).

Alpine plants would likely display a positive relationship among the three diversity levels. Considerably more complex is the effect of adaptation and evolutionary history on intra- or interspecific biodiversity, and selective effects of species diversity on genetic diversity and vice versa is thought to impinge on the above parallel neutral processes (Vellend and Geber, 2005).
There have been attempts to take into account alternative concepts of biodiversity, *e.g.* phylogenetic diversity of plants and animals, in designing networks of protected areas (Forest *et al.*, 2007), consensus has emerged that directing conservation efforts at protecting species or ecosystems richness only does not suffice to allow for the preservation of the future evolutionary potential of species. At best, all three biodiversity levels should be adequately considered when setting priorities in biodiversity conservation and optimising the network of existing protected areas. At the same time, no empirical studies are available that evaluate the interrelations among the three biodiversity levels (Lapointe and Rissler, 2005). This lack of knowledge is not surprising because the assessment of species distributions is already a challenge on its own, and because assessing intraspecific diversity is even more expensive and time-consuming. In contrast, environmental diversity can be easily assessed nowadays if digitalized geophysical maps and climate data are at hand.

Species diversity may not only be a function of the annual means of climate variables but also depends on the latter’s temporal variability. Therefore, standard deviations, in addition to means, of climatic variables may serve as surrogates for ecosystem diversity.

An ecosystem level, the composition of any organismic assemblage results from several factors. (i) The geographic distribution of organisms has always been changing over time owing to natural environmental change, in parallel with the evolution or extinction of species. (ii) Organisms have survived past periods of climatic extremes in refugia. (iii) Colonization ability and genetic diversity differ
among species. (iv) Endemism is not randomly distributed. (v) Perturbations due to natural (or anthropogenic) disturbance have lead to the destruction and fragmentation of habitats. In consequence, the same elementary processes likely caused similar general patterns of diversity (Thrush et al., 2001).

At the species level, ecologists focus on the maintenance and stabilization of species diversity and its effect on ecosystem stability (Chesson, 2000). The emergent patterns suggest that species diversity is often maintained through stabilizing processes that alter the strength of intraspecific (genotypic) and interspecific competitive interactions involved in species coexistence. The mechanisms that maintain diversity include resource partitioning and other fluctuation-independent mechanisms (Chesson, 1994) as well as fluctuation-dependent effects of temporally and spatially varying environments (Chesson 2000). Both models of genotypic and species diversity maintenance rely on environmental variation and the competitive abilities of the genotypes or species (Rainey et al. 2000), but have thus far been left unexamined simultaneously.

The ecological opportunity provided by one genotype, or species, is often required for the maintenance of a competing genotype. Previous models suggest increasing neutrality with increasing species richness with the assumption that the mechanism generating this relationship is increasing niche overlap with increasing species richness (Gravel et al., 2006). Such competitive dominance among species is explained in terms of the resource-ratio hypothesis (Pacala and Tilman, 1993) and is defined with regard to community structure (Tilman, 2004). More often the
studies on species showed the interlink between both genetic and ecosystem levels of biodiversity.

**Biodiversity monitoring through remote sensing**

Biodiversity monitoring by remote sensing, a major technology that enables us to scale up the understanding and knowledge on diversity from points to geographically wide and continuous region, has to be developed. Remote sensing brings us tremendous possibilities to estimate the climatic and anthropogenic impacts on the biodiversity, and moreover, to predict its temporal change in the future through ecosystem modeling.

Generally, measurements of remote sensing do not inherently have information about the biodiversity neither to ecosystem level, species, nor genetic level by itself. Even the highest resolution sensor of the airborne remote sensing would not identify insects. However, when the remote sensing is coupled with the related information at the ecosystem, the distribution patterns of the ecosystem and habitat which can be delineated by remote sensing provides us ideas on the biodiversity from the species level.

The biodiversity is strongly related to human well being, the monitoring of biodiversity is a significant global environmental issue. Global Earth Observation System of Systems (GEOSS) is composed of nine Societa Benefit Areas (SBAs), and biodiversity is one of the SBAs, with Group on Earth Observation (GEO) Biodiversity Observation Network (BON) as its primary task. Corresponding to GEO BON, Japanese Biodiversity Observation Network (JBON), the regional network in Japan, was have organized and held the first meeting in May 2009 in
Tokyo, Japan. Following JPON, Asia Pacific Biodiversity Observation Network (AP BON), the regional BON over Asia pacific region, held the first meeting in July 2009 Nagoya, Japan. Thus BONs have basically eight working groups (WG) and WG 7 of them is mandated for in-situ and remote sensing interaction. Remote sensing, by linking with in-situ observation, will provide us plenty of biodiversity-related information as ecosystem types and their distribution patterns, which could be the information about habitat structure for organisms. Integrated in situ observation from viewpoints of ecology, ecosystem science, and hydrometeorology are required for remote sensing studies of the biodiversity.

**The relationship between biodiversity and ecosystem functioning**

The relationship between biodiversity and ecosystem functioning has emerged as a central issue in ecological and environmental sciences during the last decade. Increasing domination of ecosystems by humans is steadily transforming them into depauperate systems. Because ecosystems collectively determine the biogeochemical processes that regulate the Earth system, the potential ecological consequences of biodiversity loss have aroused considerable interest. Recent experimental and theoretical work in this area has also led to animated debates and controversies. Human impacts on the environment from local to global scales cause not only a general decline in diversity, but also predictable functional shifts as sets of species with particular traits are replaced by other sets with different traits. This has resulted in the current debate in which scientists disagree about the relative importance of functional substitutions and declining species richness as determinants of changes in ecosystem functioning. Comparative studies have begun
to reveal the extent to which functional substitutions alter ecosystem properties such as productivity, decomposition rates, nutrient cycling, and resistance and resilience to perturbations. On the other hand, a new wave of experimental studies has manipulated species richness by using synthesized model ecosystems in both terrestrial and aquatic environments. Both approaches suggest that a large pool of species is required to sustain the assembly and functioning of ecosystems in landscapes subject to increasingly intensive land use. It is not yet clear, however, whether this dependence on diversity arises from the need for recruitment of a few key species from within the regional species pool or is due to the need for a rich assortment of complementary species within particular ecosystems.

There are two approaches to explain biodiversity and ecosystem functioning, the first classical approach attempts to identify the causes of spatial variation in diversity across environmental gradients. Variation in diversity is often correlated with productivity, but also with many other factors that influence productivity, such as soil fertility, climate, disturbance regime, or herbivory. The recent second experimental approach examines whether diversity alone has a local effect on productivity within each site, when all these other factors are held constant. The two approaches can be reconciled by considering that spatial patterns reveal correlations between diversity and productivity driven by environmental factors, whereas small-scale experiments reveal the effects of species properties and diversity on productivity that are detected after the effects of other environmental factors have been removed. Whether biodiversity loss will affect large-scale patterns of productivity hinges on the shape and steepness of the local dependence of
productivity on diversity. Generally speaking, the relative effects of individual species and species richness may be expected to be greatest at small-to-intermediate spatial scales, but these biological factors should be less important as predictors of ecosystem processes at regional scales, where environmental heterogeneity is greater. Whereas diversity was manipulated as the independent variable in recent experiments, at large scales, species diversity itself is a dynamical variable and adjusts to changes in environmental conditions. Abiotic factors then tend to be the main drivers of variations in ecosystem processes across environmental gradients. Diversity loss at regional scales and dispersal limitations due to landscape fragmentation, however, will very likely feedback and reduce the pool of potential colonists at local scales and hence the potential for local compositional adjustments to environmental changes. Species-area relations imply that the long-term maintenance of a given level of diversity at local scales requires a much higher diversity at regional scales. One of the most potent effects of declining diversity could be the decline in the rate at which appropriate potential dominants are recruited during ecosystem assembly. To understand and predict changes in biodiversity and ecosystem processes at large scales, therefore, we need to move beyond unidirectional causality approaches in which diversity is either cause or effect, and address feedbacks among biodiversity changes, ecosystem functioning, and environmental factors. Relationships between local, landscape, and regional scales also require particular attention.
**Successional diversity and forest ecosystem function**

Forest inventory data was used to examine the relationship between successional diversity and forest ecosystem function. The inventory data show that stands composed of early successional species are more productive than stands composed of late successional species, whereas stands composed of late successional species have lower turnover than stands composed of early successional species. Over the past decade numerous experiments have been conducted to examine the relationship between species diversity and ecosystem function. In most ecosystems, the relationship between diversity and productivity has been shown to be positive (Tilman *et al.*, 1997). In a few ecosystems, however, no significant relationship has been observed, while in others increasing species diversity has been shown to decrease productivity (Hooper and Vitousek, 1997). Recent syntheses of the experimental results suggest that a positive relationship between diversity and productivity depends on a positive relationship between productivity and competitive ability (Kinzig *et al.*, in press). If a species’ ability to maximize carbon gain in a particular environment also confers a competitive advantage in that environment, competition will maximize productivity by favoring the most productive species. Thus, a polyculture will be more productive than a monoculture because the most productive species will prevail in each environment within the ecosystem. However, there may be exceptions in which a species’ ability to maximize carbon gain does not confer competitive dominance.

In forests the fast-growing early successional species are replaced by slower growing late successional species. Thus, productivity is not maximized by
competition because stands dominated by slow-growing late successional species fix less carbon than forests dominated by fast-growing early successional species. Nor is productivity necessarily enhanced by successional diversity; stands containing a mix of early and late successional species may fix less carbon than stands dominated by a single fast-growing early successional species. Forests have scarcely been mentioned in the biodiversity debate due to the difficulty of conducting experiments in ecosystems with slow dynamics. Yet, there is a vast store of observational data on forests that can be used to examine whether forest ecosystems differ in some fundamental way from other kinds of ecosystems. While observational data cannot be used to establish causality, they can be used to document whether the relationship between diversity and productivity is positive, negative, or neutral. To examine this possibility, we assessed whether productivity is positively correlated with diversity per se. Of course, it is also possible that causality runs in the opposite direction; that is, more productive stands may simply permit the coexistence of more species. A strong positive relationship between productivity and diversity there are two primary conclusions that can be draw by Caspersen and Pacala (2001). First, successional composition has a significant effect on forest ecosystem function; that is, early successional stands are more productive than late successional stands, whereas late successional stands have a lower turnover than early successional stands. This result has clear implications for the management of forests as carbon sinks.

Second, successional diversity is positively correlated with productivity, as is species diversity per se. If increased diversity does in fact enhance productivity,
then this result also has important implications for the management of forests as carbon sinks. In particular, this result would suggest that forests should be managed in such a way as to maintain species diversity, while also favoring species that maximize the function of interest.

**Biodiversity and ecological interactions**

Plants are sessile and bound to a two-dimensional surface. Resources are supplied in a spatially consistent manner: light from above, nutrients from below, resulting in predictable and relatively easily modeled patterns of competition and resource use. Heterotrophs, in contrast, are generally mobile and their resources are structurally complex. Practical consequences of consumer mobility for field diversity experiments include the difficulty of maintaining specific treatment combinations of animals, as well as the “fencing effect” (Krebs et al., 1969) that prevents natural dispersal and population diffusion. As a result, testing the effects of animal diversity on ecosystem function will likely need to rely more than plant experiments have on natural experiments and patterns in unmanipulated systems (Jackson, 2001), in addition to the elegant experimental designs so successful in previous research on plant diversity effects. The impacts of multiple consumer species on prey may also be complicated by interference or other emergent effects of multiple co-occurring predators, introducing non-linearity into consumer-prey interactions.

Erlich and Raven (1964) stated that the species evolution response to the other species in the community leads to pair wise and diffuse coevolvation. These two conditions may be either mutualism or antagonistic, such interactions in an
ecosystem have chief control to decode the assemblage of plants. Mutualistic co-evolution can result in extremely complex interconnected life cycles. Under such circumstances, either one or both species are incapable of surviving without the other.

The relationship between caterpillars and their host plants provide examples of antagonistic coevolution. Many plant species have toxic substance, especially in their leaves, and many different caterpillars have overcome these poisons. There are several requirements a plant must fulfill to have an insect pollinator; the plant must attract the insect by producing a flower which advertises that the plants has provided a reason for the insect to visits, such as nectar. The insect benefits from visiting the flowers as it obtains nectar or pollen as food. The plant benefits from the visit by obtaining pollen from another flower. To take advantage of an insect visit a flower must have its reproductive organs in such a position that the insects collects pollen onto some part of its body and rubs this pollen off onto the stigma of another flower. This means the flower has to attract pollinators of the right size and shape to fit the flower. Ideally, the visiting insect should bring and deposit pollen only from another plant of the same species, and should carry the pollen it picks upto another compatible plant.

This complex interplay is the driving force for the diffuse co-evolution of insects and entomophilic angiosperms. The relationships are intermixing of mutualistic and antagonistic factors. The plant obviously wants to ensure pollination at minimum cost. It therefore produces as little pollen and nectar as possible. The pollinator wants to maximize the reward from its visits. If another species offers a
better reward, then the pollinator may switch allegiance, so different flower species are competing among themselves in a community. Some ecologists have argued that flower species have co-evolved to avoid such competition by flowering at different times of the year, or by attracting different species of insects. Both those option are found in the tropical forests. Some species flower in a great burst of showy inflorescence. All the individuals of the species flower together and attract many different species of insect. When the plants have finished flowering, the insects move onto a different species which is coming into flower. Other tropical species produce flowers all year round. The flowers are often small and hidden among the leaves. They are visited by one faithful species of pollinator which hunts out the flowers and is kept alive throughout the year by their nectar production (Chapman and Reiss, 1995).

Obligate hemiparasitism represents a further advancement in a continuum of parasitism types. In contrast to facultative hemiparasites, obligate hemiparasites must attach to a host plant to complete their life cycle. Among the obligate hemiparasites, one can differentiate two types, primitive and advanced. The primitive type includes stem parasites of Loranthaceae, Misodendraceae and some Viscaceae. These plants are photosynthetic xylem feeders, but, being stem parasites, they cannot exist independent of the host plant.

The advanced obligate hemiparasites attach not only to host xylem but also obtain host carbon via phloem connections. Concomitant with this nutritional mode is the loss of photosynthetic function, at least to some degree or during some stage of the life cycle. Examples of this nutritional mode include most species of Cuscuta
(Cuscutaceae), Cassytha (Lauraceae), Phacellaria (Santalaceae), Striga gesnerioides (Scrophulariaceae), and Arceuthobium (Viscaceae). The most extreme manifestation of the parasitic mode can be found among the holoparasites. These plants are totally achlorophyllous (or nearly so), nonphotosynthetic, and obtain all their water and nutrients from host xylem and phloem. Most holoparasites occur on host roots, however, some species of Cuscuta (e.g. C. europaea) are stem holoparasites that have lost RUBISCO, thylakoids, chlorophyll and light-dependent CO$_2$ fixation (Machado and Zetsche, 1990). Some members of Rafflesiales also occur as stem parasites (e.g. Apodanthes, Pilostyles, Rafflesia), but it is likely that these first become established on host roots and continued growth of the endophyte results in relocation to the stem (Kuijt, 1969). Holoparasitism has evolved independently in at least seven lineages: Balanophoraceae, Cynomoriaceae, Hydnoraceae, Rafflesiales, Cuscutaceae, Lennoaceae and Orobanchaceae; however, the taxonomic circumscription and phylogenetic positions of these plants is currently undergoing intensive re-evaluation. Two families, Cuscutaceae and Scrophulariaceae serve as models for studying the evolution of holoparasitism from hemiparasitism because both modes exist among particular species.

**Host - parasite interface**

Two types of host parasite interface are known to occur in balanophoraceae (Kuijt, 1969). In Dactylanthus and Helosis it is a rather simple fluted disk in which a discrete host surface meets a discrete parasite surface. In some areas the host tissues grow more vigorously than in others, so they penetrate more deeply into the parasite position of the tuber, but this extra penetration is never more than about 1
mm deep. Although every parasite cell in the interface layer contacts a host cell, most of the neighbors of any particular parasite cell are other parasite cells. Similarly, most of the cells adjacent to any host cell in the interface are other host cells. In the Balanaophora type, the interface consists of strands that are reported to contain a central mass of host root vascular tissue surrounded by and partially penetrated by parasite absorptive cells (Gedalovich-Shedletzky and Kuijt, 1990).

Understanding the fascinating changes that have shaped the evolution of parasitic plants would be greatly facilitated, from both morphological and molecular standpoints, by detailed comparative studies with their closest nonparasitic relatives. However, the phylogenetic position of parasitic angiosperms and their precise relationship to autotrophic relatives are not easy to deduce (Nickrent et al., 1998). In general, problems are encountered with both morphological and molecular evidence. Parasitism is associated with extreme reduction and modification of vegetative structures, and convergence with other parasitic taxa is common. Both phenomena are encountered in *Cuscuta*. The morphology of this parasitic genus is characterized by loss of roots, significant reduction of chlorophyll synthesis, almost complete reduction of leaves and cotyledons, and the evolution of haustoria, organs that enable these plants to connect to the hosts. Both *Cuscuta* and *Cassytha* are pale, twining stem parasites, and provide an excellent example of convergent evolution in parasitic plants (Kuijt, 1969).

Despite this effort, the phylogenetic affiliation of many paretic groups, especially the so-called nonasterid holoparasites are not known (Nickrent et al., 1998). In many cases the chloroplast and nuclear genes typically used to deduce
large-scale flowering plant relationships (Soltis et al., 1997) are lost, significantly altered, or evolving at greatly accelerated rates, making phylogenetic inferences extremely difficult. In recent progresss, the holoparasitic family hydnoracease was placed as sister to Aristolochiaceae (Nickrent et al., 2002) and enigmatic Rafflesia, genus with the largest known flowers, was found to be a member of Rosids, most closely related to the order Malpighiales (Barkman et al., 2004). In contrast to nonasterid holoplarsites, the general position of most hemiparasites as well as the “asteroid holopparasites” in the global angiosperm phylogenic is not in dispute. However, even in these cases the precise relationship to nonparasitic taxa remains uncertain. For instance, there is little doubt, based both on reproductive morphology and molecular data, that the hemiparasitic genus Cassytha is close associated with Lauraceae, but its placement, ether as a sister group to Lauraceae or nested deeper within Lauraceae, remains uncertain (Rohwer, 2000). Likewise, the small halo parasitic family Lennoaceae was recognized early on, based on floral and pollen morphology, to be closely related to Boranginaceae. Preliminary results of molecular analyses indicated that this family is indeed related to the boraginaceae subfamily Ethretioideae, but its closest relatives remain uncertain (Smith et al., 2000). In certain instances, however, the closest nonparasitic relatives of parasitic plants were ascertained with high supported using molecular data.

Parasitic plants

Parasitic plants are common in natural communities, but are largely ignored in plant community theory. Interaction between parasitic plants and host often parallel those between herbivores and plants: both types of consumers display host
preferences, reduce biomass and alter host allocation patterns, modify plant community structure and dynamics, and mediate interactions between host plants and other organisms. Many fundamental aspects of the ecology of parasitic plants remain poorly studied (Steven and Callaway, 2002).

Parasitic plants number some 3000 species, about 1% of all angiosperms (Atsatt, 1983). They occur within 16 plant families, represent a wide variety of life forms from trees and shrubs to vines and herbs, and are present in most plant communities throughout the world (Musselman and Press, 1995). There are also more than 400 species of heterotrophic vascular plants in 87 different genera that lake chlorophyll and parasitize other plants through specialized mycorrhizal connection (Leake, 1994). Moreover even fully autotrophic plants may often participate in mycorrhizally mediated parasitism (Marler et al., 1999). Despite the widespread prevalence of parasitic plants, and extensive research into their role as agricultural pests (Riches and Parker, 1995), their role in the structure and function of communities has rarely been considered from a theoretical perspective (Smith, 2000).

Despite these foraging patterns, the sedentary nature of parasitic plants constrains their ability to locate hosts, which is likely to result in selection favouring broad host ranges. Host range varies widely among species of parasitic plants (Norton and De Lange, 1999). Single species of Cuscuta and Castilleja can parasitize hundreds of host species in many different families, whereas some species of mistletoe parasitize only one host species. Although parasitic plants with
narrow host ranges exist, they are the exception rather than the rule (Musselman and Press, 1995).

Parasitic plants other than mistletoes and dodder also intensify their effects on hosts by altering host physiology. It was found that 8 species of root hemi parasites transpired at high rates during the night, with some species transpiring more during the night than the day (Press et al., 1988). Host parasitic interactions among animals and between animals and plants have provided models for understanding co-evolution (Yan and Stevens, 1995); studies of parasitic plants as selective forces and of the reciprocal response of the parasite, are far fewer (Norton and Carpenter, 1998). The lake of narrow host ranges among most parasitic plants suggests that co evolutionary selection is weak. Parasitic plants frequently reduce the reproductive output of the hosts (Puustinen and Salonen, 1999). But there are little evidences for directional selection, perhaps because the traits that mediate host choice are poorly understood.

**Food web complexity**

Most studies of diversity-ecosystem function relationships have been performed in single trophic levels, mostly plant communities, and have documented that taxonomic and functional-group diversity influences a range of ecosystem services such as primary production, nutrient fluxes and decomposition rates (Loreau et al. 2002). However, both the determinants of ecosystem services in multitrophic communities and functions performed by higher trophic levels than primary producers have been rarely reported (Wilby and Thomas, 2002). Whereas the theory behind possible effects of plant richness on functions is well developed
(Yachi and Loreau, 1999), the theoretical background on diversity ecosystem function relationship is absent for multitrophic systems, with a very few exceptions (Zheng et al., 1997). Indeed, one challenge is establishing a link between food web theory and research on diversity ecosystem function (Raffaelli et al., 2002).

Although evidence clearly connects consumers and ecosystem functioning, these connections are rarely reported explicitly. Only a few studies have shown how diversity of higher trophic levels affects functions performed at the base of food webs. In some cases, the effects of consumers on lower trophic levels are mediated mostly through indirect interactions, involving multiple trophic levels (Wooton 1994). For instance, the transmission of predatory effects through trophic cascades has important consequences on primary production and nutrient cycling (Schmitz et al., 2000). Food web theory has shown that the pattern of trophic interactions has large effects on community dynamics (Polis and Strong, 1996), specifically the configuration of weak and strong consumer-resource interactions (Neutel et al., 2002).

**Agricultural biodiversity**

The effective conservation and use of agricultural biodiversity is vital for creating and maintaining sustainable increases in the productivity of healthy food for mankind, as well as contributing to the increased resilience of agricultural systems. Major advances in the two main complementary strategies for agricultural biodiversity conservation, namely *ex situ* and *in situ*, over the last decade are presented to reflect on their current global status and trends. Agricultural biodiversity is an important component of biodiversity, which has a more direct link
to the well being and livelihood of mankind than other forms of biodiversity. In fact it is one of our most fundamental and essential resources, one that has enabled farming systems to evolve since the birth of agriculture about 10,000 years ago. Food plant and animal species have been collected, used, domesticated and improved through traditional systems of selection over many generations (Plucknett, 1987). There are many threats or drivers of changes on biodiversity that have been recognized and intensified in recent years (Millennium Ecosystem Assessment, 2005). With regard to agriculture the most important ones include changes in land use, replacement of traditional varieties by modern cultivars, agricultural intensification, increased population, poverty, land degradation and environmental change (Van de Wouw et al., 2009).

There are two main strategies for conserving agricultural biodiversity, namely ex situ and in situ conservation, both of which are equally important and should be regarded as complementary (Thormann et al., 2006). Ex situ conservation is the conservation of components of biodiversity outside their natural habitats (UNCED, 1992). It is generally used to safeguard populations that are at present or potentially under threat and need to be collected and conserved in genebanks in the form of seeds, live plants, tissues, cells and/or DNA materials. Article 2 of the CBD defines in situ conservation as “the conservation of ecosystems and natural habitats and the maintenance and recovery of viable populations of species in their natural surroundings and, in the case of domesticated or cultivated species, in the surroundings where they have developed their distinctive properties” (UNCED, 1992). It thus refers to the maintenance of a species in its natural habitat. This can
be either on farm, requiring the maintenance of the agro-ecosystem along with the cultivation and selection processes on local varieties and landraces, or in the wild, which involves the maintenance of the ecological functions that allow species to evolve under natural conditions.

Little is known about the global status of agricultural biodiversity. Although the CBD recognize genetic diversity as one of the fundamental levels of biodiversity, actions to protect genetic diversity are lacking (Laikre et al., 2010). It was also provided some examples of empirical work that demonstrates how populations and even species can collapse due to loss of genetic diversity. It also provides evidence supporting the importance of maintaining genetic variation to sustain species and ecosystems Policy makers and scientists require a better understanding of how the intraspecific diversity is changing over time and space in order to make informed decisions for their conservation. However there is no routine global scale monitoring of genetic diversity over time (Frankham, 2010), except for a few target species at national level (Laikre et al., 2008). A major challenge remains to develop simple inexpensive means to monitor genetic diversity at global scale efforts under the 2010 (Frankham, 2010).

**Biodiversity and Economic Aspect**

Economic growth is an increase in the production and consumption of goods and services, and it occurs with increasing population or increasing per capita production and consumption. An increasing Gross Domestic Product (GDP) indicates a growing economy. Gross domestic product is a measure of the production of goods and services, income derived from that production, and
expenditure thereon (Abel and Bernanke, 2003). In an ecosystem surplus production by producers supports the existence of primary-, secondary-, and higher-level consumers. In the human economy surplus production by agricultural and extractive agents is required to support the existence of manufacturing sectors (Boulding, 1993).

Some species escape easy classification among the standard trophic levels of producers or consumers. For example, detritivores consume decayed biomass and thus help maintain the ecosystem and its other species. They may be described as service providers in the economy of nature. In the human economy, too, there are service providers that interact with the agricultural, extractive, and manufacturing sectors (e.g., janitors, waiters, and information providers).

A convenient way of envisioning the basic conflict between economic growth and biodiversity conservation is by integrating the economy of nature and the human economy, in the highest trophic level. Humans have the unique, omnivorous ability to identify, harvest, and consume virtually all other edible species. When the human economy grows, the effect is a trophic compression of nonhuman species.

India’s Biodiversity Act

In order to implement the India’s Biological Diversity Act (2002) (BD Act), in its Section 8, a National Biodiversity Authority (NBA) was established in the year 2003. It provides provisions for regulated access to biological resources by bonafide end-users for various purposes including scientific research, commercial activities and sustainable use of non-timber forest produce. The Government of
India brought the CBD into force from 19th May 1994. This convention provides a framework for the sustainable management and conservation of India's natural resources. In order to regulate access to biological resources of the country with the purpose of securing equitable share in benefits arising out of the use of biological resources and associated knowledge, to conserve and sustainably use biological diversity legislation was required. Legislation was also required in order to respect and protect traditional knowledge of local communities and to secure benefit sharing with local people who have conserved the biological resources and inherited knowledge and information relating to their use of biological resources.

The Act stipulates norms for access to biological resources and traditional knowledge in three ways: (i) Access to biological resources and traditional knowledge to foreign citizens, companies and non-resident Indians (NRIs) based on ‘prior approval of NBA’ (Section 3, 4, 6 of the Act and Rule 14-20); (ii) Access permits to Indian citizens, companies, associations and other Organizations registered in India on the basis of ‘prior intimation to the State Biodiversity Board’ concerned (Section 7 of the Act). (iii) Exemption of prior approval or intimation for local people and communities, including growers and cultivators of biodiversity, and Vaids and Haqims, practicing indigenous medicines (Section 7 of the Act). The key procedures to be followed for access to biological resources and traditional knowledge are dealt with under Rule 14 of the Biodiversity Rules 2004.