FLORISTIC DIVERSITY OF KASHMIR HIMALAYAN GRASSLANDS IN RELATION TO THEIR FUNCTIONING

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CERTIFICATE

Certified that the thesis entitled "*Floristic diversity of Kashmir Himalayan grasslands in relation to their functioning*", embodies the results of original research work carried out by Mr. Anzar Ahmad Khuroo under our joint supervision. The thesis is forwarded for acceptance in fulfilment of the requirements for the award of Ph. D. Degree in Botany.

The work has not been submitted so far, for this or any other degree, of this or any other university.

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Dated: *15-04-2013* ***Anzar A Khuroo***

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ABBREVIATIONS USED

Grassland sites:

BG = Kashmir University Botanic Garden

DD = Draphoma Dachigam

DW = Dugwan Dachigam

SP = Sopore

GT = Gualta Uri

KK = Kamalkote Uri

GM = Gulmarg

PD = Pahlipora Dachigam

PG = Pahalgam

SM = Sonamarg

BT = Baltal

TJ = Thajwas

DB_GZ = Dignibal grazed

DB_UZ = Dignibal ungrazed

AR_LP = Aru long protected since 1986

AR_FP = Aru protected since last five years

AR_GZ= Aru grazed

Floristic diversity:

SR (actual) = Species Richness (actual)

SR (obs.) = Species Richness (observed)

SR (est.) = Species Richness (estimated)

Sp. = Species (singular)

Sps. = Species (plural)

Spp. = Species (plural)

SAC = Species Accumulation Curve

SRC = Species Rarefaction Curve

CI = Confidence Interval

LB = Lower Bound

UB = Upper Bound

ICE = Incidence-based Coverage Estimator

ACE = Abundance-based Coverage Estimator

MM = Michaelis-Menten estimator

Shannon Exp. = Shannon Exponential

Months of the year:

Jan = January

Feb = February

Mar = March

Apr = April

May = May

Jun = June

Jul = July

Aug = August

Sep = September

Oct = October

Nov = November

Dec = December

Biomass and primary productivity:

AG = Above-ground live biomass

SD = Standing dead biomass

LT = Litter

BG = Below-ground biomass

ANP= Aboveground Net Primary Productivity

BNP= Belowground Net Primary Productivity

TNP= Total Net Primary Productivity

NPP= Net Primary Productivity

AGp = Peak-season above-ground live biomass

SDp = Peak-season standing dead biomass

LTp = Peak-season litter

BGp = Peak-season below-ground biomass

ANPp = Peak-season aboveground Net Primary Productivity

BNPp = Peak-season belowground Net Primary Productivity

TNPp = Peak-season total Net Primary Productivity

NPPp = Peak-season Net Primary Productivity

$g\ m^{-2}$ = grams per square meter

$g\ m^{-2}\ y^{-1}$ = grams per square meter per year

The effects of biodiversity on ecosystem functioning (hereafter BEF) has long been one of the hotly debated topics and has consequently seen a flurry of research activities (Chapin *et al.* 1997; Naeem *et al*. 1999; Loreau *et al*. 2001; Hooper *et al*. 2005). In fact, during the last two decades, the BEF has emerged as a dominant issue in the biodiversity research (Sutherland *et al*. 2013). A large body of observational and experimental data support the idea that diversity supports ecosystem functioning and its relevance for the contemporary conservation concerns (Kinzig *et al*. 2002; Loreau *et al.* 2002; Srivastava, 2002). It is expected that the research insights gained from this field will allow biologists to make expert predictions regarding the consequences of species extinction on the ecosystem goods and services, on which ultimately depends the human existence. Most importantly, the BEF research findings have gradually influenced the global environmental policy-making, largely justifying biodiversity conservation in order to maintain ecosystem functioning.

The BEF hypothesis is based on the assumption that a decline in biodiversity will alter ecosystem-level processes. It represents a synthetic field of biodiversity research that seeks to understand how changes in species composition, distribution and abundance change the ecosystem functioning (Schulze & Mooney, 1993; Kinzig *et al*. 2002; Loreau *et al*. 2002; Naeem *et al*. 2002). As changes in the biodiversity are worryingly rampant from global, regional to local scales, research findings from this field have received considerable attention (Naeem *et al.* 1994; Hooper & Vitousek, 1997; Naeem & Li, 1997; Tilman *et al*. 1997; Wardle *et al.* 1997; Emmerson *et al.* 2001; Pfisterer & Schmid, 2002). At present, the question is no longer whether biodiversity matters, but how it matters (Rosenfeld, 2002). In the BEF research, biodiversity is most often measured as the number of species (i.e., species richness), although a few studies have incorporated species evenness also (Wilsey & Polley, 2004); while as the primary productivity is the most commonly used measure of ecosystem functioning. Research studies have clearly demonstrated a definite relationship between plant species richness and primary productivity; and thus the conservation of biodiversity is essential to maintain productivity under variable adverse environmental conditions (Tilman & Downing, 1994).

Much of the research work on the effects of biodiversity on ecosystem functioning has focused on the grasslands, because they are the easy ecosystems to manipulate and their aboveground net primary productivity is relatively easy to approximate, as most of the aboveground biomass is generally accrued during a single year. The grasslands, like other natural ecosystems, represent a natural capital asset that supports domestic livestock, game, and provide fiber, water for drinking and irrigation; and as provisions serve as watershed (infiltration, purification, flood control, soil stabilization), nutrient cycling, biodiversity, atmospheric carbon, human and wildlife habitat, and recreation (Austrheim & Eriksson, 2001). The grassland biome covers *ca.* 25 percent of the land surface of the earth (Sala *et al.* 1996). These are systems mostly limited by water, which are dominated by grasses and have a variable woody component.

Humans utilize these ecosystems as grazing-lands or transform them into croplands depending mostly on water availability. Most of the mesic grasslands have been converted into agricultural land, whereas a large fraction of the arid and semi-arid grasslands remain as such. Grasslands produce an array of goods and services for humankind, but only a few of them have market value. Meat, milk, wool, and leather are the most important products currently produced in grasslands that have a market value. Simultaneously, grassland ecosystems confer to humans many other vital and often unrecognized services such as maintenance of the composition of the atmosphere, maintenance of the genetic library, amelioration of weather, and conservation of soils. In many cases, the value of services provided by grasslands in terms of production inputs and sustenance of plant and animal life may be larger than the sum of the products with current market value (White *et al.* 2000). Unfortunately, the grasslands throughout world have been subjected to considerable anthropogenic pressures which have drastically reduced the provision of essential goods and services from these ecosystems (Sala *et al.* 2000).

Kashmir Himalaya, located in the north-western folds of the Indian Himalayas, has vast land area (*ca*. 16%) under grasslands which play an important role in providing economic goods and ecosystem services to the society. Livestock, particularly the migratory flocks, are entirely dependent on these grasslands. They serve as bedrock for sustaining the core economic activity of livestock rearing in the region (Masoodi, 2003; Anonymous, 2012). Apart from sustaining this pivotal economic activity, grasslands harbour a rich and endemic biodiversity, and regulate the regional carbon, nutrient and hydrological cycles.

In spite of the above-highlighted indispensable roles of grasslands in the region, very few research studies have been carried out on the structural and functional aspects of these ecosystems (Gupta & Kachroo, 1981; Bhat & Kaul, 1989). Such constraints in the availability of up-to-date baseline data are the first and formidable impediment in the formulation of practicable conservation strategies and management plans for these ecosystems. Thus, to surmount these constraints, the present study was envisaged on the Kashmir Himalayan grasslands with the following objectives.

- To study the floristic diversity in Kashmir Himalayan grasslands.
- To estimate the primary productivity of Kashmir Himalayan grasslands.
- To investigate the relationship between floristic diversity and primary productivity in Kashmir Himalayan grasslands
- To determine the impacts of livestock grazing on floristic diversity and primary productivity in Kashmir Himalayan grasslands.

GRASSLANDS

By definition, the grasslands encompass the regions which are covered by natural or semi-natural herbaceous vegetation, predominantly grasses, with or without woody plants (Singh *et al*. 1983). In the subtropical and temperate regions, shrubs often contribute the woody component to grasslands. Several types of grasslands, such as savannahs, have woody plants as an important element of their natural vegetation. Globally, the distribution of grassland ecosystems is mainly determined by the climatic variables: chiefly temperature and precipitation (Whittaker & Likens, 1975). In effect, grasslands are located in those regions of the world where availability of precipitation during sometime of the year falls much below the requirement of forest but is sufficient to support herbaceous vegetation as the dominant plant forms; thus, grasslands can be said to represent an intermediate stage between forests and deserts.

Grasslands are among the largest ecosystems in the world, covering *ca.* 30 percent of the Earth's surface (Sala *et al*. 2001). They are found in every region of the world; Sub-Saharan Africa and Asia having the largest total area under grasslands, 14.5 and 8.9 million sq km, respectively. Commonly, they occur in the semi-arid zones (28% of the world's grasslands), followed by humid (23%), cold (20%), and the arid zones (19%).

Grasslands contribute directly to the livelihoods of more than 800 million people. They are precious sources of goods and services, such as food and forage, energy and wildlife habitat; and also provide carbon and water storage and watershed protection for many major river systems. They are important for *in situ* conservation of genetic resources. Most commonly, grasslands are used to feed the livestock. From cows and buffaloes, sheep and goat herds, to horses and mules, grasslands support large numbers of domestic animals, which constitute the sources of meat, milk, wool, and leather products for humans. They also support large numbers of wild herbivores that depend on grasslands for breeding, migratory, and wintering habitat. Grassland biodiversity encompasses a wide range of goods useful to humans (House & Hall, 2001). The World Conservation Monitoring Centre (WCMC, 1992) ranked the world's grasslands in the following order of decreasing importance as the repositories of biodiversity: African Savannah, Eurasian Steppe, South American Pampas, North American Prairie, Indian Savannah, Australian grasslands.

Grasslands are the seedbed for the ancestors of major cereal crops, including wheat, rice, rye, barley, sorghum, and millet. They continue to provide the genetic material necessary to breed cultivated varieties that are resistant to crop diseases. Grasslands also provide habitat for plants and animals - soil microfauna, birds and large mammals alike. Worldwide, almost half of 234 Centers of Plant Diversity (CPDs) include grassland habitat. About 23 of 217 Endemic Bird Areas (EBAs) include grassland as the key habitat type. Out of the 136 terrestrial ecoregions identified as outstanding examples of the world's diverse ecosystems, 35 are grasslands. About 4,500 relatively large protected areas comprise at least 50% grassland; protected grasslands cover approximately 4 million sq km or 3 percent of the total land area, which represents just 7.6% of the total grassland area. Grasslands store *ca.* 34% of the global stock of carbon in terrestrial ecosystems, while forests store *ca.* 39% and agro-ecosystems *ca.* 17%. Unlike tropical forests, where vegetation is the primary source of carbon storage, most of the grassland carbon stocks are in the soil. Grasslands are particularly captivating for viewing game animals and for safari hunting. People get attracted to the large mammals, birds, diverse plant life, and generally open-air landscapes of grasslands (White *et al*. 2000).

Generally, there are three main factors - drought, fire and grazing - which determine characteristic features of the grasslands and distinguish them from other types of ecosystems (Milchunas *et al*. 1988). These factors provide the selection pressure for a relatively higher turnover of the aboveground plant parts, location of perennating organs near the soil surface, and for an important role of belowground biomass. However, both the nature and magnitude of the changes brought about by these factors on structure and function of grasslands depends upon the local conditions of a particular region. Furthermore, the *true* disturbance for grasslands is a lack of disturbance, because disturbance is an intrinsic characteristic of grassland ecosystems. The natural disturbance in grasslands is chiefly in the form of environmental fluctuations, which can be recognized as integral stochastic factors; some of these are grazing by herbivores, precipitation on seasonal, annual and decadal times, and seasonal fire.

In terms of evolutionary history, the environmental fluctuations have played a fundamental role in the evolution and maintenance of grassland biodiversity (McNaughton, 1983). Major characteristics of the grasslands: species richness and composition, plant life-forms, vegetation structure and dynamics, and primary productivity, intimately interact with herbivore grazing, and play a critical role in determining biodiversity and ecosystem functioning in the grasslands (Proulx & Mazumder, 1998; Grace & Jutila, 1999). For instance, the diversity of annual plant life-forms in the grasslands is determined by the interaction between grazing and small-scale spatial and temporal variation in primary productivity, operating mainly on the less abundant species in the grassland community (Osem *et al*. 2002). Likewise, while examining the effects of grazing on species composition changes in the Earth’s grasslands, Milchunas and Lauenroth (1993) concluded that the high primary productivity, generally associated with tall-statured grasses, was associated with significantly greater changes in species composition when grazed, as the tall-statured species were replaced by short-statured and more grazing-tolerant grasses.

In recent times, the spatial extent, structure, and composition of grasslands have been significantly altered. Modifications include anthropogenic changes, such as cultivation, urbanization, desertification, fire, livestock grazing, fragmentation, and introduction of invasive alien species. Temperate grasslands and savannas have experienced heavy conversion to agriculture, more so than other grassland types. More than half of the grassland area has some degree of soil degradation; over 5% of these grasslands are extremely degraded. Grassland biodiversity has seen marked declines, with negative effects from fragmentation and invasive alien species (White *et al*. 2000).

Presently, there are few surviving primary grasslands in the world, as most of these have been transformed into other land-use types by humans. The transformation of grasslands largely to agriculture lands has led to loss of once-extensive expanses of natural grasslands in all the continents. Worldover, overgrazing by livestock, and other unscientific management practices, have degraded grasslands severely. When compared with estimates of their extent before significant anthropogenic disturbances, grassland area has declined by 49% world-over (Klein Goldewijk, 2001). In the tropical regions, about 74% of the grassland area is still intact, with little anthropogenic disturbances. However, in temperate regions, only 27.6% of grasslands are still left as undisturbed areas, while the rest have been either partially or completely disturbed by extensive agriculture or human settlements – often associated with destruction of primary vegetation (Hannah *et al*. 1995).

Although the major goods and services provided by grasslands are still in good condition, the capacity of grassland ecosystems to continue to provide these goods and services is declining. Cultivation and urbanization of grasslands, and other modifications of grasslands through desertification and livestock grazing, can be a significant source of carbon emissions. Biomass burning, especially from tropical savannas, contributes over 40 percent of gross global carbon dioxide emissions. The introduction of invasive alien grassland plant species may decrease the total carbon storage because they have less extensive below-ground root networks for storing organic matter as compared to the native grassland plants. The invasive alien species can negatively affect grassland ecosystems through species competition and can eventually lead to decreases in biodiversity. Some North American grasslands support 10 percent to 20 percent alien plant species. Tourism and recreational activities in grasslands make important economic contributions. However, overuse and declines in wildlife populations suggest possible declines in the capacity of grasslands to continue providing these services (White *et al*. 2000)

Notwithstanding the fact that temperate grasslands of the world are among the most diverse and productive terrestrial biomes, yet they have received the least protection. According to an estimate, world-over only 0.69% of the temperate grassland biome is under some sort of protection (Rawat, 1998) Amongst the temperate grasslands, the Himalayan grasslands in high-altitude regions of South Asia are ecologically fragile and socio-economically valuable ecosystems, though relatively less-known and little investigated than their tropical counterparts.

The Himalaya, one of the global biodiversity hotspots, is the most astounding physical feature on the surface of the Earth. It is well known not only for its diverse natural landscapes but equally for the biological, hydrological, socio-cultural, and aesthetic values. Of the present-day vegetation, natural and semi-natural grasslands of the Himalaya are of particular interest due to their relatively recent origin, dynamics and close co-evolution with grazing herbivores. In fact, the geological history of these grasslands began with the progressive uplift of the Himalaya which increased aridity. Subsequently, introduction of cattle, fire and widespread impact of humans over the last 5,000 years have reduced the primary vegetal cover over large tracts of land (Blow & Hamilton, 1975). Clearing and opening of the forests for various land-use practices, frequent burning of steeper south-facing slopes for the production of hay, and intensive livestock grazing have converted a considerable area under herbaceous vegetation. Such areas include the forested blanks in humid areas, mid-elevation hay fields, fallow-lands, and village grazing grounds. Such grasslands are variously referred to as 'rangelands', 'grazinglands', 'pasturelands' or simply 'pastures'.

In the Indian Himalaya, the grasslands occupy about 35% of the geographical area. The different types of the Himalayan grasslands include the warm temperate grasslands, cold temperate and subalpine grasslands, alpine meadows, the steppes of cold-arid regions, and the alpine scrub. Although, they differ from one another in terms of origin, structure and composition; nevertheless they all support a large number of wild herbivores, domestic livestock, and agro-pastoral activities (Rawat, 1998).

BIODIVERSITY

Biological diversity, or its portmanteau – biodiversity, simply refers to the variety and variability of life. Many definitions of biodiversity have been proposed (DeLong, 1996). Of these, one of the all-embodying and widely referred to is that enshrined in Article 2 of the Convention on Biological Diversity (CBD), which states that: *'Biological* *diversity' means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems'.*

Thus, biodiversity encompasses a broad unifying concept which includes all levels, forms and elements of natural variation in the world's biota. More commonly, biodiversity is recognized as the variation at different levels of biological organization from genes through species to ecosystems; and because of this reason, it is often distinguished at three main levels: genetic diversity, species diversity and ecosystem diversity. However, these levels of biodiversity are intimately linked and essentially form multifaceted nested hierarchy. To define these levels of biodiversity objectively may not only be difficult but, in some cases, may have no biological reality at all. Nonetheless, they remain workable and indispensable tools for understanding the concept of biodiversity; species richness, in particular, has been much used due to its convenience in use. Quite often, for the ease of understanding, biodiversity is partitioned into the floral-, faunal-, fungal-, microbial-diversity, etc. depending upon the taxonomic domain of biota under consideration.

The first and foremost requirement in the biodiversity research is to select a scientific measure/quantifier of the biodiversity. It becomes critically important because the measurement of biodiversity is essential for understanding the ecosystem functioning, conservation of biodiversity and ecosystem management. However, owing to the conceptual broadness, only measures for the isolated components of biodiversity have been developed with underlying caveats. It is clear even from the CBD's definition that no single measure can adequately capture the broad concept of biodiversity.

In the contemporary biodiversity research, the relationship between biodiversity and ecosystem functioning is receiving a great deal of attention (Kinzig *et al*. 2002; Loreau *et al*. 2002). Since the outcome of such research studies strongly depends on the type of diversity measure used, a complete understanding of the various biodiversity measures developed assumes importance. In fact, biodiversity is a comparative science and the primary goal of biodiversity measurement is usually to compare or rank communities/habitats/ecosystems. Commonly, a researcher/land manager wants to know whether biodiversity has changed or not over space and time. If changed, then what are the causes/processes responsible for such changes? It becomes, therefore, crucial to know which entities should be compared, and at what scale (Harper & Hawksworth, 1995; Wilson & Chiarucci, 2000).

Quite often, diversity at the level of species has been used for the measurement of biodiversity. Species diversity measures − a rough proxy for biodiversity − are the traditional way of quantifying the biodiversity. It can be partitioned into two main components: species richness and species evenness (Simpson, 1949). Species richness is simply the number of species of a given taxon in the unit of study. Species evenness, on the other hand, describes the variability in species abundances in the unit of study. For example, a community in which all the species contribute equal numbers of individuals would be rated as extremely even. Conversely, a large difference in the relative abundances of species would result into the community being rated as the least even. In other words, species richness and species evenness can be equated with the community size and shape, respectively (Rao, 1982). In practice, measures of species-level diversity (e.g. species richness) are not applied to all the species at a sampling site but rather to a particular taxonomic group (e.g. vascular plants, birds, mammals, etc.). This reflects the very fact that different sampling techniques and scientific protocols are required to study diversity of different taxonomic groups; it also may be partly due to the specific scientific or conservation goals.

In most of the research studies, species diversity measurement is based on three assumptions: i) all the species are equal, ii) all the individuals are equal, and iii) the species abundance has been recorded using appropriate and comparable units.

Species Richness (SR)

The term “species richness” was first coined by McIntosh (1967). Although, at least theoretically, biodiversity can be estimated by diverse measures, it is most commonly measured in terms of species richness (SR). Species richness is taken as universal currency for the study of biodiversity and has proven valuable, both for theoretical and practical purposes. Some of the main reasons which make the species richness a popular measure of biodiversity are: a) the availability of information on global species numbers to some extent, b) the practical utility, such as biodiversity surrogates, and c) above all the wide applications (Gaston, 1996).

Being the oldest measure of biodiversity, species richness represents one of the simplest measures too. However, the species richness has limitations too. Firstly, there is an intense debate about which species concept should be adopted owing to the classical taxonomic problem: “lumpers” decrease SR, while the “splitters” inflate it (Gaston, 1996). Secondly, the identification of synonyms (when two or more scientific names are applied to a single species by different authors, only one scientific name is valid as per ICBN rules) can reduce the SR (Gaston & Mound, 1993). Thirdly, as of now, more than 75% of the extant species remain to be taxonomically described (May, 1990). Furthermore, sampling brings further complications. As we know that it is practically impossible, and rarely cost-effective, to record every species within a community. Hence, an effective sampling protocol must take into consideration the underlying species abundance distribution; and accordingly greater effort is required in the sites where evenness is low (Lande *et al*. 2000; Yoccoz *et al*. 2001). All such caveats should be considered while measuring the SR within a community.

There are two main approaches used to express the estimates of species richness:

i) Numerical species richness: it computes the number of species per specified number of individuals, and is mostly used for animal taxa where individuals are readily identifiable.

ii) Species density: it calculates the number of species per specified sampling unit, and is mostly preferred for plant studies, e.g., the number of species per meter square.

Gotelli & Colwell (2001) have clearly highlighted the distinction between individual-based assessment protocols, where individuals are sampled sequentially, and sample-based assessment protocols, in which sampling units, such as quadrats, are identified and all the individuals that lie within the unit are counted. The occurrence-based (or incidence) data provides an additional method of estimating species richness. Occurrences refer to the number of sampling units, such as quadrats, in which a species is present, and represents actually the species density data in another form.

In fact, one of the most daunting challenges for the global scientific community is to know about the total number of species that presently exist on the Earth. This has received increased attention in recent times, in view of growing concerns about global biodiversity loss. Notwithstanding such a huge challenge, compiling inventories of global species richness is too costly, frustratingly laborious and perennially subjected to sampling size biases (Gaston, 1996). Since the extent of global biodiversity is often inferred from the magnitude of species richness at local scales, methods for estimating species richness through extrapolation have proved helpful. As of now, three main approaches have been used for estimating species richness from the samples (Colwell & Coddington, 1994; Chazdon *et al.* 1998), which are as follows:

(i) **Species accumulation curves**: They plot the cumulative number of species recorded (S) as a function of sampling effort (n). Sampling effort can be the number of individuals collected, or the cumulative number of samples. The classical species-area curves, widely employed in plant science research, are one such form of species accumulation curves.

The sequence in which samples (or individuals) are included in a species accumulation curve can influence its overall shape. A sample with relatively higher number of species will have a much greater influence on the shape of the curve, if it is included earlier rather than later in the sequence. In such a situation, a smooth curve can be produced by randomization, wherein samples (or individuals) are randomly added to the species accumulation curves, and this procedure is repeated, e.g. 50 times. The mean and standard deviation of species richness at each value of (n) can also be computed. As the new species are added, the species accumulation curves proceed from left to right. In this way, the curves can be extrapolated to provide a reasonable estimate of the total richness of the community. The functions used in this type of extrapolation may be either asymptotic or non-asymptotic, by virtue of which increase in species richness for additional sampling effort can be easily predicted, rather than to estimate total species richness per se.

(ii) **Parametric methods**: Theoretically, it is possible to estimate the overall species richness, if the shape of a species abundance distribution model can be satisfactorily described. In this context, the two species abundance models that have been widely used are the log series and log normal distributions; and out of these, the former is the easiest to fit and the simplest to apply.

(iii) **Non-parametric methods**: Many of the non-parametric methods have been devised by Chao & Shen (2004). Unlike the preceding ones, they are simply not based on the parameter of a species abundance model. Being efficient, easy to understand and to use in the field, they offer many advantages in biodiversity measurement at present. Recently, Colwell’s (2001) Estimate S software program has increased their usage.

Furthermore, in real world, it is almost impossible to sample so intensively in order to achieve even a rough estimate of species number. In such a scenario, it becomes important to search for indirect surrogate approaches for identifying areas with potentially high species richness, and also for ranking sites along a rich-poor gradient. Broadly, there are 3 main types of surrogacy methods, as given below:

(i) Cross-taxon: here, high species richness in one taxon is used to infer high richness in others (Mortiz *et al.* 2001).

(ii) Within-taxon: here, higher taxa such as generic or familial richness are treated as a surrogate of species richness (Balmford *et al*. 1996).

(iii) Environmental: here, parameters such as temperature or topographical diversity are taken to track species richness (Gaston, 1996).

Despite some disadvantages of surrogacy methods, they have become popular and successfully characterized and mapped the species richness gradients in many cases (Williams & Gaston, 1994; Brown & Albrecht, 2001).

Species evenness

Besides species richness, natural communities also differ in abundances, and hence the additional dimension of species evenness has been used in the measurement of biodiversity. Species evenness, simply, is a measure of how the species in a community are similar / different in their abundances. Therefore, a community with equally abundant species has high species evenness. The opposite of evenness is dominance that measures the extent to which one (or a few species) dominate the community. Traditionally, high species evenness (i.e., equivalent to low species dominance) has been equated with high diversity.

Measures of diversity

There is no denying the fact that the biodiversity - being a multifaceted concept - cannot be measured by a single index. That is why, it is always feasible to first decide which facet of biodiversity an investigator desires to measure and then select the index accordingly. There is a plethora of indices from which to select and this cornucopia of diversity measures makes it often difficult to select the best estimator. Therefore, it is crucial to know in advance which aspect of biodiversity is being investigated, and then to select the best available index for capturing that aspect. Also, sampling size must be sufficient to meet the objectives of the investigation. As replication allows statistical analysis, it is of utmost importance. A rule of thumb is to have many small samples rather than a single large one.

Conventionally, diversity measures are classified as species richness measures and heterogeneity measures; the latter combines the richness and evenness components of diversity. Also evenness measures have been developed which separate evenness component of the diversity.

One of the most-appealing and long-enduring of all diversity measures is the Shannon index. The index is based on the rationale that diversity, or information, in a natural system can be measured in a similar way to the information contained in a message. It assumes that the individuals are randomly sampled from an infinitely large community (Pielou, 1975), and that all species are represented in the sample. The Shannon index is calculated from the equation:

$$H' = -\sum P_i \ln P_i$$

The quantity P_i is the proportion of individuals found in the ith species. In a sample, although the true value of P_i is unknown, but is estimated using its maximum likelihood estimator, n_i/N (Pielou, 1969). The value of the Shannon index usually falls between 1.5 and 3.5, and rarely exceeds 4 (Margalef, 1972).

Since the Shannon index takes into account the degree of evenness in species abundances, it has been used to derive a separate evenness measure. Theoretically, the maximum diversity (H_{max}) could occur where all species have equal abundances. In such a situation, if $H' = H_{max} = \ln S$, the ratio of observed diversity to maximum diversity has been used to measure evenness (J') (Pielou, 1975).

$$J' = H'/ H_{max} = H'/ \ln S$$

One of the long-persisting problems that remained with Shannon index is that it confounds two aspects of diversity: species richness and species evenness. In other words, it becomes always difficult to disentangle whether an increase in the index arose either due to greater richness or greater evenness or both. To overcome this, Hayek & Buzas (1997) stated that the Shannon index is simply the sum of the natural log of evenness [ln (E)] and the natural log of species richness [ln (S)]. Based on this, the index can be decomposed into two components: $H' = \ln S + \ln E$. Such an equation to derive H' allows the investigator to infer whether the changes in diversity are either due to richness or evenness, or both. The equation, referred to as the SHE analysis, essentially reflects the relationship between S (species richness), H (diversity) and E (evenness). It can also help in better understanding of the underlying pattern of species abundance distribution (SAD).

Another group of diversity indices are weighted by abundances of the commonest species, and are usually referred to as either dominance or evenness measures. One of the earliest and the best known dominance measures is the Simpson's index (D).

$D = \sum [n_i(n - 1)] / N (N-1)$

Where n_i = the total number of individuals in the ith species; and N = the total number of individuals. Simpson's index is usually expressed as 1-D or 1/D. As the 'D' increases, diversity decreases. The index is heavily weighted towards the most abundant species in the sample, while being less sensitive to species richness. It is one of the most meaningful and robust diversity measures available, and captures the variance of the SAD.

Although Simpson's diversity measure emphasizes the dominance, yet it cannot replace an evenness measure. A separate measure of evenness is calculated by dividing the reciprocal form of Simpson index by the number of species in the sample (Krebs, 1999).

$E1/D = (1/D) / S$

The measure ranges from 0 to 1.

Species abundance distributions (SADs)

In recent times, it is increasingly emphasized that the distribution of species abundance possesses the maximum amount of information about a community's diversity. The species abundance distributions (SADs) can provide clues to the processes that determine the biodiversity of a community. Tokeshi (1993), while strongly advocating for the study of species abundance relationships, underlined that "if biodiversity is accepted as something worth studying, then species abundance patterns deserve equal attention".

Traditionally, investigators have approached the SADs in different ways. The rank/abundance plot, first proposed by Whittaker in 1965, is one of the best known and the most informative approaches. In the rank/abundance plots, the species are plotted in sequence from the most- to the least-abundant along the x-axis. The species abundances are depicted in a $\log_{10}$ format on the y-axis, so that species whose abundance spans several orders of magnitude can be easily accommodated on the same graph. Mostly, proportional or percentage abundances are used that facilitates comparison between different data sets. In other words, abundance of all the species together is given a value of 1.0 or 100%, and then the relative abundance of each species is given as a proportion or percentage of the total.

In fact, Krebs (1999) suggested that the first thing an investigator should do with species abundance data is to plot them as rank/abundance graphs. Then the shape of the rank/ abundance plot can be used to infer which species abundance model best fits the data. For instance, the steep slopes hint towards communities with high dominance as expected in a geometric- or log-series model; while as shallower slopes signify the higher evenness consistent with a log-normal or even a broken-stick model. An important merit of rank-abundance plot is that the contrasting patterns of species richness are easily discernible and can also highlight differences in the species evenness amongst communities (Nee *et al*. 1992; Smith & Wilson, 1996). In particular, when there are relatively fewer species, information related to their relative abundances is clearly visible (Wilson, 1991). Because of this, the rank-abundance plots have been used as an effective method for showing changes following an environmental impact.

Currently, several plotting methods have been developed for presenting species abundance data. Each of these plotting methods highlights different characteristic of the species abundance data. An earnest need has been felt for the standardization of the plotting methods in order to facilitate the comparison of different data sets (May, 1975; Magurran, 1988).

Species Abundance Distribution Models

In conjunction with proliferation of the plotting methods, diverse patterns of species abundance distributions observed in the biotic communities have stimulated the development of species abundance distribution models. Consequently, a wide range of species abundance models are currently available. Not surprisingly, none of these models is universally applicable to all biotic communities, because both species richness and species evenness vary amongst the communities.

Broadly, two categories of SAD models are recognized, as stated below:

(a) **Statistical models**: these models have been initially devised as an empirical fit to the observed data. They enable the investigator to objectively compare different communities. Two well-known statistical models − the log normal and log series − continue to stand the test of time.

(b) **Biological models**: these models have the power to explain, rather than merely describe, the relative abundance of species in a community. They basically represent different scenarios of niche apportionment, and help in predicting how the available niche space might be apportioned (i.e. shared) amongst the constituent species by asking whether the observed species abundance match this prediction.

Although it is convenient to classify species abundance models as statistical or biological, in reality the distinction can not be so strict. Several of the statistical models, particularly the log series and log normal, have acquired biological explanations since their original formulation.

Taxonomic diversity

During the last two decades, taxonomic diversity has gained importance in the biodiversity studies due to its utility in setting conservation priorities (Vane-Wright *et al*. 1991; Williams, 1996), and in environmental monitoring (Warwick & Clarke, 1995). The measures of taxonomic diversity are possible only if the taxonomy of the taxa in the sampling unit is well-established. For instance, if two communities have the same species richness, evenness, and also show similar SAD, but differ in diversity of taxa to which the constituent species belong, then the community with taxonomically more varied species is more diverse. In other words, taxonomic diversity in a community will be higher where species belong to many genera (or other higher taxonomic category), as opposed to one where majority of species belong to the same genus (or other higher taxonomic category).

For measuring the taxonomic diversity, the methods based on the topology of a taxonomic tree have been devised, wherein branch length within a tree is summed up (May, 1990; Vane-Wright *et al*. 1991; Williams *et al*. 1991; faith, 1992). Taxonomic distinctness (TD) measure, which summarizes the pattern of taxonomic relatedness in a sample, is a promising method for measuring taxonomic diversity (Clarke & Warwick, 1998). The TD is a natural extension of Simpson's index, and shows the robustness under variable or uncontrolled sampling effort. An added feature of the TD is that the taxonomic evenness of the sample is also accounted for (Clarke & Warwick, 1999).

Functional diversity

Mainly spurred by the current debate on the role of biodiversity in the ecosystem functioning, the functional diversity measure has gained wide attention in recent times (Diaz & Cabido, 1997; Chapin *et al*. 2000). It is based on the rationale that the communities with the same number of species, but one possessing species with different functional traits will have a higher functional diversity (FD) than the other community with species having similar functional traits. FD is a robust tool for empirically evaluating and precisely predicting the functional consequences of biodiversity loss. It can provide vital clues on major issues in the macroecology, such as species co-existence, community assembly and saturation.

A method for quantifying FD, based on total branch length, has been developed (Petchey & Gaston, 2002). Pertinently, the FD uses a dendrogram constructed from species trait value, but only those traits linked to the ecosystem function of interest are used. A trait matrix, consisting of 's' species and 't' traits, is constructed and then converted in to a distance matrix. Standard clustering algorithms are used to generate a dendrogram which, in turn, provides the information needed to calculate the branch length. The resulting measure is continuous and can be standardized, so that it falls between 0 and 1.

Phylogenetic diversity

Phylogenetic diversity, often abbreviated as 'PD', represents one of the components of biodiversity. Simply, it is a biodiversity measure based on evolutionary relationships between species, and, therefore, sometimes referred to as ‘evolutionary diversity’. The evolutionary relationships are graphically represented by phylogenetic trees, wherein branches connect the nodes. The length of a branch connecting two nodes can be proportional to the evolutionary divergence between the nodes. In a phylogenetic tree, an internal node represents a hypothetical common ancestor of all the species originating from that node, and the terminal nodes (leaves or tips) represent the observed species or other taxa. A clade is defined as group of species with one single common ancestor (i.e., a monophyletic group).

As the phylogenetically distinct species are likely to have distinct functional traits, phylogenetic diversity is often regarded as a proxy for functional diversity. It is assumed that preserving a high level of phylogenetic diversity (and thus of functional diversity) should be a conservation priority to ensure the maintenance of ecological functioning. Also, detailed data on functional traits is still lacking for most of the species, whereas the increasingly available phylogenies provide sufficient evolutionary information for many taxa. Recent methodological advances now make it easier to access and compile phylogeny, than the more intricate measurement of functional diversity, which requires a large body of trait information. However, selection of the right measure of phylogenetic diversity is a complex task. A large number of measures have been developed to quantify different aspects of phylogenetic diversity, such as the distinctiveness of single species and the whole communities or phylogenetic richness (Winter *et al.* 2012).

Measurement of biodiversity at spatial level

Historically, and for convenience, the species is the main unit chosen to measure the biodiversity at organizational level. Nevertheless, it is equally important to choose a relevant spatial scale, because the extent of species diversity increases with an increase in the area. Generally, in the biodiversity studies, an investigator chooses the study sites from a particular geographical region. In practice, these study sites can be represented by the biotic communities, assemblages, populations, plots or even quadrats, etc. After choosing the study sites of a particular ‘scale’, three main aspects of biodiversity are recognized at the landscape scale, which are:

(i) **Alpha (α) diversity**: it refers to the diversity in species at individual study or sampling sites, or the average of the local measures across all of the sites. In reality, it is the variance in the species identity of the individuals observed at a site. For example, a monoculture has the lowest possible **α**-diversity because there is no variance in the species identity of the individuals.

The **α**-diversity deals with the diversity of spatially defined units. It is measured either by the number of species present at the site (species richness), or by some other index that takes into account the relative frequencies of the species, such as Shannon index of diversity, and measures based on Simpson's indices (Simpson, 1949).

(ii) **Beta (β) diversity**: it refers to the change in species composition from one site to another. In other words, lesser the number of species that the various sites have in common, the higher the **β**-diversity.

Although, in theory, β-diversity can be measured among different communities (Whittaker, 1972) or habitats (Schluter & Ricklefs, 1993); nonetheless, in practice, β-diversity is measured among study plots (localities) of some arbitrary size, because communities and habitats are most often impossible to delineate. Also, the area of individual plots may be as small as a few square meters and as large as hundreds of square kilometers.

Different measures of β-diversity have been proposed, which reflect quite different properties of species composition among communities. Majority of the commonlyused indices of β-diversity are based on Whittaker's βW (βW = γ/**α**), where γ is the number of species in an entire study area and **α** is the number of species per plot within the study area (Whittaker, 1960). The βW and its later variants, such as βT, βH1, βH2 (Wilson & Shmida, 1984; Harrison *et al.* 1992) essentially reflect the inverse average frequency of species and measure the among-plot variability in species composition independently of the position of individual plots on spatial or environmental gradients. However, Velland (2001) suggested that the term β-diversity should be strictly restricted to these measures only, and cautioned that the species turnover should not be used interchangeably with β diversity. Alternatively, species turnover can be measured using matrices of compositional similarity and physical or environmental distances among pairs of study plots. Furthermore, for measuring species turnover from either presence/absence data or quantitative species abundance data, Sorensen's coefficient of similarity (SS) can be used. It is calculated as:

$SS = c / \boldsymbol{\alpha}$

Where 'c' is the number of species shared by both the plots and **α** is average number of species in each plot. The values of SS range from 0 (indicating no species is common) to 1 (indicating all species are common). Also, if SS is subtracted from 1, it can give a coefficient of dissimilarity (Legendre & Legendre, 1998).

(iii) **Gamma (γ) diversity**: it refers to the total diversity measured over the entire suite of sites being considered. It can be estimated directly, or calculated from the **α**- and β-diversity.

The γ diversity is usually measured by pooling the observations from a sample (in the statistical sense), i.e., a large number of sites from the area. It is measured using the same indices as **α**-diversity. Indeed, the γ diversity (diversity of an entire region or landscape) is a function of both the **α**- and β-diversity and, therefore, depends on the levels of both of these. If **α**- and β-diversity are both low, the γ-diversity also will be low; and converse is also true. An intermediate level of γ-diversity could correspond either to high **α**-diversity (each site locally diverse) but low β-diversity (all the sites possess same suite of species) or to low **α**-diversity (each site species poor) but high β-diversity (all sites different from one another).

In essence, these three measures represent a partitioning of diversity across the relative spatial scales. Here the local scale used to define **α**-diversity could be anything from a small habitat patch to an entire reserve. This partitioning is commonly defined in terms of species richness.

PRIMARY PRODUCTIVITY

The primary productivity is a fundamental aspect of ecosystem functioning that determines the amount of energy available to sustain all forms of living organisms on the Earth, including the human beings. In other words, the process determines the energy available for other trophic levels within an ecosystem (Mc Naughton *et al.* 1989). Additionally, primary productivity is a strong regulator of flow of elements within the biosphere.

At the ecosystem scale, the gross primary productivity (GPP) is the sum of the photosynthesis by all the vegetation measured. It is the process by which carbon, and thus energy, enters the ecosystems. Carbon that enters ecosystems as GPP accumulates within the ecosystem, and returns to the atmosphere via respiration. It has been estimated that about half of the GPP is respired by plants to provide the energy that supports their own growth and maintenance (Waring & Running, 1998). Based on this, the net primary productivity (NPP) is, therefore, recognized as the net carbon gain by the vegetation and equals the difference between GPP and plant respiration. In general, the primary productivity studies focus on biomass estimation. Like GPP, the NPP too is measured at the ecosystem scale, usually over relatively long time intervals, such as year (grams biomass per square meter per year).

Primary productivity in the grasslands

The primary productivity – a key variable of terrestrial ecosystems – is an important component of global carbon cycle. As the grasslands occupy 1/5th of the earth's terrestrial land surface, they are undoubtedly the major players in the global carbon cycle (Scurlock & Hall, 1998). Grasslands are, in fact, one of the most widespread terrestrial ecosystems worldwide, covering large expanses of land, both in the tropics and the temperate regions. Although the NPP is an important component of the global carbon cycle, yet progress in developing predictive terrestrial biosphere models has been hampered by the lack of a high quality data set based upon field observations (Cramer *et al*. 1999). The knowledge about NPP of grasslands is still largely based upon the International Biological Programme (IBP) estimates in 1970s; however, at the same time, there are a number of global NPP models which tend to produce higher NPP estimates for grasslands, suggesting a model-data mismatch.

The plant biomass present within an ecosystem at a specific time is the balance between NPP and tissue turnover, and commonly expressed in terms of dry matter (Gower, 2002). Quite often, there is observed a consistent relationship between plant biomass and the climate type that characterize a particular ecosystem. For instance, NPP in the grasslands from tropical and temperate regions ranges from 14.9 and 5.6 $PgCyr^{-1}$, respectively (Saugier *et al*. 2001).

However, within an ecosystem, the disturbance frequency reduces the amount of plant biomass below the levels that the climate could support. Even the patterns of biomass allocation change which, in turn, reflect the factors that most strongly limit plant growth in ecosystems. In grasslands, water or nutrients more severely limit the primary production, and that is why most of the biomass gets transferred towards belowground in the temperate and arctic tundra biomes. For instance, while as 30% of total biomass is transferred to roots in tropical grasslands, it is relatively higher (i.e., 70%) in temperate ones.

Of the various factors, length of the growing season is the major factor explaining biome differences in the NPP. Most ecosystems experience the conditions that are too cold or too dry for significant photosynthesis, or for plant growth, to occur which critically decides the average daily rate of NPP. Where as in the tropical grasslands the length of growing season is 200 days, it is only 150 days in the temperate ones; and accordingly the NPP is 5.4 $gm^{-2}d^{-1}$ in the former and 5 $gm^{-2}d^{-1}$ in the latter, respectively.

In essence, plants retain only a part of the total biomass that they produce. The balance between annual NPP and the initial biomass determines the annual increment in the plant biomass. Some of the annual biomass loss is physiologically regulated by the plant itself; for example, the senescence of leaves and roots. In the grasslands, pertinently, senescence occurs throughout the growing season. Also, the biomass transfers to dead organic matter result from mortality of individual plants. Still other types of biomass losses occur with varying frequency and predictability, and are less directly controlled by the plants themselves, such as the losses to herbivores, pathogens and fire. Nonetheless, plants influence the rate of such losses through the physiological and chemical properties of the tissues it produces.

Measurement of NPP

NPP is defined as the total photosynthetic gain, less respiratory losses, of vegetation per unit ground area (Scurlock *et al*. 1999). For natural vegetation, this is often expressed on an annual basis. For a given period of measurement, NPP is equal to the change in both aboveground and belowground plant biomass, plus any losses over this period due to death and subsequent decomposition, herbivory and exudation/volatilization (Roberts *et al*. 1993). However, many earlier estimates of grassland NPP ignored both turnover and belowground production, and were based on aboveground peak 'standing crop' only (i.e. total clipped live and dead matter, e.g. Kucera *et al*. 1967). Even the coordinated studies of IBP on the Grassland Biome in the 1970s were based mainly on aboveground biomass changes, with relatively few estimates of belowground productivity (Singh & Joshi, 1979). Singh *et al*. (1975) and Long *et al*., (1989) reviewed and discussed the limitations of the various methodologies for estimating NPP of grasslands. They underscored that the grassland NPP estimates are strongly influenced by the methodology used for estimation. Many of the NPP estimates obtained using different computational methods were correlated with one another, and they yielded significantly different figures when applied to the same set of data. The degree of underestimation using different methods may be strongly site-specific (Linthurst & Reimold, 1978); and also the degree of underestimation may vary with the year of measurement.

The existing problems in estimating and simulating grassland productivity worldwide may be partly due to the lack of high quality NPP data estimated from field biomass measurements (both spatially and temporally). An international co-ordination for compilation of global NPP data for model validation and development, the Global Primary Productivity Data Initiative (GPPDI) has been working since 1995. However, the lack of

NPP data still inhibits the progress in estimating and modeling the global carbon cycle (Scurlock *et al*. 1999).

In general, annual NPP may be calculated for a given year (say in temperate region, January-December) or for any appropriate 12-month growing cycle, depending on latitude and environmental or management factors, which determine this cycle. The length of the growing season varies widely, from as little as 3 months in extreme continental or semi-arid conditions, to as much as 12 months in some humid tropical regions. Usually, changes in live biomass (above- and below ground) and dead matter are measured according to a standard methodology.

Current methods used in estimating grassland NPP have their own advantages and limitations (Milner & Hughes, 1968; Singh *et al*. 1975; Long *et al*. 1989; Scurlock *et al*. 2002). Various methods used by researchers for the estimation of NPP are based on a number of assumptions and underlying caveats. Besides respiration, plants loose carbon through other pathways. The largest of these releases is the transfer of carbon from plants to the soil. This occurs through litter-fall (the shedding of plant parts and death of plants), root exudation (the secretion of soluble organic compounds by roots into the soil), and carbon transfers to microbes that are symbiotically associated with roots (e.g. mycorrhizae and nitrogen-fixing bacteria). Plants also release carbon to the atmosphere through emission of volatile organic compounds or by combustion. Volatile emissions typically account for < 1% of NPP. Herbivores also remove carbon from plants, and often accounts for 5-10% of NPP in terrestrial ecosystems, with a high of > 50% in some grasslands and a low of < 1 % in some forests. Finally, carbon can be removed from vegetation by human harvest or other disturbances.

Notwithstanding the other losses, most field measurements of NPP document only the new plant biomass produced and, therefore, probably underestimate the NPP by at least 30% (Long *et al*. 1992). For practical purposes and operational difficulties, such an underestimate may not be significant. A frequent objective of measuring NPP is to estimate the rate of biomass increment. Few components of NPP, such as root production are particularly difficult to measure, and have been often assumed to be of 1:1 ratio to the aboveground production (Fahey *et al*. 1998). Fewer than 10% of the studies that report NPP actually measure components of belowground production (Clark *et al.* 2001). For these reasons, it has been suggested that considerable care must be taken when comparing data on NPP or biomass among different studies. Although some correlations between estimates obtained using different methods have been reported, the degree of underestimation may be strongly site-specific (Linthurst & Reimold, 1978; Long & Mason, 1983). The commonly used method of peak biomass, as an indicator of NPP in grasslands, might apply within certain biomes only; e. g., temperate grasslands.

A comprehensive archive of NPP and biomass dynamics data have been made available through the Oak Ridge National Laboratory (ORNL), and may be especially useful for model and hypothesis testing (Scurlock & Olson, 2002). The data on grasslands include biomass dynamics and associated environmental data, representing a broad range of grassland types, as defined by ecoregions (Bailey, 1989) or by climatic zones.

Scurlock *et al.* (2002) grouped the various methods used for NPP estimation in the grasslands under six methods, which are:

(1) **Peak biomass method**: ANPP = max {AGbiomass}

It is based on the assumptions that any standing dead matter or litter was carried over from the previous year, and death in the current year is negligible; live biomass was not carried over from previous year; below-ground production ignored, or estimated only as a fraction of above-ground production using a crude root/shoot ratio. It may be applicable to annual arable crops, but is clearly a poor estimate of production for perennial vegetation (i.e. most natural plant communities), especially where the below-ground turnover may be significant. Furthermore, it may be useful for crude comparisons between seasonal temperate grasslands, but has little meaning for the tropical grasslands, and should definitely not be used to compare temperate and tropical grasslands.

(2) **Peak biomass method**: ANPP = max {AGTotclip}

It is based on the assumptions that any standing dead matter was formed by death in the current year, hence counted as part of this year's plant production; no standing dead matter has yet fallen as litter or decomposed; neither live biomass nor standing dead matter were carried over from the previous year; below-ground production ignored, or estimated only as a fraction of above-ground production using a crude root/shoot ratio. It may be a slightly better estimate of NPP than the preceding one, where significant death occurs during the growing season.

(3) **Max-Min method**: NPP = max {AGbiomass} - min {AGbiomass

It is based on the same assumptions as the method (1), except that any live biomass carried over from the previous year is excluded. In this method, the subtraction of minimum biomass is likely to be a useful correction only under limited conditions.

(4) **IBP standard method**: NPP = sum {positive increments in AGbiomass}

(Milner and Hughes, 1968)

It is based on the assumptions that most growth occurs between successive sample intervals, i.e. simultaneous growth and death do not occur; NPP is never negative during a sample interval, below-ground production may be similarly measured, ignored altogether, or estimated only as a fraction of above-ground production using a crude root/shoot ratio. This method incorporates several distinct phases of growth within a year, but still fails to account for new shoot growth during periods of high mortality, and vice versa. Nevertheless, the sites where data on biomass dynamics are available (both above and below ground), a more dynamic comparison of net primary production may be possible. Use of this method in comparisons between temperate grasslands displaying marked seasonal changes in biomass and tropical grasslands (where biomass may not change much despite high turnover) should be avoided.

(5) **Modified IBP standard method**: NPP = sum {growth increment}

(Singh *et al*. 1975)

(Where "growth increment" = positive increment in AGbiomass, unless {AGTotdead} increases for that sample interval, in which case: "growth increment" = positive increment in AGbiomass + positive increment in AGTotdead; AGTotdead = Stdead + litter).

It is based on the assumptions that simultaneous growth, death and decomposition (i.e. continuous turnover) does not occur; NPP is never negative during a sample interval; below-ground production may be similarly measured, ignored altogether, or estimated only as a fraction of above-ground production using a crude root/shoot ratio. It is an improved method than the previous one, as the correction for material lost by death during periods of biomass increase will reduce the degree of underestimation of NPP.

(6) **"UNEP Project" method**: NPP = sum {change in AGbiomass + change in AGTotdead + (AGr x AGTotdead)} (Weigert & Evans, 1964)

(Where AGr = above-ground relative rate of decomposition)

It is based on the assumptions that a) measured changes in parameters are statistically significant over each sample interval and in practice, this may be very hard to achieve, since an impractically large number of samples would be required to detect the real but modest changes over each sampling interval; b) decomposition rate is independent of the composition of dead matter and it will decline exponentially as a function of lignin:N ratio; c) losses of AGbiomass and AGtotaldead by grazing, root exudation, etc. are negligible; d) below-ground production may be similarly measured, or estimated only as a fraction of above-ground production using a crude root/shoot ratio. It is the only method which incorporates all components required for an accurate estimate of NPP (where both above and below-ground production measured). Although such detailed data are lacking for majority of the grassland sites, it provides a useful benchmark against which to check the possible degree of underestimation using other methods. Where detailed biomass dynamics are available but no data exists on decomposition or disappearance of dead matter, it may be possible to improve on estimates by modelling decomposition using data from other similar sites.

From the theoretical and practical point of view, those methods which sum up the positive growth increments are the preferred ones to estimate and compare NPP across the widest range of sites without underestimating productivity. Accounting for the dynamics of both dead and live matter appears to be critically important in order to obtain representative estimates of NPP from different ecoregions. As the data on belowground dynamics are still lacking, it means we are neglecting what goes on beneath the soil surface. Sims and Singh (1978) reported that root productivity is most reliably estimated by summing positive increments in total root matter by depth.

Commonly, the peak live biomass is not only reported as a measurement, but also commonly used as an indicator of grassland productivity. However, when more elaborate estimates are applied, both the magnitude and the ranking of grassland productivity between different sites changes significantly. Research studies have shown that peak live biomass method is a fairly good indicator for the general ranking of grassland productivity; however, at the same time, it is clearly an underestimate of the magnitude of NPP. It has been clarified that the peak live biomass should not be used for comparison, for instance between temperate and tropical grasslands. The use of peak biomass as an indicator of NPP may apply only within a certain range of a biome, say temperate grasslands. Also, it may be used as a reasonable benchmark indicator of the magnitude of productivity for study sites within a particular sub-biome.

BIODIVERSITY AND ECOSYSTEM FUNCTIONING (BEF)

The effects of biodiversity on ecosystem functioning has long been one of the hotly debated topics and has consequently seen a flurry of research activities (Hooper *et al*. 2005). In the recent past too, the relationship between biodiversity and ecosystem functioning (hereafter BEF) has emerged as a dominant issue in the biodiversity research (Chapin *et al.* 1997; Naeem *et al*. 1999; Loreau *et al*. 2001; Sutherland *et al*. 2013). The present interest in the BEF research arises from the heightened concerns of species extinction and resulting loss of biodiversity. A large body of observational and experimental data has been accumulated (Kinzig *et al*. 2002; Loreau *et al*. 2002) and, in most cases, the data support the idea that diversity supports ecosystem functioning. Apparent links have been established between the BEF research findings and the contemporary conservation concerns (Srivastava, 2002). It is expected that the research insights gained from this field will allow biologists to make expert predictions regarding the consequences of species extinction on the ecosystem goods and services, on which ultimately depends the human existence. Most importantly, the BEF research findings have gradually influenced the global environmental policy-making, largely justifying biodiversity conservation in order to maintain ecosystem functioning.

The BEF hypothesis is based on the assumption that a decline in biodiversity will alter ecosystem-level processes. Although the hypothesis has deep academic and philosophical roots (Hector *et al*. 2001; Naeem, 2002), yet it was widely discussed by scientists in the early 1990s, owing primarily to pioneering research initiative (Lubchenco *et al*. 1991), seminal conference (Schulze & Mooney, 1993), and international collaborations (Heywood & Watson, 1995). It represents a synthetic field of biodiversity research that seeks to understand how changes in species composition, distribution and abundance change the ecosystem functioning (Schulze & Mooney, 1993; Kinzig *et al*. 2002; Loreau *et al*. 2002; Naeem *et al*. 2002). As changes in the biodiversity are worryingly rampant from global, regional to local scales, research findings from this field have received considerable attention (Naeem *et al.* 1994; Hooper & Vitousek, 1997; Naeem & Li, 1997; Tilman *et al*. 1997; Wardle *et al.* 1997; Emmerson *et al.* 2001; Pfisterer & Schmid, 2002). Although these findings are too complex and difficult to be interpreted easily (Kaiser, 2000; Naeem, 2002), synthesis and consensus are emerging, and the central challenges are being identified (Hughes & Petchey 2001; Loreau & Hector, 2001) for the BEF research to evolve from a descriptive into a predictive science. At present, the question is no longer whether biodiversity matters, but how it matters (Rosenfeld, 2002).

In the BEF relationship, the term ‘biodiversity’ encompasses a broad spectrum of biotic scales from genetic variation within species to biome distribution on the planet Earth (Wilson, 1992; Purvis & Hector, 2000). It can be described in terms of numbers of entities (how many genotypes, species, or ecosystems), the evenness of their distribution, the differences in their functional traits, and their interactions. Traditionally, biodiversity has often been used as a synonym for species richness (the number of species present); even though the different components of biodiversity (e.g., richness, relative abundance, composition) can have different effects on ecosystem functioning. Most of the BEF studies have focused on the changes in richness and composition at the species levels, without taking into consideration the species functional traits; the latter have unambiguously strong influence on the ecosystem processes. Hence, the number of species alone may not be the best predictor of ecosystem functioning; for instance, even a relatively rare species (e.g., keystone species) can strongly influence ecosystem functioning. Therefore, for a precise prediction of the ecosystem-level importance of a species, besides the species richness and relative abundance, the relationship between species or taxonomic richness and functional diversity in natural and experimental ecosystems is being vigorously explored (Diaz & Cabido, 2001; Hooper *et al*., 2002; Schmid, 2002).

Whilst the word ‘ecosystem functioning’ is a broad term, it is usually used to describe the ecological processes that control the fluxes of energy, nutrients and organic matter through an ecosystem. The most important ecosystem functions include primary productivity, nutrient cycling, and decomposition. Ecosystem functioning also encompasses a variety of phenomena, including ecosystem properties, goods, and services. Ecosystem properties include both sizes of compartments (e.g., pools of materials such as organic matter), and rates of processes (e.g., flow of energy and materials among compartments). Ecosystem goods are those ecosystem properties that have direct market value; e.g., food, medicines, tourism and recreation. Ecosystem services are those properties of ecosystem that, either directly or indirectly, benefit human endeavours, such as maintaining hydrologic cycles, regulating climate, cleansing air and water, maintaining atmospheric composition, pollination, soil genesis, storage and cycling of nutrients (Daily,1997).

Given such broadness of concepts involved in the BEF research, it is imperative to be specific about which facet of biodiversity is affecting which process of ecosystem functioning. Much of the contemporary biodiversity research (Groombridge, 1992; Hawksworth, 1995; Groombridge & Jenkins, 2000) has focused on the species richness (the number of taxonomic species) and evenness (relative abundance) of communities and ecosystems (Colwell & Coddington, 1994; May, 1995, Magurran, 2004) and spatial and temporal patterns in the distribution of species within a habitat (MacArthur 1972;

Rosenzweig, 1995; Gaston, 2000). The measures of ecosystem functioning, such as process rates and pool sizes, include both levels (e.g., average rates or sizes) and variation (amount of fluctuation). Pertinently, the variation in the ecosystem functions can result from fluctuations in the environment from year to year, biotic and abiotic disturbances.

In the BEF research, biodiversity is most often measured as the number of species (i.e., species richness), although a few studies have incorporated species evenness also (Wilsey & Polley, 2004); while as the primary productivity is the most commonly used measure of ecosystem functioning. Fundamentally, ecosystems possess the capacity to generate and to circulate material, such as carbohydrates or nutrients. Therefore, it is important to have a better understanding of how these materials are partitioned in the system and which components limit their rate and partitioning. In this context, mechanisms that are the basis for species organization in the biotic communities and which regulate processes involved in the ecosystem functioning of biodiversity need to be properly understood.

Biodiversity, at levels ranging from genetic diversity to species - and ecosystem diversity, is critical for the maintenance of both natural and artificial ecosystems. Whilst the importance of biodiversity for ecosystem functioning is almost unequivocal, it is still less clear how much biodiversity is required to provide for those functions. Lot of uncertainty exists about the critical thresholds of diversity, and the conditions or time scales over which diversity is particularly important. In theory, it is hypothesized that there may be 3 principal ways in which ecosystem processes might respond to reductions in the species richness (Lawton, 1994; Johnson *et al*. 1996).

a) **Redundancy**: a minimum number of species is necessary to carry out basic ecosystem processes, beyond which most of the species are equivalent and their loss has little significance.
b) **Rivet-popping**: viewing the species in an ecosystem just analogous to the ‘rivets’ holding together a “well-made airplane”, the loss of a few species may have no apparent effect, but beyond some threshold losses, the ecosystem processes will fail (Ehrlich & Ehrlich, 1981).

c) **Idiosyncracy**: with change in diversity, ecosystem functions also change; but the magnitude and direction of change is unpredictable because individual species have complex and varied roles (Lawton, 1994).

However, recent experimental evidence does not support these hypotheses. Most workers prefer a model with a threshold in species richness, below which ecosystem function declines steadily, and above which changes in species richness are not reflected in changes in the ecosystem functioning (Vitousek & Hooper, 1993). The model suggests that all the species are equally important, and it is traits of the species added or deleted which matters, rather than the number of species *per se*. There is also a significant asymmetry in the contribution of individual species to the ecosystem functioning (Sala *et al*. 1981). For the ecosystem processes (e.g., primary productivity), there is a good relationship between the abundance of a species and its contribution to ecosystem functioning.

Broadly, the following seven potential mechanisms have been proposed to explain the relationship between biodiversity and ecosystem functioning:

1) **Niche complementarity**: niche differentiation between species allows diverse communities to be more efficient at exploiting resources than depauperate ones, leading to greater productivity and retention of nutrients within the ecosystem. Simply, if species differ in their resource use, then the more species-diverse community will exploit the available set of resources much efficiently, with the traits of different species complementing one another (Tilman *et al*. 1997; Loreau, 2000). Such a complementarity effect results from the reduced inter-specific competition through niche-partitioning. If species use different resources or same resource in space and time, more efficient use of the available resources pool is expected by the species-diverse community. Complementarity effects are usually the greatest when species differ greatly in functional traits, either in timing (Chesson, 2000), in spatial distribution (Schenk & Jackson, 2002) or in type of resource demand (McKane *et al*. 2002).
2) **Functional facilitation**: a salutary effect of one species on the functional capability of another will lead to an increase in ecological function in more diverse

communities. Thus, simply increasing the number of species in a local assemblage could augment the number of mutual, facilitative or positive indirect interactions, thereby increasing ecosystem functioning (Mulder *et al.* 2002; Cardinale *et al*. 2002; Bruno *et al*. 2003). Best evidence of facilitation among plants can be seen when at least one species has the ability to form a symbiotic association with nitrogen-forming bacteria.

3) **Sampling effect**: it is also sometimes referred to as selection probability effect (Huston, 1997) or positive selection effect (Loreau, 2000). The sampling effect essentially combines probability theory with species-sorting mechanisms. When there is positive covariance between the competitive ability of a species and its per capita effect on ecosystem functioning, the probability of including a dominant, functionally important species will increase with diversity. In other words, if in a regional pool of a large number of species some have strong impacts on ecosystem processes, then more the species that are drawn from this pool to form a local assemblage, the greater the probability that some of these strongly impacting species will be included (Aarssen, 1997; Huston, 1997; Tilman *et al.* 1997; Holt & Loreau, 2002).
4) **Dilution effect**: lower densities of each species in high diversity communities may reduce the per capita effects of specialized enemies, such as pathogens (via reduced transmission efficiency), or predators (via reduced searching efficiency). In effect, specialized enemies create frequency-dependent selection among species (Mitchell *et al*. 2002).
5) **Insurance effect**: species that are functionally redundant in a particular situation may become functionally important during a changing environment and/or stress, thus allowing maintenance of the ecological function even after perturbation (Yachi & Loreau, 1999).
6) **Portfolio effect:** independent fluctuations of many individual species may show lower variability in aggregate than fluctuations of any one species (Doak *et al*. 1998; Tilman *et al*. 1998). It is analogous to a diversified stock portfolio, which represents a more conservative investment strategy than would any single stock. This effect does not require any interactions between species.

7) **Compensatory dynamic effects**: negative temporal covariance between species abundances creates lower variance in their aggregate properties, such as total biomass (Tilman *et al*. 1998).

In practice, quite often, all these mechanisms may be operating in tandem, and hence the research challenge is to find ways to determine their relative contribution to ecosystem functioning.

Alteration of biota in ecosystems occurs mainly through species extinctions and invasions driven by human activities that ultimately alter ecosystem goods and services. Many of these alterations are difficult, expensive, or impossible to reverse or fix with technological solutions. Some of the ecosystems are initially insensitive to species loss because: a) multiple species carry out similar functional roles; b) some species contribute little to ecosystem functions; and c) ecosystem functions are primarily controlled by abiotic factors. Nonetheless, it becomes clear that more species are needed to ensure a stable and sustainable supply of ecosystem goods and services, as large spatial areas and longer time periods are considered. Pertinently, sustainability simply refers to the capacity for a given ecosystem service to persist at a given level for a long period of time (Lubchenco *et al*. 1991). Although the higher species richness increases both species redundancy and temporal resilience of ecosystem functioning (Naeem, 1998), yet understanding the BEF relationship is not so simple, and becomes further complicated by the temporal dynamics of ecosystems. In other words, as conditions change, different species become more or less important contributors to ecosystem functioning. Because of this reason, biodiversity is crucial not only in terms of how many species are present at a given time, but in the maintenance of a pool of species that can buffer a system against the uncertainty (i.e., an insurance effect) (McGrady-Steed *et al*. 1997).

Naeem & Wright (2003) underscored that the response of an ecosystem to changes in biodiversity is determined largely by four factors: a) the species composition of the biotic community b) the abundance of each species; c) the functional traits each species possesses; and d) the biotic interactions among species that regulate magnitude and variability of expression of the function (e.g. primary productivity) under investigation. To transform BEF research into a predictive science, they further suggested for an appropriate evaluation of the above four factors, firstly by determining species composition across sites, secondly by determining abundance (commonness and rarity of species), and thirdly by determining the ecosystem functioning (estimate ecosystem functioning based on changes in the biodiversity).

Subsequently, Leps (2004) underlined the realism and generality as the two essential requirements for the BEF research studies. By realism, he meant the existence of a natural community to which the results obtained from an experimental community can be applied; while the generality increases the number of such natural communities in the real world. He suggested that the relationship between biodiversity and ecosystem functioning can be studied by a set of natural communities of different diversity, and then correlating their diversity (e.g. species richness) with some measure of their functioning (e.g. primary productivity).

BEF relationship in grasslands

Much of the research work on the effects of biodiversity on ecosystem functioning has focused on the grasslands, because they are the easy ecosystems to manipulate and their aboveground net primary productivity is relatively easy to approximate, as most of the aboveground biomass is generally accrued during a single year. However, in the latter case, it is important to note that the measurements may underestimate the primary productivity if they do not take in to account intra-annual turn over (Scurlock *et al*. 2002). For the BEF research, the species identity (i.e. differences in species composition by knowing which species are present) is also as important as species richness (i.e. knowing how many species are present).

Biodiversity plays a crucial role in ecosystem productivity and stability in the grasslands. Research studies have clearly demonstrated a definite relationship between plant species richness and primary productivity in grasslands; and thus the conservation of biodiversity is essential to maintain productivity under variable adverse environmental conditions (Tilman and Downing, 1994). Archer *et al*. (1996) argued that the functional complementarity and interchange among spatial components of the ecosystem enhance the systems' stability and resilience. Empirical analysis of ecosystem diversity, including species richness, abundance, and functional groups diversity, improves understanding of system stability and resilience in the face of disturbance (e.g. overgrazing, drought or their combination). Such a knowledge base can guide the sustainable management strategies to mitigate the adverse human impacts; the latter pose serious threats to the ecosystem resources and may cause an irreversible damage (Archer & Stokes, 2000). Further, such an understanding will help in restoration of already degraded ecosystems.

The two great challenges facing the global scientific community are the conservation of biodiversity and the continuation of the life-supporting ecosystem goods and services. In the present era of global environmental change, the conservation of biodiversity has immediate implications for sustaining ecosystem functions in the frequently fluctuating environments (Tilman *et al*. 1996). Studies demonstrate that the loss of biodiversity can influence the key ecosystem processes, such as primary productivity, nutrient cycling and decomposition process (Chapin *et al*. 1998). Globally, as the loss of biodiversity has reportedly accelerated over the recent decades, efforts to determine the relationship between biodiversity (species richness) and ecosystems processes (primary productivity) have intensified. This is a rapidly expanding area of study. As of now, both theoretical and empirical approaches strongly suggest that the relationship between species richness (SR) and primary productivity is mostly, but not always, uni-modal (hump-shaped). Theoretical approaches to understand this relationship postulate either: positive, linear relationship (MacArthur, 1972); positive, non-linear relationship (Walker, 1992); and no relationship between species richness and productivity (Lawton & Brown, 1994).

The existence of a hump-shaped (uni-modal) relationship between plant species richness and biomass (or primary productivity) has been intensely debated (Grime, 1979). Although a uni-modal relationship has been observed to fit many empirical data sets, much discussion continues about the generality of the relation and the theoretical underpinning (Aarssen, 1997; Grace, 1999). While some workers believe that the different taxonomic groups may attain peak diversity at different levels of productivity (Rosenzweig & Abramsky, 1993); others argue that the taxonomic restriction may be responsible for the unimodal nature of the relationship (Abrams, 1995). Often standing crop is used as a substitute for productivity (Tilman *et al*. 1996); the terms are frequently used interchangeably. Grace (1999) suggested that the species richness is more correlated strongly with biomass, rather than with productivity. He suggested that both the peak standing biomass and the productivity should be measured, after which their relationship with species diversity be compared individually.

The relationship between species richness and productivity has been shown to be scale-dependent, with the form of the scale-dependence being variable across different studies. While most of the studies reported no pattern at all, still unimodal relationships are as common as positive relationships at the within-community scale,. However, at the across-community scale, unimodal patterns are more than three times prevalent than positive relationships. At the continental- to global-scale, the pattern is dominated by positive relationships. Generally, the species richness is a positive function of productivity at the larger scale.

Tilman *et al*. (1997) showed that the factors that change ecosystem composition, such as species extinction and anthropogenic disturbances, strongly affect ecosystem processes. Further, different ecosystem processes are likely to be affected by different species and functional groups. Hooper and Vitousek (1997) confirmed that the functional characteristics of the component species are important for maintaining vital ecosystem processes and services. Although the ecosystem functioning is determined by the taxonomic composition of its biota, but equally important are the functional traits of individual species, their abundance and distribution, and their ecological interactions. Thus, functional diversity (FD) is as relevant as the species richness and evenness. Identifying the key traits that influence particular ecosystem functions, is an important step in understanding how biodiversity affects ecosystem functioning. Individuals or species that possess a common set of functional traits selected by the investigator, are referred to as functional types; or putting differently, functional types are the groups of species that are ecologically similar in their effects on ecosystem processes (Chapin 1993). In other words, more the number of species belonging to a single functional type, higher the functional group diversity within a natural community. As species are the main repositories of functional traits, use of the estimates of species richness (i.e., taxonomic diversity) can serve to some extent as a surrogate for measuring the functional diversity.

There is no universal classification of functional groups, because the traits that are decisive in predicting effects on ecosystem processes differ strongly among ecosystems, as well as within ecosystems for different processes. Therefore, characterization of functional groups will be quite different, depending upon the specific aim of the study. For instance, if a modeling of land use and grazing is the main aim, then the trait of palatability of plant species may result in different functional groups. Although the optimum ecosystem functioning can be maintained with reduced numbers of species in most ecosystems, yet species diversity may be important for the survival of communities in the fluctuating environments. Thus, species should not be regarded as redundant at all when considered in a long time-frame. In actuality, the communities are non-random assemblages of species, and tightly-interwoven functional links between species in a community are common. Such tight linkages between species demonstrate the difficulty of lumping species into functional groups, because the individual species within such a group may not be independent but tightly fixed to the existence of a species in quite another functional group.

The exact traits responsible for governing a given ecosystem function will vary, depending on the numerous factors, including temperature, soil or water conditions, precipitation, nutrient availability, and time since disturbance. In many cases, traits important in determining a given ecosystem function may be shared among multiple species within a community. Notwithstanding such a redundancy of trait distribution, there have been attempts to classify species with similar traits into functional groups. Recent studies on functional groups also provide considerable exposure towards many issues surrounding the identification of functional groups, especially for plants (Smith *et al* 1997; Diaz & Cabido 2001; Lavorel & Garnier, 2002). Traditionally, the grouping of plants according to life-forms has a long history. Such a plant functional group classification relevant to the grasslands is that based on eco-physiological traits, which segregates plant species into 8 functional groups; these are:

(i) C4 grasses
(ii) C3 grasses & sedges
(iii) annuals and biennial forbs
(iv) ephemeral spring forbs
(v) spring forbs
(vi) summer forbs
(vii) autumn forbs
(viii) legumes
(ix) woody shrubs.

In general, the grassland communities with a higher diversity of species and functional groups are more productive, and utilize resources more efficiently by intercepting more light, taking up more nitrogen, and occupying more of the available space. One major mechanism that could allow production to increase with diversity in grasslands is the complementary use of resources, such as through light partitioning, in which differences in morphology between species enhance the structural complexity of the vegetation, and allow for more complete utilization of PAR (photosynthetically active radiation). Diversity is also likely to increase complementarity in nutrient uptake, either by different grassland species acquiring nutrients from different portions of the available pool in space, time, or chemical form, thereby increasing, e.g. total nitrogen retention (Spehn *et al*. 2002), or decreasing losses due to leaching (Scherer-Lorenzen *et al*. 2003). The number, relative abundances, identity and interactions of species, all affect ecosystem processes. As no single species can perform all the functional roles that are exhibited at a particular trophic level, a diverse group of species is functionally essential, because it increases the range of organismic traits that are represented in an ecosystem and, therefore, the range of conditions under which ecosystem properties can be sustained.

The relationship between herbaceous biomass and species richness in the grasslands has often been hump-shaped with a peak in species richness at a low to intermediate level of biomass (Grime, 1997). Grime (1979) concluded that the maximum species richness can be expected between 350 and 500 gm^{-2} of biomass. However, Bhattarai *et al*. recently (2003) observed maximum species richness at relatively lower biomass (i.e. 120 gm^{-2}), which is ascribed to differences in the overall productivity. At higher levels of biomass, the decline in species richness is believed to be due to competitive exclusion (Gough *et al*. 1994).

Srivastava (2002) found about 71% of BEF studies showed a positive effect of diversity on ecosystem functioning. These positive effects were more often log-linear, implying that many species can be lost from an ecosystem before there is much decrease in the ecosystem function (Schwartz *et al*. 2000; Wardle, 2002). Mittelbach *et al*. (2001) reported that at smaller spatial scales, hump-shaped relationships are dominant for within-community type, as well as for across-community type, but percentage of the occurrence of hump-shaped curves is higher at the across-community scale. Hump-backed pattern is a result of the floristic heterogeneity, and will only be detected when sampling covers the total gradient of different microhabitats. Lawler *et al.* (2002) believed that on increasing spatial or temporal scale, the relationship will change from log-linear to linear. Other workers have underscored that the effect of species diversity depends on the ecosystem studied, and the function measured (Schmid, 2002; Duffy, 2003).

In the grasslands, the diversity effects are more pronounced on the above- than on the below-ground processes (Spehn *et al*. 2005). Highly diverse grassland communities commonly show a consistent pattern of higher complementarity effect values, greater resource use in terms of light and space, higher aboveground productivity and, therefore, larger pools of nitrogen in the plant biomass. This linkage of greater complementarity, resource use, and productivity is consistent with theories based on niche differentiation and resource partitioning (Tilman *et al*. 1997; Loreau, 1998; 2000); but is partly also based on facilitation (Callaway, 1995; Mulder *et al*. 2002; Bruno *et al.* 2003).

EFFECTS OF GRAZING ON GRASSLANDS

Almost all energy on the planet Earth is fixed via photosynthesis, and it is thus an evolutionary mandate that the higher trophic level organisms consume plants and the successful plant species will be those which can tolerate or escape herbivore grazing.

Owing to this continuous co-evolutionary struggle, to grow and consume plants, the plant-grazer interaction is one of the most actively investigated areas of research. Research findings from this area of study provide vital insights for evaluating the effects of grazing on the plant diversity and ecosystem functioning, which in turn has implications for the provision of life-sustaining ecosystem goods and services.

Grazing, in traditional sense of the term, refers to the consumption of plant parts by animals (i.e. herbivores) without causing the death of an individual plant. The most common example is the consumption of plant species as fodder by domesticated livestock in the world's grasslands. Livestock (or in other words domesticated animals) grazing is one of the most ubiquitous human activities on the globe, and occurs more commonly than any other land use. It is estimated that the livestock graze about 1/3rd to 1/2nd of our planet's total land area. It has played a prominent and largely beneficial role in the human society for thousands of years – rendering food, fuel, fertilizer, and clothing. In spite of these benefits, during the recent past, the negative impacts of livestock grazing on global biodiversity have become a serious environmental concern. In many regions of the world, excessive grazing in the grassland ecosystems has decreased the density and biomass of many plant and animal species. World over, this has led to reduction of biodiversity, alteration of nutrient cycles, distribution and spread of alien invasive species, acceleration of soil erosion, and diminishing productivity and land use options for the future generations.

In retrospect, the natural grasslands occurred extensively around the globe in areas with fairly long dry season. Although the seasonal drought and the fire are typically important in determining the diversity and distribution of grasslands, but the herbivore grazing – in removing biomass – often plays an overriding role (Peet *et al.* 1999). Grazing is one of the most omnipresent forms of disturbance within the grasslands, and hence plays diverse roles. The role of grazers is so striking that now vast expanses of semi-natural successional grasslands occur extensively in many areas of the world where deforestation has taken place. In addition, many of the natural grasslands have been degraded and extensively modified by the domesticated livestock. Grazing also plays a major role in determining plant diversity by strongly affecting species composition, abundance and community organization within the grasslands. It has been generally observed that the intermediate levels of disturbance promotes the maintenance of higher biodiversity within a given plant community (Svensson *et al*. 2012). As plants constitute the major structural and functional component around which other biotic community members are organized, the effects of grazing on plant diversity, therefore, have strong indirect effects on other trophic levels, and on the biodiversity, of the ecosystem as a whole.

On a worldwide basis, response of grasslands to the domestic livestock grazing has been variable (Milchunas & Lauenroth, 1993). In some areas, where the native vegetation is evolutionarily well-adapted, changes in grassland biodiversity have been appreciably non-significant (Milchunas *et al*. 1988). In other areas, especially in tropical and subtropical grasslands, changes have been dramatically drastic, and a shift from a grass-dominated vegetation to a one dominated by forbs or shrubs has been noticeably observed (Walker *et al*. 1999). Besides, the conversion of natural grasslands to cropland or seeded pastures has major impacts on their biodiversity and ecosystem functioning. All such major transformations of grassland ecosystems, and resultant effects on their biodiversity, modify the water, carbon and nutrient cycles to an extent that largely jeopardizes the provision of ecosystem goods and services in grasslands.

The manner in which the grazing affects the diversity and productivity of grasslands is a challenging question, because it encompasses the full complexity of ecosystem dynamics. As Olff & Ritchie (1998) rightly suggest that the question for researchers is no longer “do herbivores have an effect?but why do effects differ?” Although herbivore grazing is considered to be an important factor for plant species diversity, it is difficult to generalize about the impact (Crawley, 1997). Broadly, the effects of grazing on species diversity in grasslands depends on type of grazers, grazing intensity and frequency (Hobbs & Huenneke, 1992), as well as on habitat characteristics, such as productivity (Proulx & Mazumder, 1998), climate and evolutionary history (Milchunas *et al.* 1998), and the regional species pool (Zobel, 1998).

As compared with insects and small mammals, the large grazers, such as livestock, have relatively consistent effects; with diversity increasing at moderate levels of grazing and decreasing under heavy non-selective defoliation – a pattern hypothesized by Milchunas *et al*. (1988). They argued that diversity in the sub-humid grasslands (625-1000 mm precipitation) increases sharply from no grazing to very light grazing, but declines progressively as grazing moves from very light to intense. It has also been predicted that the semi-arid grasslands would show little or no diversity increase under intermediate grazing by large generalist herbivores. Subsequently, Milchunas & Lauenroth (1993) broadly supported the idea that floristic change in response to grazing is the least in low productivity grasslands. It was postulated that at small (local) scales (< 100 m^2) density is increased by grazing due to reduced plant competition and enhanced regeneration. Over larger areas, the grazing inhibits diversity due to the pool of potential species being restricted to grazing-tolerant plants.

From these studies, the importance of spatial and temporal scales for understanding effects of grazing on the patterns of biodiversity has gained an increasing attention (Huston, 1999; Austin, 1999). Thus, the variation in grazer effects on plant diversity in grasslands arises mainly from the spatial or temporal scales at which diversity is measured or affected. Grazers can influence the plant species richness at both the local scale (plant-neighbourhood) and at the regional scale (spatial range of an individual or population of grazers). Local disturbances and selective grazing can enhance diversity at the local scales, but strong selection for grazing-tolerant plant species within the species pool might reduce diversity at the larger scales (Gibson & Brown, 1999). Similarly, the temporal scale is also important for the grazer effects, because short-term increases in the plant diversity from grazing can ultimately disappear owing to grazing-induced succession to a few grazing-tolerant plant species (Davidson, 1993; Anderson & Briske, 1995). As against this, periodic outbreaks of grazers at intervals could maintain high plant diversity in the grasslands (Root, 1996).

Some research studies have provided a few mechanistic explanations for understanding the scale-dependent diverse effects of grazing on the plant diversity in grasslands. It has been suggested that the local plant species richness in grasslands is maintained by a dynamic interaction between local colonization (via dispersal and establishment) from species pools at larger spatial scales, and local extinction processes.

Both the colonization and extinction-related processes are mutually inclusive. Therefore, high plant diversity in grasslands is predicted when local extinction rates of species are lower than the local colonization rates. As reduction of dominant competitors by herbivore grazing can enhance the persistence of those plant species that colonize the disturbed areas, the ultimate effects of grazers on plant diversity might depend on their relative impact on the biomass and reproduction of dominant plant species, the density and type of regeneration sites, and the supply of propagules from the rare plants.

Huston (1994) argued that the effects of grazing on diversity should reverse between productive and unproductive environments; this was subsequently supported by Proulx & Mazumder (1998) in a review comprising different habitats. Frequent livestock grazing leads to decline in the richness and abundance of perennial plant species, of which the native plant species are of particular concern. Long-term grazing may influence plant communities by acting as selective filter, eliminating intolerant species, and also through the evolution of persistent species. Grazing history may, therefore, influence the ability of a community to respond to environmental changes, including changes in the grazing intensity (Mack & Thompson, 1982).

Although variation in the effects among different grazer and environment types remain largely intractable (Davidson, 1993); still research insights are emerging into the relationship between grazer body size, variation in their digestive capability, spatial scale of effect, and vulnerability to predators. Large grazers, such as domestic livestock at high density (intensive grazing), can graze unselectively and/or create widespread erosive, detrimental soil disturbances, leaving only a few tolerant plant species, thus reducing the overall plant diversity in grasslands.

Results from the grazer exclosure experiments in different types of grasslands suggest that the effects of grazers vary predictably across environmental gradients. The characteristics of grazer and plant species composition found in different environments could determine how the grazing affects plant diversity in the grasslands, as summarized below:

(i) In the grasslands with infertile soil but limiting precipitation (e.g., arid desert grasslands), dominant plant species can either be ephemeral or have water retention mechanisms (via thorns, hairs, wax layers and secondary chemicals) that also deter herbivory. Such type of grasslands can, therefore, support a few small grazers. The latter would selectively graze rare, palatable plant species and, thus, increase the local extinction rates. Also, low natural abundance of grazers implies that few plant species in the species pool have evolved a tolerance to grazing. High stocking rates of large grazers, such as livestock, are, therefore, likely to reduce the plant diversity dramatically in such grasslands.

(ii) In the grasslands with fertile soils but limiting precipitation (e.g., savannas), dominant plant species are palatable and support high densities of grazers. These plant species tolerate rather than avoid grazing because of high re-growth opportunity owing to the fertile soils. Therefore, complete exclusion of grazers could have negative effects on plant diversity, because only a few grazing-intolerant plant species remain in the species pool to colonize un-grazed areas.

(iii) In the grasslands with infertile soil but non-limiting precipitation (e.g., unfertilized meadows), dominant plant species have low tissue nutrient concentrations but are sufficiently productive to induce competition for light (Berendse, 1985). These plant species will be consumed only by the large grazers because they can tolerate low plant tissue quality (Belovsky, 1986). Grazing will shift competition from light to soil nutrients, thus allowing more plant species to coexist. Therefore, herbivores, especially, the large grazers could dramatically increase diversity in such type of grasslands.

(iv) In the grasslands with fertile soils and non-limiting precipitation (e.g., flood-plain grasslands), dominant plant species are productive and competition for light assumes significance in the absence of grazing. Strong light competitors, such as tall grasses and woody plants, which dominate these grasslands in the absence of grazers are likely to be unpalatable (because of high stem-leaf ratio) to all but the large herbivores when mature. Grazing by large herbivores opens the canopy, so that only a few grazing-tolerant plant species replace the tall species. These tolerant species which are able to re-grow quickly after being grazed, support high densities of grazers but reduce the plant diversity.

Besides, several conceptual models have also been proposed to predict impacts of grazing on the structural and functional properties in the grasslands. Based on a world-wide data-set analysis, Milchunas & Lauenroth (1993) proposed that alteration in grassland structure and diversity due to grazing was primarily a function of the environmental moisture (or its correlate aboveground net primary productivity), the evolutionary history of grazing, and the recent level of consumption by the herbivores in each particular grassland ecosystem. These factors determine species composition and the prevailing growth forms, and morphological traits that characterize the plant community in the grasslands which, in turn, determines the response of individual species to grazing. They predicted negative effects of grazing for sites with long evolutionary history of grazing, higher aboveground net primary productivity (ANP), and many years of protection from herbivores. Not surprisingly, differences in the ANP between differentially grazed sites may be closely related to the number of years of protection from grazing. The grazing response has largely been significantly related with plant species growth form, rather than to plant species composition, which may differ across sites. The difference in plant species composition and growth forms between lightly-grazed and heavily-grazed sites reflects generally a shift from tussock grass species, and tall stature and rosette growth forms in the former, to unpalatable, grazing avoiding- or tolerant-species, short graminoids, creeping forbs and prostrate growth forms in the latter.

Grazing by large herbivores is a major structural force in the grassland ecosystems through its impact on the primary productivity which, in turn, regulates energy flow through the ecosystems. It has been shown that the characteristics of ecosystems subjected to the grazing, such as the primary productivity, plant life forms and resulting vegetation physiognomy, can interact with grazing in determining plant community structure and diversity. Amongst these characteristics, the primary productivity is particularly important, as it determines both the standing biomass and extent of grazing, while at the same time it modulates plant interactions and is linked to the community structure (Waide *et al*. 1999). Furthermore, across plant communities, primary productivity is a comparable indicator of spatial or temporal variation in resources availability.

The varied and too complex plant responses to grazing in the grasslands are difficult to predict. Often, our knowledge of the impacts of grazing on plant communities comes from exclosures. However, to understand plant responses to grazing, it is crucial to explore the impacts of variation in grazing on the community structure and function. At present, studies mostly deal with the impacts of grazing on the community dynamics of a relatively small number of individual species. A considerable interest has been generated in the response of functional groups of plants to grazing (Sternberg *et al*. 2000).

Despite the importance of primary productivity (i.e., both aboveground and belowground), little information is available on the effects of herbivores on belowground net primary productivity (BNP). Some field studies conducted so far indicate that grazing can produce no effect (Milchunas & Lauenroth, 1993), can decrease (Biondini et al., 1998) or can increase root biomass or BNP (van der Moarel & Titlyanova, 1989). Clearly, a general pattern of response of ANP and BNP to grazing in the grassland ecosystems is yet to be ascertained.

In general, herbivores often increase organic breakdown and the mineralization of potentially limiting nutrients such as N, P and K. For instance, the aboveground productivity of moderately grazed plots in the famous Serengeti increased to about 2-fold greater than that of ungrazed plots. In addition, un-grazed grasslands were senescent, whereas those grazed by large herbivores produce younger and more palatable shoots. Overgrazing is often associated with a decrease in the concentrations of N, P, and K in the herbage.

A principal cause of deterioration of grasslands is the selective grazing, due to which the most preferred species are always hit hard. The degenerate stage consists of inferior, less-palatable species and a decline in the standing crop of aboveground vegetation. The degradation of grasslands follows a general pattern, where grazing alters the species composition from long-lived perennials to short-lived perennials and annuals (O'Connor *et al*. 2001). Because of a reduced vegetal cover, which stimulate the surface run-off, and soil and nutrient loss, the overall productivity declines with increasing degradation, a pattern that is widely found in the grasslands (Du Preez & Synman, 1993).

Grazers may indirectly affect biodiversity via trampling, tunneling, seed dispersal, nutrient regeneration and selection for plant traits that have cascading effects on other ecosystem processes. Trampling by hooves of livestock damages the soil sub-system, and infiltration rate also gets decreased. The cumulative outcome of these impacts is massive surface run-off and soil loss from heavily grazed sites (Ambasht, 1985).

Grazing has multiple effects beyond the mere reduction of plant biomass. The reduction of transpiring surface by defoliation conserves the soil moisture, and the extent to which the vegetation is stimulated by grazing depends on the soil water status. Defoliation has a profound effect upon plant physiological processes, but the effects of grazing on primary productivity can’t be sought solely in eco-physiology. A full understanding of vegetation responses to herbivory must encompass processes from the individual to the ecosystem level. Also, the release or recycling of nutrients due to grazing may be an important component of stimulation of productivity by grazing. Other grazing effects leading to compensatory growth occur at the level of interactions within the plant community. The reduction in plant competition in the grasslands maintained in a short grazing lawn, and competitive release due to canopy opening in taller vegetation, may be also important.

The Kashmir Himalaya, located at a bio-geographically pivotal position, is a unique biotic province of the Northwestern Himalaya (Rodgers and Panwar, 1988) in the Indian Himalayan State of Jammu and Kashmir. Lying between the coordinates of 32º 20′ to 34º 50′ North latitude and 73º 55′ to 75º 35′ East longitude, the region has a total area of about 15,948 km^2, nearly 64% of which is mountainous (Fig. 3.1). Topographically, the region mainly comprises a deep elliptical bowl-shaped valley bounded by the Pir Panjal range of Lesser Himalaya in the south and south-west, and the Zanskar range of the Greater Himalaya in the north and north-east. The altitude of the main valley at its summer capital, Srinagar, is 1,600 m above mean sea level (amsl), and the highest peak among its surrounding mountains is that of the 'Kolahoi' with an altitude of 5,420 m (amsl) (Hussain, 2002).

The valley is traversed by the river Jhelum and its tributaries which feed many world famous freshwater lakes, such as the Wular, Dal and Anchar lakes. Climate of the region, marked by four well-defined seasons of winter, spring, summer and autumn, resembles that of mountainous and continental parts of the temperate latitudes. The temperature ranges from an average daily maximum of 31º C and the minimum of 15º C during summer, to an average daily maximum of 4º C and the minimum of -4º C during winter. It receives an annual precipitation of about 1,050 mm, mostly in the form of snow during the winter months.

Geologically, the valley of Kashmir has been formed by folding and faulting during uplift of the Himalaya between the Indian subcontinent and the rest of Asia. A conspicuous geological feature of the region is the presence of 'Karawas', which are plateau-like tablelands formed during the Pleistocene Ice age and are composed of clay, sand and silt of lacustrine origin (de Terra, 1934).

Owing to such a wide edapho-climatic and physiographic heterogeneity, the Kashmir Himalayan region harbours diverse vegetation types, including the grasslands.

Grasslands in the Kashmir Himalaya

A large area, ca. 16%, of the Kashmir Himalaya is under grasslands, which include the grazing lands in the plains of valley, grassy vegetation on hillocks of terraced lands, and the subalpine-alpine meadows (Anon., 2012). Whereas the subalpine and alpine meadows in the region are the natural grasslands with primary vegetation, those in the plains and the leeward hillocks of side-valleys are the semi-natural ones with secondary vegetation, owing their existence mainly to a variety of management practices.

The Kashmir Himalayan grasslands are unique ecosystems. Locally known in the region as *gasse charai, margs, bahaks, neur, nyai*, etc., these grasslands are extensive, flat, undulating or sloppy stretches of land covered predominantly with herbaceous vegetation. These important lands of grass provide precious economic goods and ecosystem services, harbour a rich and endemic biodiversity, and regulate the regional carbon and hydrological cycles. Indeed, livestock rearing contributes 16% to the GDP of Jammu and Kashmir State (Anon., 2012).

The subalpine and alpine grasslands, commonly called as meadows, are covered with snow for almost 6-9 months; hence their availability for livestock grazing is limited to 3-6 months in a year. Whereas the subalpine meadows are the non-forested habitat types occurring below the tree-line and surrounded by forests, the alpine ones are vast expanses of grassy patches that occur above the tree-line. The climate of these meadows is relatively cold, wet, windy and cloudy. Onset of vegetation greenness in these grasslands occurs by the mid to late May, depending on the seasonal variation in snowfall depth and the spring temperatures. Grasses and forbs progress at a more rapid phenological rate than shrubs in these grasslands, initiating greening up and senescence earlier in the season.

Vegetation greenness is at its maximum in the early- to mid-July, and by the late August to early September, most of the vegetation ceases its growth. These meadows are characterized by a dense growth of grasses and herbaceous dicots, and sometimes dwarf shrubs (e.g. *Salix flabellaris*).

The grasslands are utilized for grazing under the following livestock rearing systems:

i) The sedentary system is practiced by the people living between altitudes of 1,500- 2,500m. The livestock is let loose in the common village grasslands for grazing, usually under the supervision of some hired person (locally known as '*goor*').The grazing continues for up to 10 months in a year.

ii) The semi-migratory systems practiced by the people living below 2,000m altitude. The professional grazers (locally known as '*pohol*') are hired; they collect the livestock and take these to the subalpine and alpine grasslands during summer. The livestock is brought back in autumn and is fed on crop residues and tree-leaf fodder during winter.

iii) The migratory system, a typical example of transhumance, in which people along with their livestock migrate continuously from one place to another in search of herbage and moderate climate. Usually the nomads, such as *Gujjars* and *Bakerwals,* stay in the hillocks of Jammu during winter and migrate up in to the grasslands during summer.

In the present study, sites were selected in such a way that they represent the major expressions of grasslands in the Kashmir Himalaya (Table 3.1). Most of the sites are characterized by gentle to steep slopes with the exception of level to undulating terrain in the valley-basin grasslands.

Table 3.1 Details of the grassland sites selected for the present study in the Kashmir Himalaya

Name of the grassland site	Abbreviation	Latitude/Longitude	Altitude (m. amsl)
Botanic Garden, University of Kashmir*	BG	34º 03′ 890″ 74º 50′ 700″	1600
Draphoma, Dachigam*	DD	34º 07′ 390″ 74º 56′ 480″	1800
Dugwan, Dachigam*	DW	34º 07′ 190″ 74º 59′ 100″	3000
Dignibal, Ungrazed*	DB_UZ	34º 12′ 260″ 74º 51′ 500″	1750
Dignibal, Grazed*	DZ_GZ	34º 12′ 260″ 74º 51′ 500″	1750
Pahlipora, Dachigam#	PD	34º 06′ 580″ 74º 58′ 300″	2000
Pahalgam#	PG	34º 01′ 860″ 75º 19′ 490″	2200
Sopore#	SP	34º 02′ 294″	1590

		74º 28′ 863″	
Gulmarg#	GM	34º 03′ 044″ 74º 23′ 889″	2650
Baltal#	BT	34º 16′ 144″ 75º 23′ 570″	3050
Sonamarg#	SM	34º 18′ 264″ 75º 16′ 489″	2690
Thajwas#	TJ	34º 17′ 900″ 75º 18′ 900″	3100
Gualta, Uri#	GT	34º 05′ 520″ 74º 01′ 630″	1280
Kamalkote, Uri#	KK	34º 07′ 320″ 74º 16′ 700″	1550
Aru, Long protected#	AR_LP	34º 05′ 567″ 75º 15′ 991″	2450
Aru, Five-year protected#	AR_FP	34º 05′ 567″ 75º 15′ 991″	2450
Aru, Grazed#	AR_GZ	34º 05′ 567″ 75º 15′ 991″	2450
Total sites=14			

*= Monthly grassland sites #= Peak-season grassland sites

4.1) Floristic diversity

Taxonomic inventory

All the grassland sites were regularly sampled for the collection of plant specimens and field sampling during the study period (April 2003 to March 2005). Standard taxonomic procedures were followed for the collection and further processing of plant specimens (Lawrence, 1951; Bridsen & Forman, 1992). The processed plant specimens were identified with the help of relevant taxonomic literature and deposited in the University of Kashmir

Herbarium (KASH). The taxonomic inventory has been arranged according to the APG-III classification system (Stevens, 2001 onwards).

For the plant species in the taxonomic inventory of grasslands, the various growth-form and life-span categories have been recognized according to the terminology proposed by Hickey & King (2000). The nine growth-forms recognized during the present study are:

(i) **Herb**: A non-woody plant, or one that is woody only at the base.
(ii) **Subshrub**: A low shrub, sometimes with partially herbaceous stems.
(iii) **Shrub**: A woody, perennial plant, generally smaller than a tree, and with several stems arising from the ground level.
(iv) **Grass**: Plants belonging to the family Poaceae.
(v) **Sedge**: Plants belonging to the family Cyperaceae.
(vi) **Rush**: Plants belonging to the family Juncaceae.
(vii) **Climber**: A plant that grows upwards by climbing on other support, or by clinging to them with the tendrils.
(viii) **Twinner**: A plant that grows upwards by twinning round nearby plants.
(ix) **Fern**: A group of flowerless plants with sporangia/sori on their leaves (fronds).

The following six life-span categories were recognised for the plant species recorded in taxonomic inventory in the present study:

(i) **Annual**: A plant that completes its life cycle within a single year.
(ii) **Biennial**: A plant that completes its life cycle within two years; producing only vegetative growth in the first year, and flowering in the second.
(iii) **Perennial:** A plant that lives for a number of years, extending above ground at least more than two years.
(iv) **Annual-Biennial**: A plant species showing both annual and biennial life spans.
(v) **Annual-Perennial:** A plant species showing both annual and perennial life spans.
(vi) **Biennial-Perennial:** A plant species showing both biennial and perennial life spans.

The flowering phenology was determined by conducting monthly visual surveys in grassland sites of the study area. The flowering phenology refers to the month(s) of the calendar year during which a plant species remains in flowering stage. It has been calculated on the basis of total number of species from the entire species pool (n=300 species, excluding the 2 fern species) which are at flowering stage during a particular month of the calendar year.

Species richness actual

The taxonomic census of species composition in each grassland site, hereafter referred to as species richness actual, was carried out by intensive survey of *ca.*100m^2 grassland plot.

This size was assumed to be large enough to serve as the basis for the community species pool (Zobel *et al*., 1998). Mostly angiosperms were encountered, although in a few grassland sites ferns were also recorded.

Species richness observed

The total number of species recorded after laying a fixed number of samples is referred to as species richness observed in the present study. For this purpose, thirty 1 m^{-2} quadrats were randomly laid at each grassland site to obtain incidence/abundance data; the quadrat was squarish in shape. No distinction was made between genet and ramets which measure incidence/abundance. An individual in some cases represented the whole genet (annuals, biennials and non-clonal perennials) whereas for other species we sampled ramets, defined as the plant part originating from one single root position. For tussock grasses, the whole tussock was taken as an individual.

Species accumulation curves (SAC), also called as collectors curves, have been plotted to compute the species richness observed. The SAC depicts the cumulative number of species recorded as a function of sampling effort (Colwell & Coddington, 1994; Gotelli & Colwell 2001). In other words, these curves compute the number of species expected in the pooled samples given the empirical data (incidence/abundance data). On the other hand, species rarefaction curves (SRC), also called as Coleman curves, plot the cumulative number of species recorded as a function of number of individuals sampled (Gotelli & Colwell, 2001). In other words, SRC represents the statistical expectation of the corresponding SAC (Magurran, 2004). While as SACs are viewed as moving from left to right, the SRCs move from right to left.

The SACs have been computed with confidence intervals: lower bound of 95% confidence interval, and upper bound of 95% confidence interval (Colwell, 2000). A confidence interval gives an estimated range of values (calculated from a given set of sample data), which is likely to include an unknown population parameter. The selection of a confidence level for an interval determines the probability that the confidence interval produced will contain the true parameter value. Usually, 95% confidence intervals are used; e.g., a 95% confidence interval covers 95% of the normal curve, i.e. the probability of observing a value outside of this area is less than 0.05 (Sokal & Rohlf, 1995).

Species richness estimated

A number of methods for estimation of species richness have been developed (Colwell & Coddington, 1994; Chazdon *et al*. 1998). These are called as non-parametric methods, because they are not based on the parameter of a species abundance model (Magurran, 2004). The following species richness estimators were used during the present study:

(i) **Chao 1**: It is based on the number of rare species in a sample. The Chao 1 species richness estimator is a function of the ratio of singletons and doubletons, and will exceed SR (observed) by ever greater margins as the relative frequency of singletons increases. It requires abundance data (Chao, 1984, 1987; Colwell, 2000).

(ii) **Chao 2**: It is a modified variant of Chao 1, which takes into account the distribution of species amongst samples. It requires incidence data (Colwell & Coddington, 1994).

(iii) **ACE** (Abundance-based Coverage Estimator): It is based on the abundances of species with between 1 and 10 individuals, the latter range selected on the basis of empirical data. The estimate is completed by adding on the number of abundant species (i.e. those represented by > 10 individuals) (Chao *et al*. 1993).

(iv) **ICE** (Incidence-based Coverage Estimator): It is based on the number of species found in ≤ 10 sampling units (Lee & Chao, 1994; Colwell & Coddington, 1994).

(v) **Jack 1** (First-order Jackknife richness estimator): It uses the number of species that occur only in a single sample (Heltshe & Forrester, 1983).

(vi) **Jack 2** (Second-order Jackknife richness estimator): It takes into account the number of species found in one sample only, as well as in two samples (Smith & van Belle, 1984).

(vii) **Bootstrap**: The estimator is based on bootstrapping techniques - a parametric procedure that reduces bias in the estimate of the population value for a statistic (Smith & van Belle, 1984; Sokal & Rohlf, 1995).

(viii) **MM** (Michaelis-Menten): There are many possible functions, asymptotic and non-asymptotic, that might fit a species accumulation curve. The asymptotic function most commonly used for richness estimation is the Michaelis-Menten (MM) function (Soberón & Llorente, 1993; Colwell & Coddington 1994; Colwell *et al*. 2004).

Species diversity

Species diversity measure − a rough proxy for biodiversity − is the traditional way of quantifying the biodiversity. It can be partitioned into two main components: species richness and species evenness (Simpson, 1949). Species richness is simply the number of species of a given taxon in the unit of study. Species evenness, on the other hand, describes the variability in species abundances in the unit of study. Magurran (2004) recommended a suite of 4 species diversity indices that combine information on richness and relative abundance in different ways. These include:

(i) **Fisher's alpha diversity index (α):** It refers to α parameter of a fitted logarithmic series distribution (Fisher *et al.* 1943). Its calculation is a necessary prelude to fitting the log series distribution model. If the number of species (S) and the total number of individuals (N) are known, the value of α can be calculated (Magurran, 2004)

(ii) **Shannon diversity index (H′):** One of the most commonly and the earliest used species diversity index, independently developed by Shannon and Wiener (Shannon & Weaver, 1949). It is based on the rationale that the diversity (or information) in a natural system can be measured in a way similar to the information contained in a message or a code (Magurran, 2004). It assumes that individuals are randomly sampled from an infinitely large community, and all the species are represented in the sample (Pielou, 1975).

(iii) **Exponential Shannon diversity index (eH′): A** more meaningful measure, as it gives the number of species that would have been found in the sample had all the species been equally common (Magurran, 2004).

(iv) **Simpson diversity index (1/D):** The index is based on the probability that any two individuals drawn at random from an infinitely large community belong to the same species (Simpson, 1949). It is usually expressed as reciprocal form (1/D). The index is biased towards the most abundant species in the sample and less sensitive to species richness.

Species evenness

Species evenness, simply, is a measure of how the species in a community are similar or different in their abundance. Therefore, a community with equally abundant species has high species evenness. Since the Shannon diversity index takes into account the degree of evenness in species abundance, it has been used to derive a separate species evenness measure. During the present study, the ratio of observed diversity to maximum diversity has been used to measure the species evenness (J′) (Pielou, 1975).

Shared species and similarity measures

During the present study, the average value of shared species estimates between each pair of months in the monthly sampled grassland sites, and each pair of samples in peak-season sampled grassland sites, have been computed. While as Jaccard’s and Sørensen’s similarity indices have been computed on the basis of incidence data, those of Morisita-Horn’s and Bray-Curtis’s similarity indices have been computed on the basis of abundance data (Chao *et al.* 2005).

Beta-diversity

Beta-diversity is a measure of the change in diversity between samples/sites/habitats. In other words, it reflects change in biotic or species composition. For measuring beta-diversity between grassland sites, Whittaker's measure (βw) was used; the values ranging from 0-1 (Whittaker, 1960). For measuring overall beta-diversity across all the grassland sites, Harrison's measure (β_{H1}) was used (Harrison *et al.* 1992), whose values range from 1-100. Both these require the incidence data.

Software used

- Estimate S Version 8.2.0 (Colwell, 2012)

4.2) Primary productivity

Primary productivity was estimated on the basis of biomass harvested on monthly basis in producer sub-compartments of grasslands (Milner & Hughes, 1968; Singh & Yadava, 1974; McNaughton *et al.*, 1996). For determining the aboveground primary productivity, five replicate quadrats of 1 m^{-2} were used (Ram *et al.* 1989). The plant material falling within these quadrats was clipped at ground level, collected in transparent polythene bags and brought to laboratory, where it was separated into live and standing dead components. After clipping of plant material, fallen litter within these quadrats was hand-picked into polythene bags and brought to laboratory, where it was further processed by floatation method (Tiwari, 1986).

For the belowground biomass, a monolith of the size 20 x 20 x 30 cm at the periphery of the quadrats was dug out, avoiding the central portion. Monoliths were then washed with a fine jet of water, using successively 2 mm and 0.5 mm mesh screens (Ram *et al.* 1989; Karunaichamy & Paliwal, 1989). After processing, all the samples of each producer sub-system (i.e., aboveground live, standing dead, litter and belowground) were kept in paper-made bags and then oven dried at 80^{o} C to constant weight for 48 hours. The oven dried samples were then weighed on a top loading electronic balance.

Monthly grassland sites

In the monthly sampled grassland sites, the annual net primary productivity (NPP) has been calculated based upon the calendar year (March-February). The length of the growing season varied widely, from as little as 3 months (in alpine, DW), 7 months (in subalpine, DD) up to 9 months (in valley-plains, BG). The aboveground net primary productivity (ANP) was calculated as the sum of positive changes in aboveground live biomass (AG); also added were the positive changes in standing dead (SD) and litter (LT) compartments. The belowground net primary productivity (BNP) was estimated as the summation of positive changes in the below-ground biomass (BG) on successive sampling dates (Singh and Yadava, 1974). Total net primary productivity (TNP) was calculated as the sum of ANP and BNP (Singh *et al*. 1980; Bawa, 1995).

Peak-season grassland sites

The sum of peak standing crop (mostly during the months of July-August) from different aboveground producer sub-systems (aboveground live, standing dead, litter) and belowground compartments of the grassland sites was used to estimate ANPp and BNPp, respectively. The sum of ANPp and BNPp was taken to estimate the TNPp.

4.3) Relationship between floristic diversity and primary productivity

The purpose of the preceding experimental protocol was to serve as a basis to investigate the floristic diversity and primary productivity in the Kashmir Himalayan grasslands, and to act as a baseline to test the relationship of floristic diversity and productivity in the grassland ecosystem. The relationship was tested for between all the variables of floristic diversity, such as SR (actual), SR (obs.), SR (est.), species diversity, species evenness and the various components of primary productivity, such as ANP, BNP and TNP.

4.4) Impacts of livestock grazing on floristic diversity and primary productivity

To investigate the impacts of livestock grazing on floristic diversity and primary productivity, two monthly sampled grassland sites at Dignibal, Srinagar: grazed and ungrazed were studied. Whereas in the former, the domestic livestock have been grazing for a long time, in the latter, protection has been provided against grazing by domestic livestock. For the same purpose, three peak-season grassland sites were selected at Aru, Pahalgam: i) Aru long-protected, the site has been protected from livestock grazing since 1986, ii) Aru five-year protected, the site has been protected from livestock grazing from last 5 years, and iii) Aru grazed, the site was grazed by domestic livestock for long.

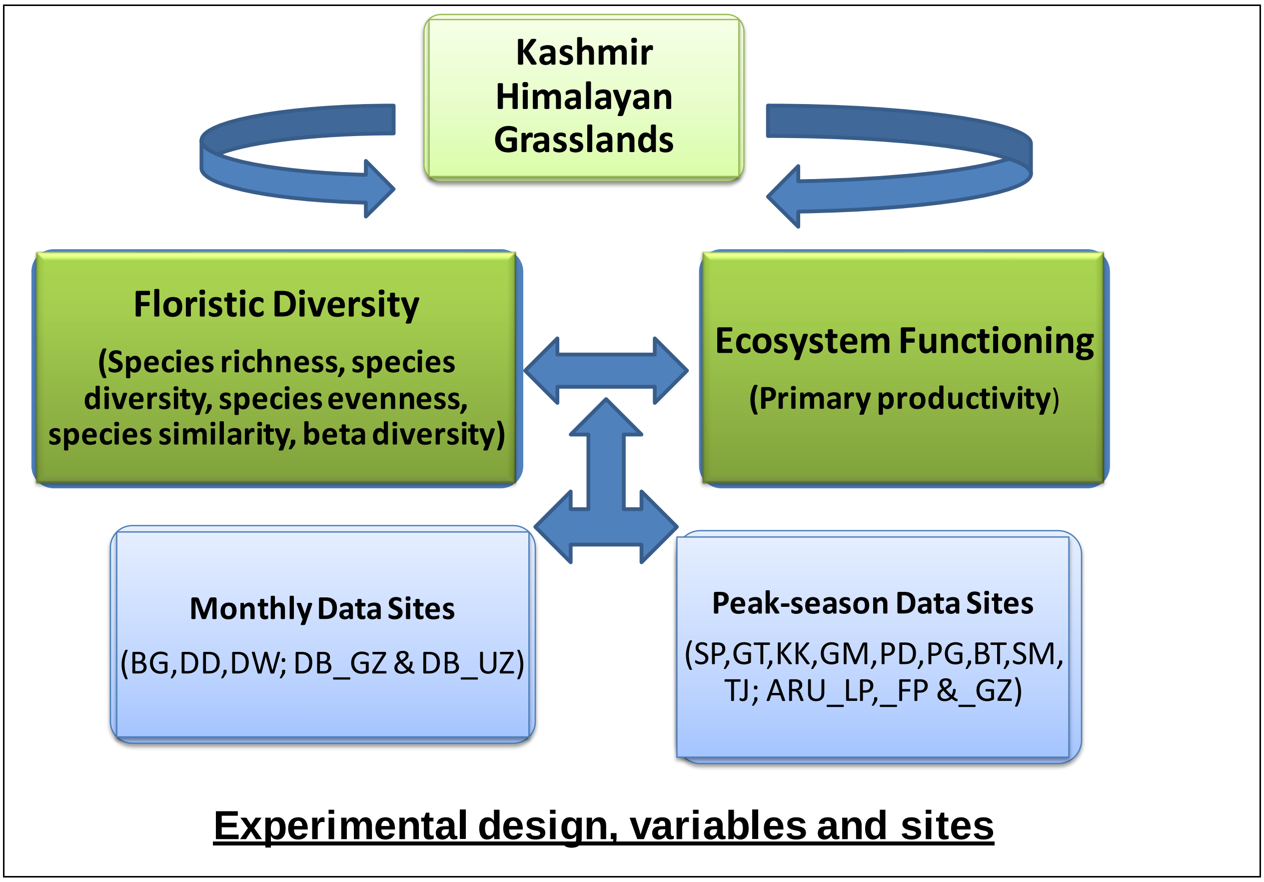

Fig. 4.1 Flow-chart showing the experimental design, variables of floristic diversity, ecosystem functioning investigated, and grassland sites in the Kashmir Himalaya

Table 4.1 Summary of sampling carried out to investigate the floristic diversity and primary productivity in grassland sites of the Kashmir Himalaya

Name of the grassland site	Abbrevia-tion	Total no. of quadrats laid (Biomass)	Soil cores dug	Total no. of quadrats laid (Incidence/ Abundance)
Botanic Garden, University of Kashmir*	BG	60	60	120
Draphoma, Dachigam*	DD	60	60	120
Dugwan, Dachigam*	DW	15	15	30
Dignibal, Ungrazed*	DB_UZ	60	60	120
Dignibal, Grazed*	DZ_GZ	60	60	120
Pahlipora, Dachigam#	PD	5	5	30
Pahalgam#	PG	5	5	30
Sopore#	SP	5	5	30
Gulmarg#	GM	5	5	30
Baltal#	BT	5	5	30
Sonamarg#	SM	5	5	30
Thajwas#	TJ	5	5	30
Gualta,Uri#	GT	5	5	30
Kamalkote, Uri#	KK	5	5	30
Aru, Long protected#	AR_LP	5	5	30
Aru, Five-year protected#	AR_FP	5	5	30
Aru, Grazed#	AR_GZ	5	5	30
Total sites=14		**315**	**315**	**870**

*= Monthly grassland sites
#= Peak-season grassland sites

5.1) Floristic diversity

Taxonomic inventory

During the present study, 302 species belonging to 199 genera and 52 families were recorded from all the grassland sites in Kashmir Himalaya. All the plant species recorded belong to angiosperms, with the exception of 2 fern species, namely: *Adiantum capillus-veneris* and *Ophioglossum vulgatum* (Table 5.1, Fig. 5.1).

Table 5.1 Taxonomic inventory comprising of scientific name with author citation, synonym(s) and family of each species recorded from all the grassland sites in the Kashmir Himalaya

Scientific name	**Synonym (s)**	**Family**
Achillea millefolium L.		Asteraceae
Adiantum capillus-veneris L.		Adiantaceae
Aegilops tauschii Cosson		Poaceae
Agrimonia pilosa Hk. f.		Rosaceae
Agrostis stolonifera L.		Poaceae
Ainsliaea aptera DC.		Asteraceae
Ajuga parviflora Bth.		Lamiaceae
Alchemilla ypsilotoma Rothm.		Rosaceae
Anagallis arvensis L.		Primulaceae
Anaphalis royleana DC.		Asteraceae
Anaphalis triplinervis Clarke		Asteraceae
Anchusa arvensis subsp. *orientalis* (L.) Nordh.	*Lycopsis orientalis* L.	Boraginaceae
Androsace rotundifolia Hardw.		Primulaceae
Androsace sarmentosa subsp. *primuloides* (Duby) Govaerts	*Androsace primuloides* Duby	Primulaceae
Androsace sempervivoides Jacq. ex Duby		Primulaceae
Anemone tschernjaewii Regel	*Anemone biflora* DC.	Ranunculaceae
Anemone obtusiloba D. Don		Ranunculaceae
Angelica glauca Edgew.		Apiaceae
Anthemis cotula L.		Asteraceae
Aquilegia fragrans Bth.		Ranunculaceae
Arabidopsis thaliana Heynh.		Brassicaceae
Arabis amplexicaulis Edgew.		Brassicaceae
Arabis nova Vill.		Brassicaceae
Arabis pterosperma Edgew.		Brassicaceae
Arenaria serpyllifolia L.		Caryophyllaceae
Arisaema jacquemontii Blume		Araceae
Artemisia absinthium L.		Asteraceae
Artemisia annua L.		Asteraceae
Artemisia indica Willd.		Asteraceae
Artemisia scoparia Waldst. & Kit.		Asteraceae
Arthraxon prionodes Dandy.		Poaceae
Asparagus filicinus Ham.		Asparagaceae
Aster asteroides O. Ktze		Asteraceae
Aster falconeri Hutch.		Asteraceae
Astragalus densiflorus Kar. & Kir.		Fabaceae
Astragalus grahamianus Royle ex Bth.		Fabaceae
Astragalus stewartii Baker		Fabaceae
Asyneuma thomsonii Bornm.		Campanualceae
Avena sativa L.		Poaceae
Bellis perennis L.		Asteraceae

Bergenia stracheyi Engl.		Saxifragaceea
Bidens cernua L.		Asteraceae
Bothriochloa ischaemum Keng		Poaceae
Brachyactis pubescens Aitch.		Asteraceae
Bromus arvensis L.		Poaceae
Bromus inermis Leyss.		Poaceae
Bromus japonicus Thunb.		Poaceae
Bupleurum longicaule Wall. ex DC.		Apiaceae
Calamintha umbrosa Fisch. & Mey.		Lamiaceae
Caltha palustris var. *alba* (Cambess) Hook.f. & Thomson	*Caltha alba* Cambess.	Ranunculaceae
Campanula latifolia L.		Campanualceae
Cannabis sativa L.		Cannabaceae
Capsella bursa-pastoris Medic.		Brassicaceae
Cardamine impatiens L.		Brassicaceae
Carduus edelbergii Rech. f.		Asteraceae
Carex fedia Nees		Cyperaceae
Carex nivalis Boott		Cyperaceae
Carex nubigena D. Don		Cyperaceae
Carex setosa Boott		Cyperaceae
Carex wallichiana Prescott ex. Wall.		Cyperaceae
Carpesium cernuum L.		Asteraceae
Carum carvi L.		Apiaceae
Centaurea iberica Trev. ex Spreng.		Asteraceae
Cerastium cerastoides (L.) Britton		Caryophyllaceae
Chaerophyllum villosum Wall. ex DC.		Apiaceae
Chenopodium album L.		Amaranthaceae
Chrysopogon echinulatus W. Wats.		Poaceae
Cichorium intybus L.		Asteraceae
Cirsium falconeri (Hook.f.) Petr.		Asteraceae
Clinopodium vulgare L.		Lamiaceae
Codonopsis ovata Bth.		Campanualceae
Conium maculatum L.		Apiaceae
Convolvulus arvensis L.		Convolvulaceae
Conyza canadensis Cronq.		Asteraceae
Conyza japonica (Thunb.) Less. ex DC.	*Conyza multicaulis* DC.	Asteraceae
Corydalis cashmeriana Royle		Papaveraceae
Crambe cordifolia Boiss.		Brassicaceae
Crepis kashmirica Babcock		Asteraceae
Crepis sancta Babcock		Asteraceae
Cynodon dactylon Pers.		Poaceae
Cynoglossum wallichii var. *glochidiatum* (Wall. ex Benth.) Kazmi	*Cynoglossum glochidiatum* Wall. ex Benth.	Boraginaceae
Cyperus iria L.		Cyperaceae
Cyperus rotundus L.		Cyperaceae
Cypripedium cordigerum D. Don		Orchidaceae
Dactylis glomerata L.		Poaceae
Dactylorhiza hatagirea (D.Don) Soó		Orchidaceae
Daucus carota L.		Apiaceae
Delphinium roylei Munz		Ranunculaceae
Digitaria cruciata A. Camus		Poaceae
Dioscorea deltoidea Wall. ex Kunth		Dioscoreaceae
Dipsacus inermis Wall.		Dipsacaceae

Doronicum kamaonense (DC.) Alv.Fern.	*Doronicum roylei* DC.	Asteraceae
Draba lanceolata Royle		Brassicaceae
Duchesnea indica Focke		Rosaceae
Epilobium laxum Royle		Onagraceae
Eremogone griffithii (Boiss.) Ikonn.	*Arenaria griffithii* Boiss.	Caryophyllaceae
Eritrichium canum (Benth.) Kitam.	*Eritrichium strictum* Decne.	Boraginaceae
Erodium cicutarium L' Herit. ex Ait.		Geraniaceae
Eryngium billardierei F.Delaroche		Apiaceae
Erysimum melicentae Dunn.		Brassicaceae
Euphorbia cornigera Boiss.		Euphorbiaceae
Euphorbia helioscopia L.		Euphorbiaceae
Euphorbia hispida Boiss.	*Euphorbia emodi* Hk.f.	Euphorbiaceae
Euphorbia wallichii Hk. f.		Euphorbiaceae
Fimbristylis dichotoma Vahl.		Cyperaceae
Fritillaria cirrhosa D.Don	*Fritillaria roylei* Hook.	Liliaceae
Fragaria nubicola Lindl. ex Lac.		Rosaceae
Fumaria indica Haussk.		Papaveraceae
Gagea lutea (L.) Ker Gawl.	*Gagea elegans* Wall. ex G.Don	Liliaceae
Galium aparine L.		Rubiaceae
Galium asperuloides Edgew.		Rubiaceae
Galium boreale L.		Rubiaceae
Galium pauciflorum Bunge		Rubiaceae
Galium tricornutum Dandy	*Galium tricorne* Stokes	Rubiaceae
Galium verum L.		Rubiaceae
Gaultheria trichophylla Royle		Ericaceae
Gentiana capitata Ham. ex D. Don		Gentianaceae
Gentiana carinata Griseb.		Gentianaceae
Gentiana kurroo Royle		Gentianaceae
Gentiana membranulifera T.N.Ho	*Gentiana marginata* Griseb.	Gentianaceae
Geranium pratense L.		Geraniaceae
Geranium pusillum Burm. f.		Geraniaceae
Geranium wallichianum D.Don ex Sweet		Geraniaceae
Gerbera gossypina Beauv.		Asteraceae
Geum elatum G. Don		Rosaceae
Geum urbanum L.		Rosaceae
Hackelia macrophylla I. M. J.		Boraginaceae
Hackelia uncinata C. E. C. Fisch.		Boraginaceae
Herminium lanceum (Thunb. ex Sw.) Vuikj	*Spiranthes lancea* (Thunb. ex Sw.) Bakh.f. & Steenis	Orchidaceae
Hypericum perforatum L.		Hypericaceae
Impatiens edgeworthii Hk. f.		Balsaminaceae
Indigofera heterantha Wall. ex Brand		Fabaceae
Inula royleana Clarke		Asteraceae
Iris ensata Thunb.		Iridaceae
Iris hookeriana Foster		Iridaceae
Juncus articulatus L.		Juncaceae
Juncus bufonius L.		Juncaceae
Juncus himalensis Kl. & Garcke		Juncaceae
Jurinea macrocephala Hk. f.		Asteraceae
Lactuca dissecta D. Don		Asteraceae
Lactuca serriola L.	*Lactuca scariola* L.	Asteraceae
Lagotis cashmeriana Rupr.		Plantaginaceae
Lathyrus aphaca L.		Fabaceae

Leontopodium jacotianum Beauv.		Asteraceae
Leontopodium leontopodinum Hand. Mazz.		Asteraceae
Leonurus cardiaca L.		Lamiaceae
Lepidium capitatum H. & T.		Brassicaceae
Lepidium didymum L.	*Coronopus didymus* (L.) Sm.	Brassicaceae
Lespedeza elegans Camb.		Fabaceae
Lespedeza juncea var. *sericea* (Thunb.) Lace & Hauech	*Lespedeza cuneata* (Dum.Cours.) G.Don	Fabaceae
Leucanthemum vulgare (Vaill.) Lam.	*Chrysanthemum leucanthemum* L.	Asteraceae
Lithospermum arvense L.		Boraginaceae
Lolium perenne L.		Poaceae
Lolium temulentum L.		Poaceae
Lomatogonium carinthiacum A. Br.		Gentianaceae
Lotus corniculatus L.		Fabaceae
Lycopus europaeus L.		Lamiaceae
Malva neglecta Wall.		Malvaceae
Malva sylvestris L.		Malvaceae
Marrubium vulgare L.		Lamiaceae
Mazus japonicus Kuntze		Phrymaceae
Medicago lupulina L.		Fabaceae
Medicago minima Grufb.		Fabaceae
Medicago polymorpha L.		Fabaceae
Mentha longifolia Huds.		Lamiaceae
Micromeria biflora Bth.		Lamiaceae
Milium effusum L.		Poaceae
Morina longifolia Wall. ex DC.		Dipsacaceae
Myosotis caespitosa Schultz.		Boraginaceae
Myosotis palustris (L.) Nath.		Boraginaceae
Myosotis stricta Link ex Roem. & Schult.	*Myosotis micrantha* Pall. ex Lehm.	Boraginaceae
Myosotis sylvatica Hoffm.		Boraginaceae
Myriactis nepalensis Less.		Asteraceae
Myriactis wallichii Less.		Asteraceae
Nasturtium officinale R. Br.		Brassicaceae
Nepeta cataria L.		Lamiaceae
Nepeta laevigata (D.Don) Hand.-Mazz.	*Nepeta spicata* Benth.	Lamiaceae
Nepeta raphanorhiza Bth.		Lamiaceae
Oenothera glazioviana Micheli		Onagraceae
Oenothera rosea Ait.		Onagraceae
Ophioglossum vulgatum L.		Ophioglossaceae
Origanum vulgare L.		Lamiaceae
Oryzopsis munroi Stapf ex Hook.f.		Poaceae
Oxalis acetosella L.		Oxalidaceae
Oxalis corniculata L.		Oxalidaceae
Oxyria digyna Hill		Polygonaceae
Oxytropis lapponica Gay		Fabaceae
Papaver dubium L.		Papaveraceae
Pedicularis punctata Dcne.		Orobanchaceae
Pennisetum flaccidum Griseb.		Poaceae
Pennisetum orientale L. C. Rich.		Poaceae
Persicaria hydropiper (L.) Delarbre	*Polygonum hydropiper* L.	Polygonaceae
Persicaria nepalensis (Meisn.) Miyabe	*Polygonum nepalense* Meisn.	Polygonaceae
Phalaris arundinacea L.		Poaceae

Phelum alpinum L.		Poaceae
Phlomis bracteosa Royle ex Bth.		Lamiaceae
Phlomis cashmeriana Royle ex Bth.		Lamiaceae
Plantago depressa Willd.		Plantaginaceae
Plantago himalaica Pilger		Plantaginaceae
Plantago lanceolata L.		Plantaginaceae
Plantago major L.		Plantaginaceae
Poa alpina L.		Poaceae
Poa angustifolia L.		Poaceae
Poa annua L.		Poaceae
Poa bulbosa L.		Poaceae
Poa pratensis L.		Poaceae
Poa stewartiana Bor		Poaceae
Polemonium caeruleum L.		Polemoniaceae
Polygala sibirica L.		Polygalaceae
Polygonum affine D. Don		Polygonaceae
Polygonum amplexicaule D. Don		Polygonaceae
Polygonum aviculare L.		Polygonaceae
Polygonum plebeium R.Br.		Polygonaceae
Polygonum rumicifolium Royle ex Bab.		Polygonaceae
Polypogon fugax Nees ex Steud.		Poaceae
Potentilla argentea L.		Rosaceae
Potentilla argyrophylla Wall. ex Lehm.		Rosaceae
Potentilla biflora Willd. ex Sch.		Rosaceae
Potentilla doubjouneana Camb.		Rosaceae
Potentilla supina L.		Rosaceae
Primula denticulata Sm.		Primulaceae
Primula rosea Royle		Primulaceae
Prunella vulgaris L.		Lamiaceae
Pseudostellaria himalaica Pax		Caryophyllaceae
Ranunculus arvensis L.		Ranunculaceae
Ranunculus diffusus DC.		Ranunculaceae
Ranunculus distans Royle	*Ranunculus laetus* Wall. ex Royle	Ranunculaceae
Ranunculus muricatus L.		Ranunculaceae
Ranunculus palmatifidus H. Riedl		Ranunculaceae
Ranunculus sceleratus L.		Ranunculaceae
Rorippa islandica (Oeder) Borbás		Brassicaceae
Rosa macrophylla Lindl.		Rosaceae
Rubia cordifolia L.		Rubiaceae
Rumex dentatus L.		Polygonaceae
Rumex hastatus D. Don		Polygonaceae
Rumex nepalensis Spreng.		Polygonaceae
Saccharum filifolium Steud.		Poaceae
Salix flabellaris N. J. And.		Salicaceae
Salvia hians Royle ex Bth.		Lamiaceae
Salvia moorcroftiana Wall. ex Bth.		Lamiaceae
Sambucus wightiana Wall.		Adoxaceae
Saussurea albescens Sch.		Asteraceae
Saussurea heteromalla Hand. Mazz.		Asteraceae
Saxifraga sibirica L.		Saxifragaceea
Scandix pecten-veneris L.		Apiaceae
Scrophularia polyantha Royle ex Bth.		Scrophulariaceae

Scutellaria prostrata Jacq. ex Bth.		Lamiaceae
Senecio laetus Edgew.	*Senecio chrysanthemoides* DC.	Asteraceae
Setaria viridis P. Beauv.		Poaceae
Sibbaldia cuneata Kunze		Rosaceae
Silene apetala Willd.		Caryophyllaceae
Silene baccifera (L.) Roth	*Cucubalus baccifer* L.	Caryophyllaceae
Silene coronaria (Desr.) Clairv. ex Rchb.	*Lychnis coronaria* Desr.	Caryophyllaceae
Silene vulgaris Garcke		Caryophyllaceae
Sisymbrium irio L.		Brassicaceae
Sisymbrium loeselii L.		Brassicaceae
Sisymbrium officinale (L.) Scop.		Brassicaceae
Sium latijugum Clarke		Apiaceae
Solanum americanum Mill.	*Solanum nigrum* L.	Solanaceae
Solanum pseudocapsicum L.		Solanaceae
Solenanthus circinatus Ledeb.		Boraginaceae
Solidago virgaurea L.		Asteraceae
Sonchus asper Hill.		Asteraceae
Sonchus oleraceus L.		Asteraceae
Sorghum bicolor (L.) Moench	*Sorghum vulgare* Pers.	Poaceae
Stachys floccosa Bth.		Lamiaceae
Stachys sericea Wall. ex Bth.		Lamiaceae
Stellaria media Cyr.		Caryophyllaceae
Stellaria uliginosa Murray	*Stellaria alsine* Grimm	Caryophyllaceae
Stipa sibirica Lamk.		Poaceae
Swertia petiolata D. Don		Gentianaceae
Swertia tetragona Clarke		Gentianaceae
Tagetes minuta L.		Asteraceae
Taraxacum officinale Weber		Asteraceae
Thalictrum minus L.		Ranunculaceae
Thalictrum pedunculatum Edgew.		Ranunculaceae
Themeda anathera Hack.		Poaceae
Thlaspi arvense L.		Brassicaceae
Thlaspi cochlearioides H. & T.		Brassicaceae
Thlaspi griffithianum Boiss.		Brassicaceae
Thymus serpyllum L.		Lamiaceae
Torilis japonica DC.		Apiaceae
Tragopogon dubius Scop.		Asteraceae
Tragopogon kashmirianus G. Singh		Asteraceae
Trifolium pratense L.		Fabaceae
Trifolium repens L.		Fabaceae
Trigonella emodi Bth.		Fabaceae
Tulipa clusiana DC.	*Tulipa stellata* Hook.	Liliaceae
Urtica dioica L.		Urticaceae
Valeriana jatamansi Jones	*Valeriana wallichii* DC.	Caprifoliaceae
Valerianella dentata Poll.		Caprifoliaceae
Verbascum thapsus L.		Scrophulariaceae
Verbena officinalis L.		Verbenaceae
Veronica arvensis L.		Plantaginaceae
Veronica beccabunga L.		Plantaginaceae
Veronica biloba L.		Plantaginaceae
Veronica persica Poir.		Plantaginaceae
Vicia sativa L.		Fabaceae
Viola biflora L.		Violaceae

Viola canescens Wall. ex Roxb.		Violaceae
Viola odorata L.		Violaceae
Xanthium strumarium L.		Asteraceae

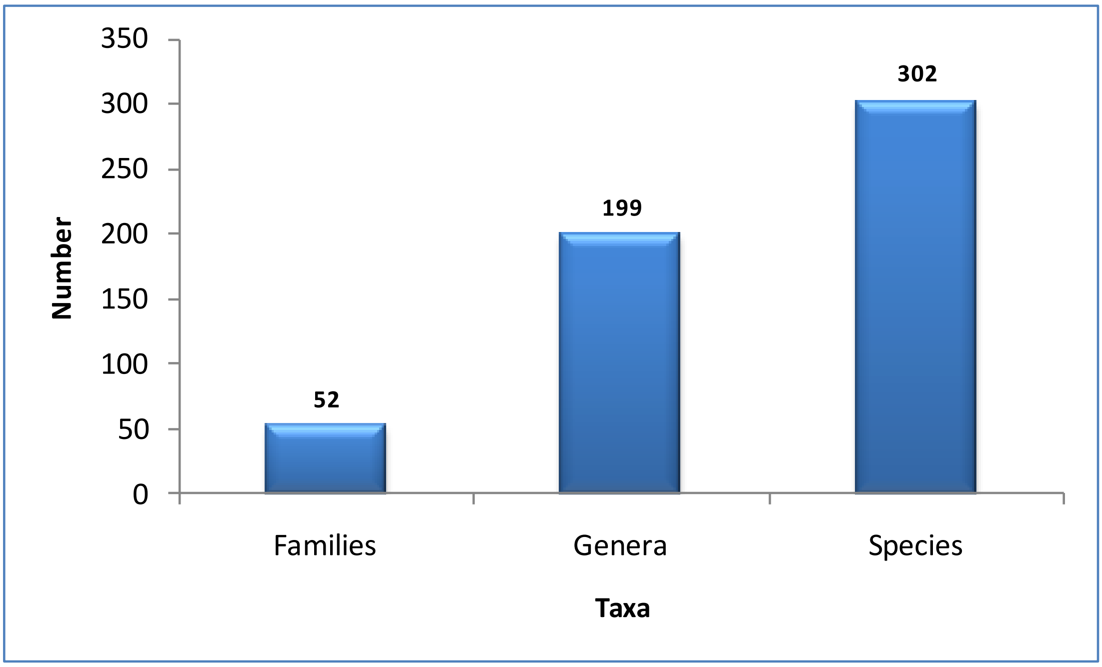

Fig. 5.1 Total number of families, genera and species documented from all the grassland sites in the Kashmir Himalaya

Out of the total families recorded, Asteraceae is the dominant family represented by 45 species, followed by Poaceae and Lamiaceae with 32 and 21 species, respectively. Of the total 302 species, the first 15 largest families contribute relatively higher number of 232 species, and the remaining 37 families share 70 species (Fig. 5.2). While as on the lower extreme, there are 19 families which are represented by a single species, there is just a single family with more than 40 species. A decreasing trend is evident: with more number of families with lesser number of species and vice versa (Fig. 5.3).

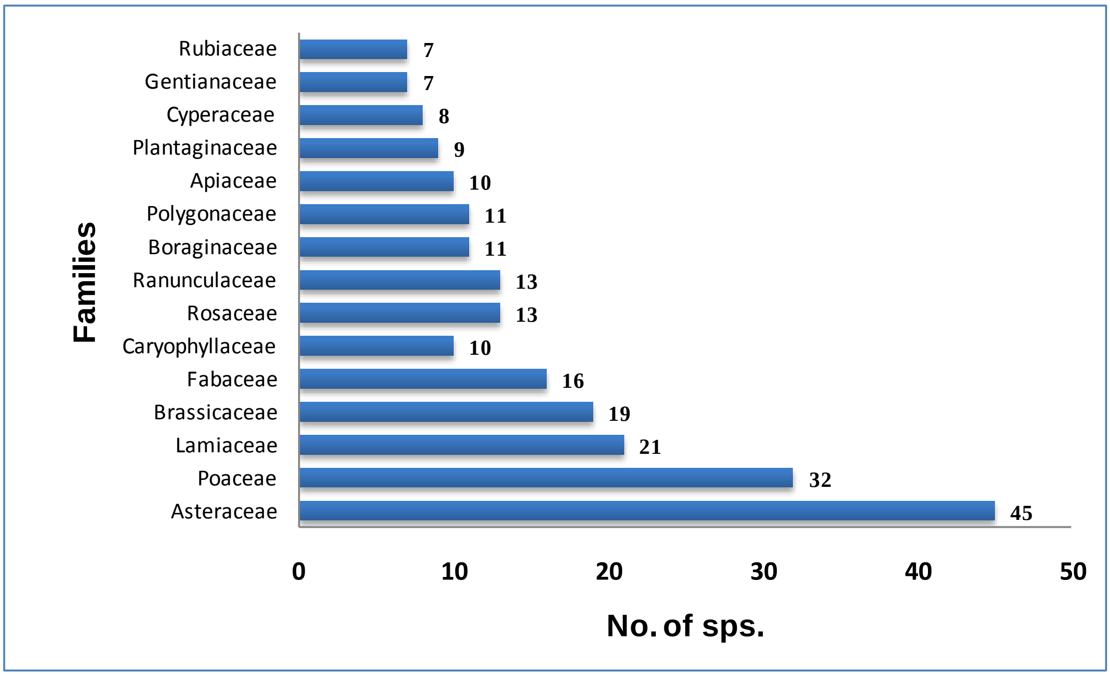

Fig. 5.2 The first fifteen families in terms of number of species recorded from all the grassland sites in the Kashmir Himalaya

Out of the total genera recorded, the highest number of 6 species is respresented by 3 genera *Galium*, *Poa* and *Ranunculus*, followed by 5 species in *Carex*, *Polygonum* and *Potentilla* (Table 5.1). While as 142 genera are represented by a single species, there are 33, 11, 7, 3 and 3 genera which are represented by 2, 3, 4, 5 and 6 species, respectively. As in case of families, there is a decreasing trend of more number of genera with lesser number of species and vice versa (Fig. 5.4).

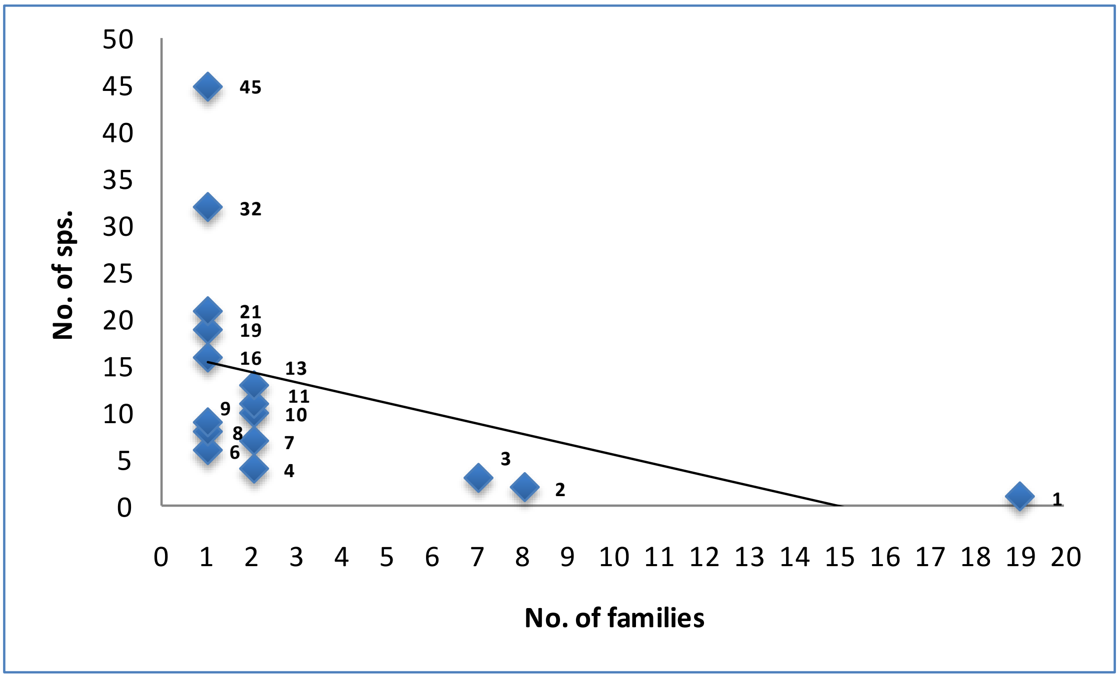

Fig. 5.3 Distribution of species within families in the species pool of all the grassland sites in the Kashmir Himalaya

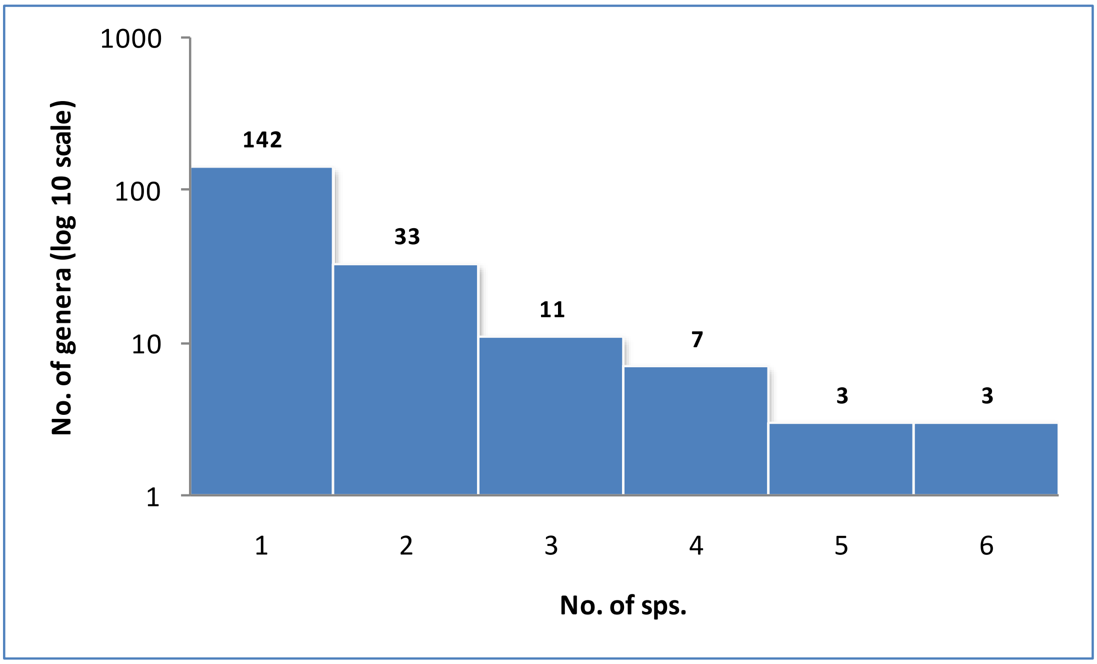

Fig. 5.4 Representation of genera with varying number of species in the species pool of all the grassland sites in the Kashmir Himalaya

Growth form, life span and flowering phenology

The plant species recorded in the grassland sites possess different types of growth forms, life spans and flowering phenology (Table 5.2).

Table 5.2 Growth-form, life-span and flowering phenology of the species pool recorded in all the grassland sites in Kashmir Himalaya

Name of the taxa	Growth form	Life span	Flowering month (s)
Achillea millefolium L.	Herb	Perennial	June-August
Adiantum capillus-veneris L.	Fern	Perennial	
Aegilops tauschii Cosson	Grass	Annual	June-July
Agrimonia pilosa Hk. f.	Herb	Perennial	July-August
Agrostis stolonifera L.	Grass	Perennial	June-August
Ainsliaea aptera DC.	Herb	Perennial	May-June
Ajuga parviflora Bth.	Herb	Annual/Perennial	May-July
Alchemilla ypsilotoma Rothm.	Herb	Perennial	July-August
Anagallis arvensis L.	Herb	Annual	March-October
Anaphalis royleana DC.	Herb	Perennial	July-September
Anaphalis triplinervis Clarke	Herb	Perennial	July-August
Anchusa arvensis subsp. *orientalis* (L.) Nordh.	Herb	Annual	May-July
Androsace rotundifolia Hardw.	Herb	Perennial	May-June

Androsace sarmentosa subsp. *primuloides* (Duby) Govaerts	Herb	Perennial	July-August
Androsace sempervivoides Jacq. ex Duby	Herb	Perennial	July-August
Anemone tschernjaewii Regel	Herb	Perennial	March-April
Anemone obtusiloba D. Don	Herb	Perennial	May-July
Angelica glauca Edgew.	Herb	Perennial	June-July
Anthemis cotula L.	Herb	Annual	April-July
Aquilegia fragrans Bth.	Herb	Perennial	June-July
Arabidopsis thaliana Heynh.	Herb	Annual	April-October
Arabis amplexicaulis Edgew.	Herb	Perennial	April-May
Arabis nova Vill.	Herb	Annual	May-June
Arabis pterosperma Edgew.	Herb	Perennial	June-July
Arenaria serpyllifolia L.	Herb	Annual	March-September
Arisaema jacquemontii Blume	Herb	Perennial	May-June
Artemisia absinthium L.	Herb	Perennial	June-July
Artemisia annua L.	Herb	Annual	July-August
Artemisia indica Willd.	Herb	Perennial	August-September
Artemisia scoparia Waldst. & Kit.	Herb	Perennial	July-August
Arthraxon prionodes Dandy.	Grass	Perennial	June-July
Asparagus filicinus Ham.	Herb	Perennial	May-June
Aster asteroides O. Ktze	Herb	Perennial	June-August
Aster falconeri Hutch.	Herb	Perennial	July-August
Astragalus densiflorus Kar. & Kir.	Herb	Perennial	June-July
Astragalus grahamianus Royle ex Bth.	Subshrub	Perennial	March-May
Astragalus stewartii Baker	Subshrub	Perennial	August-September
Asyneuma thomsonii Bornm.	Herb	Perennial	July-August
Avena sativa L.	Grass	Annual	April-May
Bellis perennis L.	Herb	Perennial	February-May
Bergenia stracheyi Engl.	Herb	Perennial	June-July
Bidens cernua L.	Herb	Annual	August-September
Bothriochloa ischaemum Keng	Grass	Perennial	June-August
Brachyactis pubescens Aitch.	Herb	Perennial	July-September
Bromus arvensis L.	Grass	Annual	May-July
Bromus inermis Leyss.	Grass	Perennial	July-August
Bromus japonicus Thunb.	Grass	Annual	June-July
Bupleurum longicaule Wall. ex DC.	Herb	Perennial	July-August
Calamintha umbrosa Fisch. & Mey.	Herb	Perennial	May-August
Caltha palustris var. *alba* (Cambess) Hook.f. & Thomson	Herb	Perennial	May-July
Campanula latifolia L.	Herb	Perennial	June-August
Cannabis sativa L.	Herb	Annual	June-August
Capsella bursa-pastoris Medic.	Herb	Annual	February-November
Cardamine impatiens L.	Herb	Biennial	May-July
Carduus edelbergii Rech. f.	Herb	Perennial	July-September
Carex fedia Nees	Sedge	Perennial	May-June
Carex nivalis Boott	Sedge	Perennial	August-September
Carex nubigena D. Don	Sedge	Perennial	June-August
Carex setosa Boott	Sedge	Perennial	July-August
Carex wallichiana Prescott ex. Wall.	Sedge	Perennial	June-July
Carpesium cernuum L.	Herb	Annual	July-August
Carum carvi L.	Herb	Perennial	July-September
Centaurea iberica Trev. ex Spreng.	Herb	Perennial	June-July
Cerastium cerastoides (L.) Britton	Herb	Perennial	May-July

Chaerophyllum villosum Wall. ex DC.	Herb	Perennial	July-August
Chenopodium album L.	Herb	Annual	August-September
Chrysopogon echinulatus W. Wats.	Grass	Perennial	July-August
Cichorium intybus L.	Herb	Perennial	May-August
Cirsium falconeri (Hook.f.) Petr.	Herb	Perennial	August-September
Clinopodium vulgare L.	Herb	Perennial	May-June
Codonopsis ovata Bth.	Herb	Perennial	July-August
Conium maculatum L.	Herb	Biennial	July-August
Convolvulus arvensis L.	Twinner	Perennial	April-October
Conyza canadensis Cronq.	Herb	Annual	August-September
Conyza japonica (Thunb.) Less. ex DC.	Herb	Annual/Biennial	July-August
Corydalis cashmeriana Royle	Herb	Perennial	May-July
Crambe cordifolia Boiss.	Herb	Perennial	April-May
Crepis kashmirica Babcock	Herb	Perennial	June-August
Crepis sancta Babcock	Herb	Annual	March-June
Cynodon dactylon Pers.	Grass	Perennial	May-September
Cynoglossum wallichii var. *glochidiatum* (Wall. ex Benth.) Kazmi	Herb	Biennial	May-July
Cyperus iria L.	Sedge	Annual	June-August
Cyperus rotundus L.	Sedge	Perennial	April-September
Cypripedium cordigerum D. Don	Herb	Perennial	June-July
Dactylis glomerata L.	Grass	Perennial	June-August
Dactylorhiza hatagirea (D.Don) Soó	Herb	Perennial	June-July
Daucus carota L.	Herb	Biennial	July-August
Delphinium roylei Munz	Herb	Perennial	June-July
Digitaria cruciata A. Camus	Grass	Annual	July-August
Dioscorea deltoidea Wall. ex Kunth	Climber	Perennial	May-June
Dipsacus inermis Wall.	Herb	Perennial	July-August
Doronicum kamaonense (DC.) Alv.Fern.	Herb	Perennial	June-August
Draba lanceolata Royle	Herb	Perennial	May-July
Duchesnea indica Focke	Herb	Perennial	May-September
Epilobium laxum Royle	Herb	Perennial	July-August
Eremogone griffithii (Boiss.) Ikonn.	Herb	Perennial	June-July
Eritrichium canum (Benth.) Kitam.	Herb	Perennial	June-July
Erodium cicutarium L' Herit. ex Ait.	Herb	Annual	May-August
Eryngium billardierei F.Delaroche	Herb	Perennial	June-August
Erysimum melicentae Dunn.	Herb	Biennial	May-June
Euphorbia cornigera Boiss.	Herb	Perennial	July-August
Euphorbia helioscopia L.	Herb	Annual	April-September
Euphorbia hispida Boiss.	Herb	Annual	August-September
Euphorbia wallichii Hk. f.	Herb	Perennial	June-August
Fimbristylis dichotoma Vahl.	Sedge	Perennial	August-September
Fritillaria cirrhosa D.Don	Herb	Perennial	June-July
Fragaria nubicola Lindl. ex Lac.	Herb	Perennial	May-August
Fumaria indica Haussk.	Herb	Annual	April-July
Gagea lutea (L.) Ker Gawl.	Herb	Perennial	March-May
Galium aparine L.	Herb	Annual	April-August
Galium asperuloides Edgew.	Herb	Annual	May-July
Galium boreale L.	Herb	Perennial	June-August
Galium pauciflorum Bunge	Herb	Annual	May-July
Galium tricornutum Dandy	Herb	Annual	April-June
Galium verum L.	Herb	Annual	May-August
Gaultheria trichophylla Royle	Subshrub	Perennial	June-July

Gentiana capitata Ham. ex D. Don	Herb	Annual	June-July
Gentiana carinata Griseb.	Herb	Annual	May-July
Gentiana kurroo Royle	Herb	Perennial	September-October
Gentiana membranulifera T.N.Ho	Herb	Annual	July-August
Geranium pratense L.	Herb	Perennial	July-August
Geranium pusillum Burm. f.	Herb	Annual	May-July
Geranium wallichianum D.Don ex Sweet	Herb	Perennial	July-August
Gerbera gossypina Beauv.	Herb	Perennial	May-June
Geum elatum G. Don	Herb	Perennial	June-August
Geum urbanum L.	Herb	Perennial	April-May
Hackelia macrophylla I. M. J.	Herb	Perennial	May-July
Hackelia uncinata C. E. C. Fisch.	Herb	Perennial	July-August
Herminium lanceum (Thunb. ex Sw.) Vuikj	Herb	Perennial	June-August
Hypericum perforatum L.	Herb	Perennial	June-July
Impatiens edgeworthii Hk. f.	Herb	Annual	July-September
Indigofera heterantha Wall. ex Brand	Shrub	Perennial	May-July
Inula royleana Clarke	Herb	Perennial	July-August
Iris ensata Thunb.	Herb	Perennial	April-May
Iris hookeriana Foster	Herb	Perennial	May-July
Juncus articulatus L.	Rush	Perennial	June_August
Juncus bufonius L.	Rush	Annual	May-July
Juncus himalensis Kl. & Garcke	Rush	Perennial	July-August
Jurinea macrocephala Hk. f.	Herb	Perennial	July-September
Lactuca dissecta D. Don	Herb	Annual	June-August
Lactuca serriola L.	Herb	Annual/Biennial	July-August
Lagotis cashmeriana Rupr.	Herb	Perennial	June-July
Lathyrus aphaca L.	Climber	Annual	April-May
Leontopodium jacotianum Beauv.	Herb	Perennial	June-August
Leontopodium leontopodinum Hand. Mazz.	Herb	Perennial	July-September
Leonurus cardiaca L.	Herb	Perennial	July-August
Lepidium capitatum H. & T.	Herb	Annual/Biennial	April-June
Lepidium didymum L.	Herb	Annual/Biennial	March-May
Lespedeza elegans Camb.	Subshrub	Perennial	July-September
Lespedeza juncea var. *sericea* (Thunb.) Lace & Hauech	Herb	Perennial	July-August
Leucanthemum vulgare (Vaill.) Lam.	Herb	Perennial	May-July
Lithospermum arvense L.	Herb	Annual	March-May
Lolium perenne L.	Grass	Perennial	June-July
Lolium temulentum L.	Grass	Annual	April-June
Lomatogonium carinthiacum A. Br.	Herb	Annual	August-September
Lotus corniculatus L.	Herb	Perennial	July-August
Lycopus europaeus L.	Herb	Perennial	July-September
Malva neglecta Wall.	Herb	Perennial	April-September
Malva sylvestris L.	Herb	Biennial	June-August
Marrubium vulgare L.	Herb	Perennial	May-September
Mazus japonicus Kuntze	Herb	Annual	June-August
Medicago lupulina L.	Herb	Annual/Perennial	April-July
Medicago minima Grufb.	Herb	Annual	April-June
Medicago polymorpha L.	Herb	Annual	March-June
Mentha longifolia Huds.	Herb	Perennial	June-August
Micromeria biflora Bth.	Subshrub	Perennial	May-July

Milium effusum L.	Grass	Perennial	July-August
Morina longifolia Wall. ex DC.	Herb	Perennial	July-August
Myosotis caespitosa Schultz.	Herb	Perennial	June-July
Myosotis palustris (L.) Nath.	Herb	Perennial	April-May
Myosotis stricta Link ex Roem. & Schult.	Herb	Annual	May-June
Myosotis sylvatica Hoffm.	Herb	Perennial	July-August
Myriactis nepalensis Less.	Herb	Perennial	June-September
Myriactis wallichii Less.	Herb	Annual	September-October
Nasturtium officinale R. Br.	Herb	Biennial/Perennial	May-July
Nepeta cataria L.	Herb	Perennial	May-July
Nepeta laevigata (D.Don) Hand.-Mazz.	Herb	Perennial	July-August
Nepeta raphanorhiza Bth.	Herb	Perennial	May-July
Oenothera glazioviana Micheli	Herb	Annual/Biennial	August-September
Oenothera rosea Ait.	Herb	Perennial	May-August
Ophioglossum vulgatum L.	Fern	Perennial	
Origanum vulgare L.	Herb	Perennial	June-August
Oryzopsis munroi Stapf ex Hook.f.	Grass	Perennial	June-July
Oxalis acetosella L.	Herb	Perennial	June-July
Oxalis corniculata L.	Herb	Perennial	March-October
Oxyria digyna Hill	Herb	Perennial	June-August
Oxytropis lapponica Gay	Herb	Perennial	June-August
Papaver dubium L.	Herb	Annual	April-May
Pedicularis punctata Dcne.	Herb	Perennial	June-August
Pennisetum flaccidum Griseb.	Grass	Perennial	May-July
Pennisetum orientale L. C. Rich.	Grass	Perennial	June-September
Persicaria hydropiper (L.) Delarbre	Herb	Annual	May-September
Persicaria nepalensis (Meisn.) Miyabe	Herb	Annual	June-August
Phalaris arundinacea L.	Grass	Perennial	May-July
Phelum alpinum L.	Grass	Perennial	July-August
Phlomis bracteosa Royle ex Bth.	Herb	Perennial	June-August
Phlomis cashmeriana Royle ex Bth.	Herb	Perennial	June-July
Plantago depressa Willd.	Herb	Perennial	July-August
Plantago himalaica Pilger	Herb	Perennial	July-August
Plantago lanceolata L.	Herb	Perennial	April-August
Plantago major L.	Herb	Perennial	July-September
Poa alpina L.	Grass	Perennial	June-July
Poa angustifolia L.	Grass	Perennial	April-August
Poa annua L.	Grass	Annual	May-September
Poa bulbosa L.	Grass	Annual/Perennial	April-September
Poa pratensis L.	Grass	Perennial	June-August
Poa stewartiana Bor	Grass	Annual	June-July
Polemonium caeruleum L.	Herb	Perennial	June-July
Polygala sibirica L.	Herb	Perennial	April-June
Polygonum affine D. Don	Herb	Perennial	August-September
Polygonum amplexicaule D. Don	Herb	Perennial	July-August
Polygonum aviculare L.	Herb	Annual	April-October
Polygonum plebeium R.Br.	Herb	Annual	May-July
Polygonum rumicifolium Royle ex Bab.	Herb	Perennial	June-July
Polypogon fugax Nees ex Steud.	Grass	Annual	May-July
Potentilla argentea L.	Herb	Perennial	July-September
Potentilla argyrophylla Wall. ex Lehm.	Herb	Perennial	June-August
Potentilla biflora Willd. ex Sch.	Herb	Perennial	June-August

Potentilla doubjouneana Camb.	Herb	Perennial	July-August
Potentilla supina L.	Herb	Annual/Biennial	April-September
Primula denticulata Sm.	Herb	Perennial	May-June
Primula rosea Royle	Herb	Perennial	May-July
Prunella vulgaris L.	Herb	Perennial	July-September
Pseudostellaria himalaica Pax	Herb	Perennial	May-July
Ranunculus arvensis L.	Herb	Annual	March-May
Ranunculus diffusus DC.	Herb	Perennial	May-June
Ranunculus distans Royle	Herb	Perennial	April-May
Ranunculus muricatus L.	Herb	Annual	March-May
Ranunculus palmatifidus H. Riedl	Herb	Perennial	June-July
Ranunculus sceleratus L.	Herb	Annual	May-September
Rorippa islandica (Oeder) Borbás	Herb	Annual/Biennial	April-July
Rosa macrophylla Lindl.	Shrub	Perennial	June-July
Rubia cordifolia L.	Climber	Perennial	June-July
Rumex dentatus L.	Herb	Annual	June-August
Rumex hastatus D. Don	Subshrub	Perennial	July-September
Rumex nepalensis Spreng.	Herb	Perennial	June-August
Saccharum filifolium Steud.	Grass	Perennial	April-May
Salix flabellaris N. J. And.	Subshrub	Perennial	June-July
Salvia hians Royle ex Bth.	Herb	Perennial	July-August
Salvia moorcroftiana Wall. ex Bth.	Herb	Perennial	May-June
Sambucus wightiana Wall.	Subshrub	Perennial	June-July
Saussurea albescens Sch.	Herb	Perennial	July-September
Saussurea heteromalla Hand. Mazz.	Herb	Perennial	June-July
Saxifraga sibirica L.	Herb	Perennial	July-August
Scandix pecten-veneris L.	Herb	Annual	March-May
Scrophularia polyantha Royle ex Bth.	Herb	Perennial	June-August
Scutellaria prostrata Jacq. ex Bth.	Herb	Perennial	August-September
Senecio laetus Edgew.	Herb	Perennial	August-September
Setaria viridis P. Beauv.	Grass	Annual	May-August
Sibbaldia cuneata Kunze	Herb	Perennial	July-August
Silene apetala Willd.	Herb	Annual	April-May
Silene baccifera (L.) Roth	Herb	Perennial	June-July
Silene coronaria (Desr.) Clairv. ex Rchb.	Herb	Perennial	July-August
Silene vulgaris Garcke	Herb	Perennial	June-July
Sisymbrium irio L.	Herb	Annual	May-June
Sisymbrium loeselii L.	Herb	Annual/Biennial	March-October
Sisymbrium officinale (L.) Scop.	Herb	Annual	June-July
Sium latijugum Clarke	Herb	Perennial	July-August
Solanum americanum Mill.	Herb	Annual	September-November
Solanum pseudocapsicum L.	Herb	Perennial	July-September
Solenanthus circinatus Ledeb.	Herb	Perennial	May-June
Solidago virgaurea L.	Herb	Perennial	September-October
Sonchus asper Hill.	Herb	Annual	September-December
Sonchus oleraceus L.	Herb	Annual	February-May
Sorghum bicolor (L.) Moench	Grass	Perennial	June-August
Stachys floccosa Bth.	Herb	Perennial	July-August
Stachys sericea Wall. ex Bth.	Herb	Perennial	June-August
Stellaria media Cyr.	Herb	Annual	April-September
Stellaria uliginosa Murray	Herb	Perennial	July-August
Stipa sibirica Lamk.	Grass	Perennial	July-August

Swertia petiolata D. Don	Herb	Perennial	July-September
Swertia tetragona Clarke	Herb	Annual	August-September
Tagetes minuta L.	Herb	Annual	September-November
Taraxacum officinale Weber	Herb	Perennial	February-December
Thalictrum minus L.	Herb	Perennial	July-August
Thalictrum pedunculatum Edgew.	Herb	Perennial	May-June
Themeda anathera Hack.	Grass	Perennial	June-August
Thlaspi arvense L.	Herb	Annual	April-June
Thlaspi cochlearioides H. & T.	Herb	Perennial	May-July
Thlaspi griffithianum Boiss.	Herb	Perennial	May-June
Thymus serpyllum L.	Herb	Perennial	May-October
Torilis japonica DC.	Herb	Annual	June-August
Tragopogon dubius Scop.	Herb	Perennial	May-August
Tragopogon kashmirianus G. Singh	Herb	Biennial	July-August
Trifolium pratense L.	Herb	Perennial	April-August
Trifolium repens L.	Herb	Perennial	May-September
Trigonella emodi Bth.	Herb	Perennial	June-August
Tulipa clusiana DC.	Herb	Perennial	March-April
Urtica dioica L.	Herb	Perennial	July-August
Valeriana jatamansi Jones	Herb	Perennial	May-July
Valerianella dentata Poll.	Herb	Annual	April-June
Verbascum thapsus L.	Herb	Biennial	June-September
Verbena officinalis L.	Herb	Perennial	July-August
Veronica arvensis L.	Herb	Annual	March-April
Veronica beccabunga L.	Herb	Perennial	May-August
Veronica biloba L.	Herb	Annual	May-September
Veronica persica Poir.	Herb	Annual	February-December
Vicia sativa L.	Climber	Annual	June-August
Viola biflora L.	Herb	Perennial	June-July
Viola canescens Wall. ex Roxb.	Herb	Perennial	April-June
Viola odorata L.	Herb	Perennial	March-May
Xanthium strumarium L.	Herb	Annual	July-August

In terms of growth forms, a large majority, i.e. 242 species are herbs, followed distantly by grasses with 32 species. Other growth forms, such as sedges and subshrubs are represented by 8 species each; climbers and rushes by 4 and 3 species, respectively; shrubs and ferns by 2 species each; and twinner by a single species (Fig. 5.5).

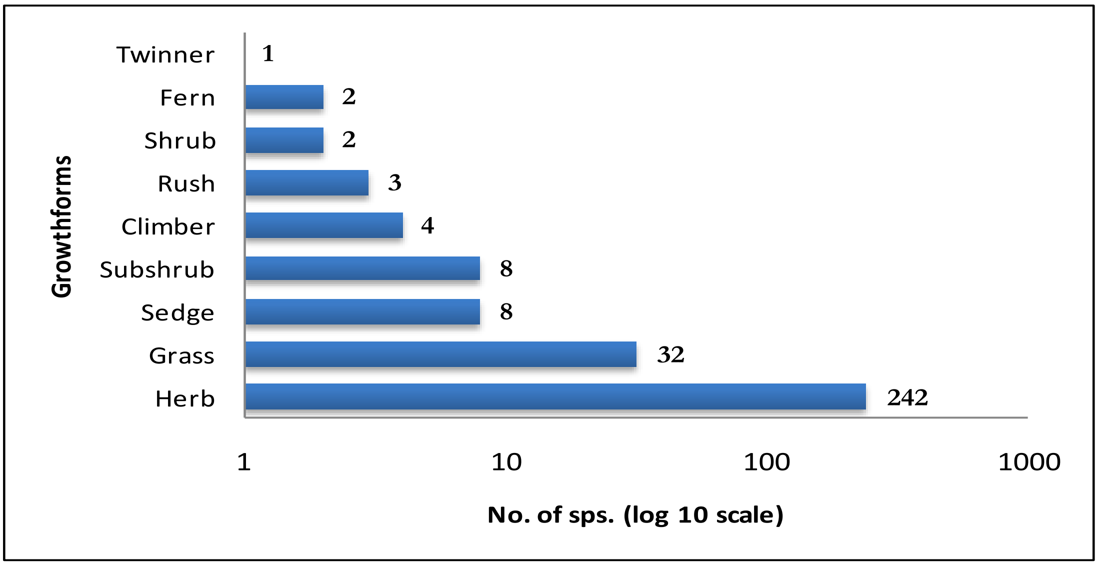

Fig. 5.5 Representation of different growthforms in the species pool of all the grassland sites in Kashmir Himalaya

The life-spans of the total grassland plant species recorded are of various types. Majority of the plant species show perennial life-span, followed by annual and biennial life-spans with 76 and 8 species, respectively. Some of the plant species show more than one type of life-spans: there are 8 species which show both annual and biennial life-spans, 3 species with both annual and perennial life-spans, and a single species with both biennial and perennial life-spans (Fig. 5.6).

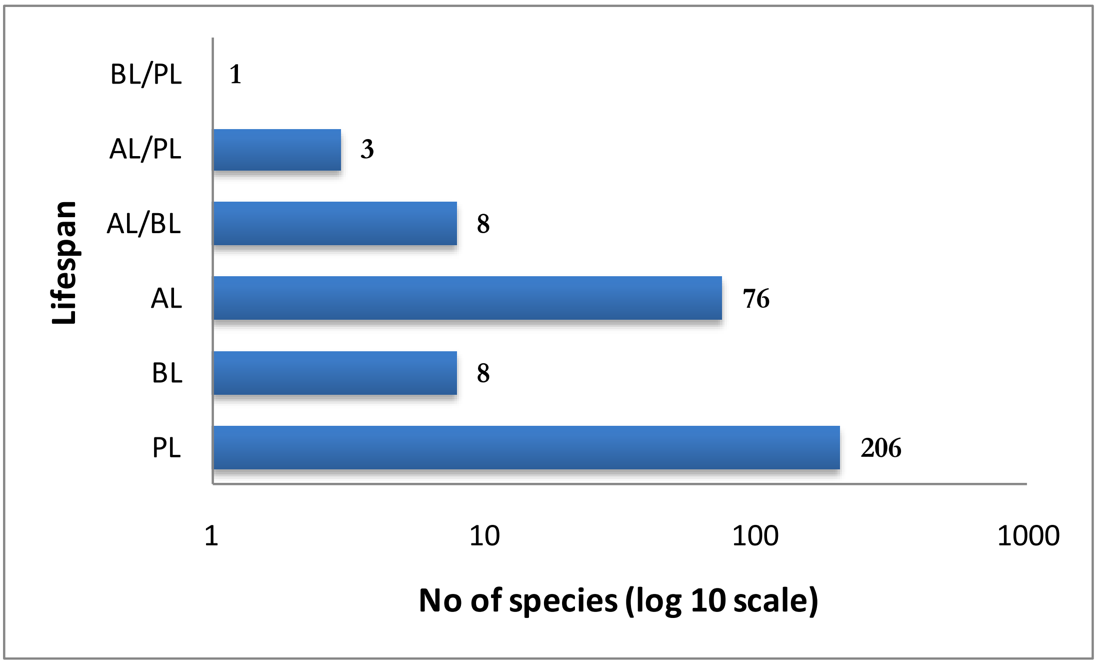

Fig. 5.6 Representation of different forms of lifespan in the species pool of all the grassland sites in Kashmir Himalaya

In terms of the number of months during which a plant species remains in flowering stage, the flowering phenology of the total species pool showed a characteristic hump-shaped pattern with peak reached during the middle of the summer season. Whilest, there is no species in flowering stage during the harsh winter month of January, it gradually starts flowering in subsequent months with 5, 22, 58, 122 and 187 species in flowering stage during the months of February, March, April, May, and June, respectively; and finally reaching a peak during July with 227 species. This is followed by a relatively sharp decline with 167, 65, 16, and 6 species in flowering stage during the months of August, September, October, and November, respectively, which finally reaches to just 3 species during the month of December (Fig. 5.7).

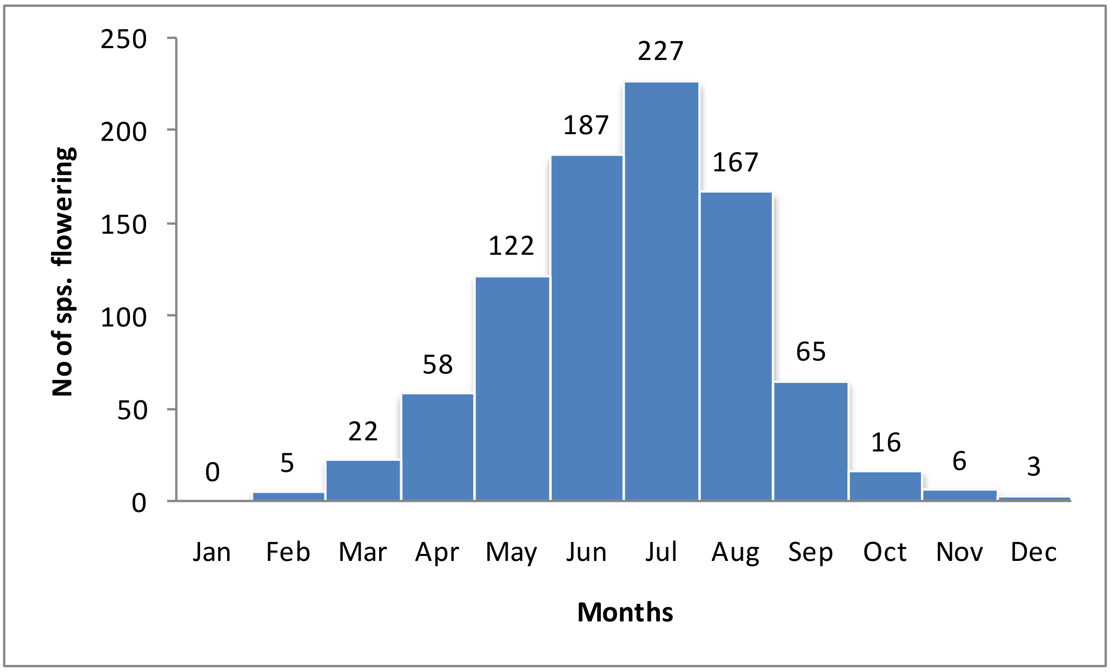

Fig. 5.7 Cumulative monthly flowering calendar of angiosperms (n=300) in the species pool of all the grassland sites in Kashmir Himalaya

The total plant species pool shows a distinct occurrence pattern in the grassland sites (Table 5.3). While as, on one hand, nearly half, i.e. 136 species are restricted to a single grassland site, on the otherhand, 74, 43, 20, 12, and 9 species occur in 2, 3, 4, 5, and 6 grassland sites, respectively. Likewise, 2 species each occur in 7, 8, 10, and 12 grassland sites. No species shows occurrence in 9, 11, 13 and 14 grassland sites. A linear decreasing trend is evident: with more species present in less number of grassland sites, and vice versa (Fig 5.8).

6.1) Floristic Diversity

Taxonomic inventory of grassland flora

A total of 302 plant species has been recorded from all the grassland sites selected in the Kashmir Himalaya in the present study (Table 5.1). These species belong to 199 genera in 52 families (Fig. 5.1). With the exception of two species, *Adiantum capillus-veneris* and *Ophioglossum vulgatum,* which belong to pteridophytes, all other 300 species belong to angiosperms (Table 5.1). Therefore, the grassland flora in Kashmir Himalaya is predominately rich in the angiosperms, as elsewhere in the world.

In the Kashmir Himalayan grassland flora recorded in the present study, family Asteraceae contributed the largest number of 45 species (Fig. 5.2); this is also true for the overall flora of the Kashmir Himalaya (Dar *et al*. 2002). Poaceae, the grass family, was the second largest family with 32 species. The predominance of grasses can be explained by the fact that, even by definition (Singh *et al*. 1983), grasslands are the ecosystems with herbaceous vegetation, predominantly grasses. Shaheen *et al*. (2011) also reported the predominance of Asteraceae, followed by Poaceae in the grassland flora of western Himalaya, Pakistan.

The proportion of 302 plant species across the 52 families was quite un-even. Thus, some of the families were relatively species-rich, while as majority of them contain relatively lesser number of species; on the lower extreme, 19 families were represented by a single species (Fig. 5.3). Similar pattern of un-even distribution of the species across199 genera recorded in the present study was much evident. Several genera, such as *Galium, Poa, Ranunculus, Carex, Polygonum* and *Potentilla* contribute 5 or more species each to the grassland flora (Table 5.1); these genera are known to have much representation in the temperate regions of Northern hemisphere. Nearly two-third (i.e. 142) of the genera recorded in the grassland sites were represented by a single species each; while the remaining genera contribute 2 to 6 species each (Fig. 5.4). Such a taxonomic dispersion of species amongst their respective genera and families makes the flora of grasslands of the Kashmir Himalaya taxonomically more diverse. Such a taxonomically diverse grassland flora stores a rich repository of evolutionary history.

Growth form patterns

The grassland vegetation is most often dominated by the herbaceous elements, with some occasional woody/shrubby growth-forms. It is equally true for the Kashmir Himalayan grasslands, because majority of the species (292 spp.) recorded during the present study were herbs, the remaining 10 species being woody sub-shrubs and shrubs (Table 5.2, Fig. 5.5). The herbs, in addition to forbs,included 43 species of grasses, sedges and rushes, 5 species of climbers and twinners, and 2 species of ferns. The occurrence of diverse growth forms in the Kashmir Himalayan grasslands is an indirect measure of their higher functional diversity. This is significant because some of the recent studies have shown that higher functional diversity can enhance the capacity of the natural ecosystems to adapt and mitigate the impacts of changing climate (Hillebrand & Matthiessen, 2009).

Pertinently, the predominance of such types of plant growth forms in the grassland ecosystems was also reported from the Garhwal Himalaya (Sundriyal, 1992; Nautiyal *et al*., 2001), as well as from the Valley of Flowers National Park, Uttaranchal (Singh *et al*. 2005).

Life span patterns

Plants generally possess different life span strategies which, in turn, strongly shape the spatio-temporal patterns of biodiversity in the natural ecosystems. As commonly observed in the temperate grasslands, the majority (206) of species in the Kashmir Himalayan grasslands were perennials, followed by annuals (76 spp.), and biennials (8 spp.) (Fig. 5.6). In many species, more than one type of life span strategy is operative. For example, some of the plant species recorded during the present study, such as *Rorippa islandica, Sisymbrium loeselii,* etc., showed both annual and biennial life spans. Whereas the plant species with perennial life span were predominant in the grassland sites at higher altitudes, such as Draphoma, Dugwan, Baltal, and Thajwas; those with annual life span were relatively common in the low altitude grassland sites, such as Botanic Garden and Sopore (Table 5.2).

Flowering phenology

The timing and duration of flowering in the constituent species pool of a plant community is an explicit indicator of the ecosystem structure, function and spatio-temporal dynamics. With well-marked four seasons, winter, spring, summer and autumn, in the Kashmir Himalaya, the flowering phenology of the grasslands flora was strongly cued to the changing seasons. Out of the total species pool of 300 angiosperms, the maximum number of 227 species were in flowering during the peak summer month of July. Because of harsh winter, none of the species showed flowering during the month of January. In short, the flowering phenology showed a typical uni-modal distribution trend, with peak during middle of the summer season (Fig. 5.7).

Similar results were reported in grasslands of the Central Himalaya **(**Negi *et al* 1992), and those of the Garhwal Himalaya (Nautiyal *et al*., 2001).

Species richness (SR)

Species richness - the simplest measure of biodiversity – refers to the number of species occurring in a sampling unit/site/habitat/region. In the present study, species richness was computed on the basis of 3 measures: SR actual, SR observed and SR estimated.

Based on the occurrence data, the value of SR actual for the species pool of all the grassland sites was 302 species (Table 5.3). However, occurrence of these species in different grassland sites showed a characteristic pattern, with more number of species present in less number of grassland sites, and vice versa. On the one extreme, there were 136 species (out of 302 species) which occur in a single grassland site; on other extreme, there were just 2 species which occur in 12 sites (out of 14) (Fig. 5.8). Such an occurrence pattern hints towards an appreciably higher species turnover across the grassland sites.

Based on the species accumulation curve, the value of SR observed was also 302 species (Table 5.4). However, the curve showed a steep linear increase with an increasing number of grassland sites (Fig. 5.9). It clearly suggests that the Kashmir Himalayan grasslands are not only species rich, but also unique in terms of species composition across the region.

Undoubtedly, the species accumulation curve provides the empirical evidence that species richness in the Kashmir Himalayan grasslands is likely much beyond the SR observed value of 302 species. The likely higher number of species richness was well-substantiated by the values of SR estimated, which range from the lowest value of 360.42 for Bootstrap to the highest value of 499.6 for Michaelis-Menton SR estimators (Table 5.5, Fig. 5.10). In other words, it can be predicted that, in the Kashmir Himalaya, increasing the number of grassland sites could increase the value of SR observed from 50 to 200 species.

Shared species and similarity indices

The number of species shared between the different sites in a region is a quantitative indicator of biotic homogeneity or heterogeneity. Based on the occurrence data of species, the average value of shared species computed for all grassland sites in the Kashmir Himalaya was 9.87. Likewise, the values of Jaccard and Sorenson indices of similarity were 0.12 and 0.2, respectively (Table 5.6, Fig. 5.11). Such low values of shared species and similarity indices point towards the biotic heterogeneity of the Kashmir Himalayan grasslands.

Beta-diversity in grassland flora

Beta-diversity is a measure of the extent to which the diversity of two or more samples/sites differs. In other words, it reflects the biotic change or the species replacement (Magurran, 2004). Based on the Whittaker's measure (βw), whose values range from 0-1 (Whittaker, 1960), the lowest value of 0.5 was obtained between Sopore and Pahalgam grassland sites (Table 5.7). Although the two sites are located at different altitudes and far way from each other, the species composition in both was dominated by the common and widespread ruderal species, e.g. *Stellaria media, Capsella bursa-pastoris, Cynodon dactylon*, *Veronica persica*, etc. (Table 5.3). On the other extreme, the highest βw value of 0.976 was obtained between Thajwas and Dignibal. Whereas the former is an alpine site, the latter is on the tableland hillock near plains; thus, there is almost complete species replacement between these two sites. Such a high value of beta-diversity is more specifically because of the characteristic species composition in the former site, several of its species being endemic to the region, e.g. *Lagotis cashmeriana* and *Potentilla doubjouneana* (Table 5.3).

During the present study, beta-diversity between more than two grassland sites, i.e. between all the 14 sites, was computed on the basis of Harrison measure (β_{H1}), whose values range from 0-100. The value of β_{H1} obtained between all these sites was 40.07. Such a higher value of β_{H1} reflects appreciably higher biotic change in the Kashmir Himalayan grasslands

Floristic diversity

Historically, research studies to investigate the floristic diversity in grasslands have used sampling data on monthly as well as peak-season basis. The monthly data method is always preferable, as the inter-monthly changes are captured as well. However, it is time-consuming, laborious and doesn't allow larger number communities to be studied in shorter period of time. That is why the peak-season data method is an alternative and, to a large extent, useful tool to compare a large set of communities in a short time in temperate climates. Whereas the monthly sampling in selected grassland sites helps in determining the **magnitude** of floristic diversity, the peak-season sampling gives an idea about the **nature** of floristic diversity. Therefore, during the present study, data was obtained from the selected grassland sites using both the methods.

Monthly sites

In the Kashmir Himalaya, altitude is the major factor that greatly influences the structure and species composition of vegetation types. Based on this underlying factor, the monthly data obtained from 3 selected grassland sites - Botanic Garden, Draphoma and Dugwan located at different altitudes (Table 3.1), showed changing patterns of floristic diversity along elevation gradient in the study area (Fig. 3.1).

In terms of monthly occurrence of the number of species, the total species pool in each of these sites showed a uni-modal distribution pattern, with peak number of species during the middle of the summer season (Table 5.12, Fig 5.25). However, the peak value in Botanic Garden was recorded in the month of June, while these values in Draphoma and Dugwan were recorded in July. Such a monthly difference in the peak values of species occurrence may be attributed to the location of Botanic Garden at lower altitude in the plains of valley, and thus earlier start of the growing season in this grassland site as compared to other two sites, which are located at much higher altitudes along the elevation gradient in the study area. Such a differential start of growing season was also reported from meadows of the Central Himalaya (Rikhari *et al.* 1992).

Both theoretical and empirical studies have shown that there is generally a decrease in biodiversity as we move along an elevation gradient (Rahbek, 1995). The same decreasing trend was evident in the floristic diversity in grasslands of the Kashmir Himalaya. Thus, the Botanic Garden recorded the highest species richness of 86 spp., which decreased to 75 spp. at Draphoma and to 62 spp. at Dugwan along an increasing elevation gradient (Fig. 5.26). A similar decreasing trend was observed in terms of Shannon's species diversity index (Fig. 5.27). Similar trend of decreasing species diversity with increasing altitude has been reported in the grasslands of Uttarakhand, Western Himalaya (Kala *et al*. 2002).

The number of species shared between different months of the calendar year provides an indirect measure of the monthly species turnover (i.e. change in species composition) within a biotic community (Vellend, 2001). In other words, lesser the value of species shared between different months, more is the value of species turnover on monthly basis. During the present study, the lowest value of shared species was obtained for Draphoma grassland site, followed by Dugwan and the Botanic Garden; and similar trend was observed for the Sorensen's similarity index. However, exactly the same values of Jaccard's index were recorded for Draphoma and Dugwan, followed by a higher value in the Botanic Garden (Table 5.13, Fig. 5.28). Such a trend indicates a higher number of species turnover during different months of the growing season in grasslands at higher altitudes than those at lower altitudes. Therefore, it can be deduced that as against species richness, species turnover shows an increasing trend with increase in altitude in the Kashmir Himalayan grasslands. A plausible explanation for this may be the shorter duration of growing season at higher altitudes.

Similar results were also obtained by Bhattarai *et al*. (2004), who reported that the species turnover was much higher in high-altitude grassland sites of the Central Himalayas, Nepal.

Peak-season sites

In the present study, 9 grassland sites were selected from which data was obtained on peak-season basis (Table 3.1, Fig. 3.1). Among the peak-season sites, the values of SR actual ranged from the highest of 54 species recorded in Gualta to the lowest of 27 species in Pahalgam (Fig. 5.66); similar pattern was also observed in terms of Shannon's diversity index and species evenness (Table 5.41, Figs. 5.68 & 5.69). Although these sites are located at different altitudes along the elevation gradient in the region, yet no distinct trend was observed in terms of the different measures of floristic diversity. The highest values of SR actual, species diversity and species evenness, except SR observed, recorded in Gualta, Uri may be attributed to the relatively lower altitude (1500 m) of this grassland site than the rest of the main Valley; this consequently brings the influence of the sub-tropical climate in the site. In case of SR observed, Gulmarg (and not Gualta) recorded the highest value (Fig. 5.67); whether it was due to higher species turnover at high altitude, or some sampling artefact, needs further investigation. On the other extreme, Pahalgam consistently recorded the lowest values in terms of all the measures of floristic diversity: SR actual, SR observed, species diversity and species evenness. Most likely, local factors, such as human disturbances, dominance of ruderal weedy species, etc. may be responsible for such low values of floristic diversity at Pahalgam.

On an average, species diversity of 4.2 (monthly sites) to 2.3 (peak-season sites) was recorded in the Kashmir Himalayan grasslands (Figs. 5.27 & 5.68). Such a range of values of species diversity recorded during the present study is comparable to that reported in similar studies in some other regions of the Himalaya: 1.53-2.88 in the Western Himalaya (Samant *et al*. 1998), 2.39-4.63 in the Gharwal Himalaya (Nautiyal & Guar, 1999), 2.5-3.10 in the trans-Himalayan alpines of Nepal (Panthi *et al*. 2007), 3.13 in the western Himalaya, Pakistan (Shaheen *et al*. 2011).

The species shared between the sampling units gives insights about the local-scale species turnover within a grassland site. In other words, higher the value of species shared between the sampling units (30 quadrats were laid to record abundance data in each of the peak-season site), expectedly lower will be the value of SR observed and, to a large extent, species diversity. Based on this rationale, it would be expected to obtain the highest and the lowest values of shared species (or similarity indices) in Pahalgam and Gualta, respectively. Expectedly, the lowest values of 4 similarity indices - Jaccard, Sorensen, Morisita-Horn and Bray-Curtis – were obtained in Gualta; however the highest values were recorded only for Bray-Curtis in Pahalgam, and the same for Jaccard and Sorensen in Sopore, and for Morisita-Horn in Baltal (Table 5.42, Fig. 5.71). Also, unexpectedly, the highest and the lowest values of shared species were obtained for Sopore and Kamalkote, respectively (Fig. 5.70). Whether such a departure from the expected values is due to statistical artifact, or the values of shared species and similarity indices partially determine the expected values of SR observed and species diversity, merits further investigation.

6.2) Biomass and primary productivity in the Kashmir Himalayan grasslands

Biomass in the Kashmir Himalayan grasslands

During the present study, biomass in the producer sub-system of grassland sites was partitioned into four compartments: above-ground live, standing dead, litter biomass, and below-ground. Within the monthly data grassland sites, the the growing season started during early spring in the Botanic Garden, and remained consistent for longer period of time during summer, and then declined gradually; however, it started during late spring in Draphoma and early summer in Dugwan, followed by abrupt increase and then sharp decline during subsequent months. The production of the above-ground biomass in the monthly sites showed a gradual increase from the months of spring season, reached to its peak during summer and started declining during autumn, and finally remained at very low levels, or absent, during winter (Fig. 5.78a). Expectedly, the standing dead biomass was more pronounced during the late autumn and early winter months (Fig. 5.78b). Interestingly, the litter biomass showed the peak values twice or thrice (Fig. 5.78c). The below-ground biomass was more pronounced during early winter months in the Botanic Garden, but during late autumn in the Draphoma and Dugwan sites (Fig. 5.78d).

Within the 9 peak-season data sites, the highest value of peak above-ground live biomass was recorded in the Pahlipora and that of the lowest value in Thajwas grassland. Similarly, the highest value of peak standing dead biomass was recorded in Gualta Uri, again the lowest value in Thajwas; and those of peak litter biomass was highest in Pahlipora, but the lowest value in Kamalkote Uri grassland. Likewise, the highest value of peak below-ground biomass was expectedly recorded in Thajwas grassland, but the lowest value in Sopore one (Table 5.50, Fig. 5.80).

Altitude, being the primary factor in the Kashmir Himalaya, plays a crucial role in determining the spatial and temporal dynamics of temperature and precipitation regimes, along the elevation gradient. For instance, relatively lower temperature and more precipitation available in the form of snow for major part of the year, except 3-4 months, at high-altitudes make the growing season of alpine vegetation shorter and faster. Because of this reason, during the present study, the biomass in the monthly alpine site of Dugwan was estimated only for a shorter duration of 3 months - June, July and August. As the growth season is completed very fast, in short duration of time, at higher altitudes, the peak values of biomass were reached abruptly, and the decline was also too sharp in Draphoma and Dugwan grasslands, as compared to that of the Botanic Garden (Fig. 5.78a). Similarly, within the peak-season data sites, the alpine site of Thajwas recorded the highest value of belowground biomass (Fig. 5.80). Therefore, the preceding biomass patterns recorded in both monthly- and peak-season data grassland sites clearly highlight the principal role of altitude in determining the biomass production/accumulation in different compartments of the producer sub-system.

Net primary productivity in the Kashmir Himalayan grasslands

Net primary productivity (NPP) is the fundamental process in ecosystem functioning. It is defined as the net accumulation of dry matter by green plants per unit time and space, and is often expressed on an annual basis (Nayak *et al.* 2013). For a given period of measurement, NPP is equal to the change in both the aboveground and belowground plant biomass, plus any losses over this period due to death and subsequent decomposition and herbivory (Long *et al*., 1989; Roberts *et al*., 1993). NPP provides the energy and the matter that drives most of the biotic processes on Earth. Long-term sustenance of NPP over space can also contribute to enrich our planet both environmentally and economically.

During the present study, the annual net primary productivity (NPP) in the monthly data grassland sites has been calculated based upon the calendar year (March-February). The length of the growing season varied widely from as little as 3 months in Dugwan, 7 months in Draphoma, and up to 9 months in the Botanic Garden grassland. Along the elevation gradient in the 3 monthly sites, the highest value of total annual net primary productivity (TNP) was recorded in the mid-elevation grassland site of Draphoma with 9161.19 $gm^{-2}y^{-1}$, followed by Dugwan and Botanic Garden sites, with 7533.6 and 3408.21 $gm^{-2}y^{-1}$, respectively (Table 5.49, Fig. 5.79a). Even on considering the values of ANP and BNP separately in the monthly data sites, the same pattern is clearly evident, i.e. Draphoma has the highest values of ANP and BNP, as compared to Dugwan and Botanic Garden sites. From this, it can be deduced that a mid-domain effect may be operative in case of NPP in the Kashmir Himalayan grasslands along an elevation gradient. However, such a mid-domain pattern was absent in the floristic diversity of monthly data sites, where a decreasing trend was observed from the lower to higher altitudes (Fig. 5.26 & 5.27).

In the peak-season data sites, the sum of peak standing crop (mostly during the months of July-August) from different aboveground producer sub-systems (aboveground live, standing dead, litter) and belowground compartments of the grassland sites was used to estimate the peak above-ground net primary productivity (ANPp) and peak below-ground net primary productivity (BNPp), respectively. The highest value of 778.4 g m^{-2} ANPp was recorded at the mid-elevation site at Pahlipora and the lowest of 85.4 g m^{-2} at the high-elevation site of Thajwas (Table 5.51, Fig. 5.81a). Likewise, the highest value of 1075.4 g m^{-2} BNPp was expectedly recorded at Thajwas, but the lowest of 199.2 gm^{-2} at Sopore site (and not at Pahlipora). As the sum of ANPp and BNPp, the peak total net primary productivity (TNPp) was the highest at Pahlipora and the lowest at Sopore site. Furthermore, in percentage terms, the contribution of ANPp to the TNPp was more than 50% in Sopore, Gualta and Pahlipora grasslands, but it same was less than 30% in the rest of the peak-season data sites (Fig. 5.81b).

Not only from the ecosystem functioning perspective, but also from the livestock point of view, understanding the temporal and spatial distribution of TNP and its partitioning among the ANP and BNP compartments assumes critical importance in the sustainable management of grasslands. Although, in terms of absolute value, Draphoma grassland had the highest values of ANP, BNP and TNP, yet in percentage terms, ANP had relatively more contribution to the TNP in Botanic Garden grassland, as compared to those of Draphoma and Dugwan ones (Fig. 5.79b). In other words, more proportion of TNP was available in the form of ANP in the low-elevation grassland site of Botanic Garden than those of the mid- and high-elevation sites of Draphoma and Dugwan. Also, the temporal availability of ANP was more evenly distributed across different months of the calendar year in Botanic Garden grassland, as compared to restricted distribution in those of Draphoma and Dugwan sites (Figs. 5.73b, 5.75b & 5.77b). In the latter two sites, the TNP was available more in the form of BNP during major part of the calendar year. Thus, even if these sites had higher values of TNP as compared to that of the Botanic Garden site, but it was available in the form of ANP in lesser quantity and mostly as the BNP. Similar patterns were obviously evident in the peak-season data sites; while as the mid-elevation site at Pahlipora had the highest value of ANPp and the high-elevation site at Thajwas had the highest value of BNPp (Fig. 5.81b).

Namgail *et al* (2012) also reported similar hump-shaped pattern in the aboveground phytomass of the grasslands along an altitudinal gradient in Trans-Himalaya. Earlier, Shankar *et al*. (1993) have reported a similar increase in the mean belowground phytomass of grasslands in the Eastern Himalaya.

6.3) Relationship between floristic diversity and primary productivity

In the recent past, the relationship between biodiversity and ecosystem functioning (BEF) has emerged as a dominant issue in the biodiversity research (Chapin *et al*., 1998; Naeem *et al*. 1999; Loreau *et al*., 2001; Sutherland *et al*., 2013). It is expected that the research insights gained from this field will allow biologists make expert predictions regarding the consequences of species extinction on the ecosystem goods and services, on which ultimately depends the human existence. In general, the relationship between floristic diversity and productivity has been shown to be scale-dependent, with the form of the scale-dependence being variable across different studies. Whereas most of the studies reported no pattern at all, unimodal relationships are as common as positive relationships at the within-community scale. However, at the across-community scale, unimodal patterns are more than three times prevalent than the positive relationships. At the continental- to global-scale, the pattern is dominated by positive relationships (Adler *et al*. 2011).

In BEF research studies, it is extremely important to explicitly investigate the relationship between the specific measure(s) of biodiversity and the ecosystem functioning. During the present study, relationship has been investigated between different measures of floristic diversity (SR actual, SR observed, Shannon’s species diversity and species evenness) and the measures of net primary productivity (ANP, BNP, TNP). In this relationship, the floristic diversity was considered as an independent variable and the net primary productivity as a dependent variable. In other words, the investigation aimed at understanding the influence of the change (decrease or increase) in floristic diversity on the magnitude of net primary productivity in the Kashmir Himalayan grasslands.

In the present study, the relationship was tested in the monthly data grassland sites between the SR actual on one hand and all the 3 measures of net primary productivity - ANP, BNP and TNP - on the other hand (Tables 5.52, 5.53 & 5.54). In the Botanic Garden site, no relationship was found between SR actual and ANP (Fig. 5.82a). However, a weak negative relationship was found between SR actual and both BNP and TNP at this site - with increase in SR actual, there was decline in the BNP and TNP (Figs. 5.82b & 5.82c). In contrast to Botanic Garden site, a positive relationship between SR actual and ANP was observed in Draphoma grassland - with increase in SR actual, there was increase in the ANP (Fig. 5.83a). As in the Botanic Garden grassland, a weak negative relationship between SR actual and BNP was observed in Draphoma grassland (Fig. 5.83b); however, no relationship was found between SR actual and TNP in this monthly data site (Fig. 5.83c). In the Dugwan grassland, a positive relationship was observed between SR actual and the ANP, BNP and TNP (Figs. 5.84a, b & c).

In the peak-season grassland sites, the relationship was tested between the various measures of floristic diversity - SR actual, SR observed, Shannon’s species diversity and species evenness - on one hand, and all the 3 measures of peak net primary productivity - ANPp, BNPp and TNPp - on the other hand. A positive linear relationship was observed between all the measures of floristic diversity and ANPp, except for SR observed and ANPp, where no relationship was found at all (Figs. 5.85a, b, c & d). In contrast, no relationship was observed between SR actual and BNPp (Fig. 5.86a), a positive relationship between SR observed and BNPp (Fig. 5.86b), while a negative relationship was observed between Shannon’s species diversity, species evenness and BNPp (Fig. 5.86c & d). Whereas a positive relationship was observed between the SR actual, SR observed and TNPp (Fig. 5.87a & b), there was no relationship between Shannon’s species diversity and TNPp (Fig. 5.87c), and a negative relationship between the species evenness and TNPp (Fig. 5.87d).

Singh *et al.* (2005), while carrying out a study in the grasslands of Uttaranchal, Indian Himalaya, reported similar results at least for the ANPp. They concluded that the above-ground net primary productivity computed on the basis of peak-season plant biomass was significantly correlated with species diversity and species richness.

6.4) Impacts of livestock grazing on floristic diversity and primary productivity

During the present study, impact of livestock grazing on floristic diversity and on primary productivity in the Kashmir Himalayan grasslands was investigated. Grazing, in traditional sense of the term, refers to the consumption of plant parts by animals (i.e. herbivores) without causing the death of an individual plant. The most common example is the consumption of plant species as fodder by domesticated livestock in the world's grasslands. In spite of the benefits, the negative impacts of livestock grazing on global biodiversity have become a serious environmental concern during the recent past. In many regions of the world, excessive grazing in the grassland ecosystems has decreased the density and biomass of many plant species. World over, this has led to the reduction of biodiversity, the alteration of ecosystem structure and functioning and, in particular, to the diminishing productivity in grasslands.

Impacts of livestock grazing on floristic diversity

Experimental plots were set up both in the monthly- and peak-season sites to investigate the impacts of livestock grazing on floristic diversity (Table 3.1). Among the monthly sites, whereas the Dignibal-ungrazed site recorded the species richness value of 50 spp., it declined to nearly its half, i.e. 27 spp., in the Dignibal-grazed site (Table 5.55, Fig. 5.88). Such a drastic reduction in the number of species was the obvious manifestation of excessive livestock grazing. In terms of the peak value of species richness, whereas 36 species were recorded during the month of July in the Dignibal-ungrazed site, the same was reduced to its half, i.e. 18 spp., in the Dignibal-grazed site, but during the month of June (Fig. 5.88). This means that the livestock grazing, in addition to reducing the number of species, can also shift the monthly occurrence patterns of floristic diversity in the grasslands. Besides the simple measure of species richness, different values of the Shannon's species diversity index were recorded for the two sites: whereas its value was

3.81 in Dignibal-ungrazed site, it was reduced to 3.22 in Dignibal-grazed site (Fig. 5.91). As species diversity captures both species richness and species evenness components, therefore it seems clear that the lower value of species diversity in the Dignibal-grazed site possibly shows the impact of livestock grazing on the species evenness component as well.

Kala *et al*. (2002) reported similar results while working on the grasslands in the Valley of Flowers and Great Himalayan National Parks, Uttrakhand, Western Himalaya. In this study, both, the species richness and species diversity, showed drastic decline in the grazing sites as compared to the protected sites. However, Agrawal (1990) reported contrary results, with higher species richness in grazed sites as compared to the protected ones, in the temperate grasslands of the Western Himalaya.

It has been already emphasized that the shared species values and similarity indices can provide insights about the local-scale species turnover, which ultimately determines the beta diversity. The average value of shared species was relatively lower in Dignibal-grazed site as compared to Dignibal-ungrazed site. In contrast, the values of Jaccard's and Sorenson's similarity indices were relatively higher in Dignibal-grazed site, as compared to Dignibal-ungrazed site (Fig. 5.92). The difference in the the average value of shared species between Dignibal-grazed and Dignibal-ungrazed sites was greater as compared to differences in values of Jaccard's and Sorenson's similarity indices. Therefore, based on the wide difference in the value of shared species, it can be deduced that the local-scale species turnover rate is relatively more in the Dignibal-grazed site as compared to Dignibal-ungrazed site. It may be ascribed to the intense selection pressure from grazing that drives the local-scale species turnover.

As true for the monthly sites, contrasting patterns of floristic diversity were observed in the peak-season sites too. To investigate the impacts of livestock grazing on floristic diversity, three peak-season sites were selected at Aru: Aru-long protected, Aru-5-year protected, and Aru-grazed. The highest value of SR actual, with 59 species, was recorded at Aru-long protected site; it was closely followed by Aru-5-year protected site, with 58 species; but drastically reduced to nearly one-fourth at Aru-grazed site, with just 14 species (Table 5.58, Fig. 5.93). Similar decreasing trend was observed in terms of Shannon's species diversity index and species evenness values. However, in terms of SR observed, it was the Aru-5-year protected site which recorded the highest of 24 species; this was closely followed by Aru-long protected site, with 21 species; but reduced to nearly one-third at Aru-grazed site, with 9 species only (Fig. 5.94). Such contrasting values obtained in terms of SR actual, SR observed, species diversity and species evenness, between the protected and grazed sites clearly underline the impact of livestock grazing on floristic diversity.

Likewise, in the monthly sites at Dignibal, the value of shared species was relatively lower in Aru-grazed site as compared to that in Aru-long protected- and Aru-5-year protected site. Contrary to this, the values of Jaccard, Sorenson, Morisita-Horn and Bray-Curtis similarity indices were relatively higher in Aru-grazed site as compared to that in Aru-long protected - and Aru-5-year protected sites (Table 5.59, Fig. 5.95). Clearly, the difference between the values of shared species and similarity indices was relatively less between the Aru-long protected and Aru-5-year protected sites, but much higher between these two protected sites and the Aru-grazed site.

Impacts of livestock grazing on primary productivity

In the preceding sections, it was clearly evident that the excessive livestock grazing drastically reduces the floristic diversity in the Kashmir Himalayan grasslands. Likewise, the livestock grazing also impacts the potential primary productivity in the monthly- as well as peak-season grassland sites. In the monthly sites, whereas the annual ANP value of 1874.96 $gm^{-2}y^{-1}$ was recorded at Dignibal-ungrazed site, the same value for Dignibal-grazed site was 557.04 $gm^{-2}y^{-1}$ (Table 5.64, Fig. 5.100). Thus, grazing results in the reduction of nearly two-thirds of the potential ANP in the Dignibal-grazed site. On the other hand, the annual BNP value of 2016.25 $gm^{-2}y^{-1}$ was recorded at Dignibal-ungrazed site, and the same for Dignibal-grazed site was 2008 $gm^{-2}y^{-1}$ (Table 5.65, Fig. 5.101). Nearly the same values of BNP in the two sites show that the grazing had no impact at all on the annual BNP in the Dignibal-grazed site. On comparing the recorded annual TNP value, it was 3891.21 $gm^{-2}y^{-1}$ in Dignibal-ungrazed site, but is reduced to 2565.04 $gm^{-2}y^{-1}$ in the Dignibal-grazed site (Table 5.66, Fig. 5.102). Therefore, on the basis of TNP, grazing still leads to a loss of one-third of the potential ANP in the Dignibal-grazed site. Furthermore, in terms of percentage, the contribution of ANP to TNP was *ca.* 50% in Dignibal-ungrazed site, while it was *ca.* 20% in the Dignibal-grazed site (Figs. 103a & b).

Similar results were obtained in the grasslands of Almora, Uttarakhand, wherein significantly higher value of the aboveground primary productivity was found in the control plots as compared to the grazed ones (Prakash & Paliwal, 2012). Sundriyal (1992) reported an increase in the above-ground productivity, and decrease in the below-ground productivity, from the grazed grassland sites in Garhwal Himalaya. Contrary to this, during the present study, there was a substantial decrease in above-ground productivity, and no increase at all in the below-ground productivity, at the Dignibal-grazed site in the Kashmir Himalaya.

Patterns similar to the impacts of livestock grazing on primary productivity observed at monthly sites in Dignibal, emerged on comparison of peak-season long-protected, 5-year protected, and grazed sites in Aru (Table 5.68, Fig. 5.105). The values of ANPp recorded at the long-protected, 5-year protected, and grazed sites in Aru were 302, 216.6 and 101.2 g m^{-2}, respectively. This means that the long-protected site had the highest ANPp, which was reduced to its one-third in the grazed site. As expected, the BNPp showed an opposite trend, with values of 669.8, 926.8 and 1228.4 gm^{-2} recorded at long-protected, 5-year protected and grazed sites in Aru, respectively. In this case, it is the selection pressure of livestock grazing that has most likely shifted the primary productivity towards below-ground compartment. Interestingly, on the other hand, the values of TNPp recorded were 971.8, 1143.4, and 1329.6 g m^{-2} at long-protected, 5-year protected and grazed sites in Aru, respectively (Table 5.68). However, in percentage terms, the contribution of ANPp to the TNPp was *ca.* 30% in the long-protected site; but it was reduced to *ca.* 20% and 10% in the 5-year protected and grazed sites, respectively (Fig. 5.106). This means that even if the value of TNP is the highest in the grazed site, yet only 10% of it is available in the form of ANP.

The Kashmir Himalaya, located in the north-western folds of the Himalayan biodiversity hotspot, has vast area of its total land (*ca*. 16%) under grasslands. These grasslands play an important role in providing the precious economic goods and life-supporting ecosystem services. They serve as the bedrock for sustaining the core economic activity of livestock rearing in the region. Apart from sustaining this pivotal economic activity, the grasslands harbour a rich and endemic biodiversity, and regulate the regional carbon, nutrient and hydrological cycles.

In spite of these indispensable ecological and economic roles of grasslands in the region, very few scientific studies have investigated the structural and functional aspects of these ecosystems. As a result of this, paucity of scientific knowledge base on grasslands has been the first and formidable impediment in the formulation of practicable conservation strategies and management plans for these ecosystems. Thus, to fill these knowledge gaps, the present study was undertaken on the Kashmir Himalayan grasslands with the aim to: i) study their floristic diversity, ii) estimate their net primary productivity, iii) investigate the relationship between their floristic diversity and net primary productivity, and iv) determine the impacts of livestock grazing on their floristic diversity and net primary productivity.

Floristic diversity

During the present study, 302 species, belonging to 199 genera in 52 families, were recorded from all the sampled grassland sites in the Kashmir Himalaya. In terms of growth-form diversity, the majority of these species were herbs (232 species), followed by 43 species of grasses, sedges and rushes, 10 species of sub-shrubs and shrubs, 5 species of climbers and twinners, and 2 species of ferns.. Lifespan-wise, the majority of these species (i.e. 206 spp.) were perennials, followed by annuals (76 spp.), and biennials (8 spp.). Of the total species pool, the highest number (227 spp.) exhibited flowering during the month of

July, the lowest number (3 spp.) during December, while none of the species flowered during January. The beta-diversity across all the grassland sites studied here, computed on the basis of Harrison measure (β_{H1}), was 40.07 (values range from 0-100).

In the three monthly sampled sites, the Botanic Garden site (alt. 1600 m) recorded the highest species richness (86 spp.), which showed a decreasing trend of 75 spp. and 62 spp. in Draphoma (alt. 1800 m) and Dugwan (alt. 3000 m), respectively, while moving along an increasing altitudinal gradient. A similar decreasing trend was observed in terms of Shannon's diversity index, with values of 4.31 in Botanic Garden site, 4.3 in Draphoma site, and 4.06 in Dugwan site; though with very little difference between the first two sites. In the nine peak-season sampled sites, the species richness (actual) ranged from the highest value of 54 spp. recorded in Gualta site, to the lowest value of 27 spp. in Pahalgam site. A similar pattern was observed in terms of Shannon's diversity index and species evenness; whereas in case of the former, it ranged from the highest value of 2.79 in Gualta site to the lowest value of 1.59 in Pahalgam site; in case of the latter, the values ranged from 0.857 in Gualta site to 0.620 in Pahalgam site. However, in terms of species richness (observed), although Pahalgam site again recorded the lowest of 13 species, it is the Gulmarg site which recorded the highest of 36 species.

Net primary productivity

Amongst the three monthly sampled sites, the highest value of total annual net primary productivity (TNP) was recorded in Draphoma site with 9161.19 $gm^{-2}y^{-1}$, followed by Dugwan and Botanic Garden sites, with 7533.6 and 3408.21 $gm^{-2}y^{-1}$, respectively. However, in terms of percentage, the above-ground net primary productivity (ANP) had relatively less contribution to the TNP in Draphoma and Dugwan sites, as compared to Botanic Garden site. In the peak-season data sites, the value of the peak net primary productivity (TNPp) ranged from the highest of 1237 gm^{-2} observed in the Pahlipora site to the lowest of 545.8 gm^{-2} in Sopore site. However, in terms of percentage, the peak above-ground net primary productivity (ANPp) had relatively higher contribution to the TNPp in Sopore site (more than 60%), and the lowest contribution in Thajwas site (less than 10%)

Relationship between floristic diversity and primary productivity

In the monthly data sites, whereas the species richness (actual) showed a positive correlation with ANP in all the three sites, BNP in Dugwan, and TNP in Draphoma and Dugwan; it showed a negative correlation with BNP in Botanic Garden and Draphoma, and TNP in Botanic Garden site. The ANP, BNP and TNP in all the three sites showed a negative correlation with Shannon's diversity index, number of shared species, Jaccard's and Sorensen's similarity indices, except for positive relation of TNP with Shannon's diversity index and number of shared species. In the peak-season data sites, the ANPp showed a positive correlation with species richness (actual), Shannon's diversity index and species evenness index, but negative one with species richness (observed). While as the BNPp showed a positive correlation with species richness (actual) and species richness (observed), it showed a negative one with Shannon's diversity index and species evenness index. The TNPp showed a negative correlation with species richness (actual), species richness (observed), Shannon's diversity index, and species evenness index.

Impacts of livestock grazing on floristic diversity and net primary productivity

In the monthly sampled sites, while as Dignibal-ungrazed site recorded the species richness value of 50 spp., it declined nearly to a half, i.e. 27 spp., in the Dignibal-grazed site. In the Dignibal-ungrazed site, while as the highest number of 36 species was recorded during the month of July, the number was reduced to a half, i.e. 18 spp., in the Dignibal-grazed site, but during the month of June. The Shannon's diversity index in the Dignibal-ungrazed site was 3.81, and that of Dignibal-grazed site was reduced to 3.22. In the 3 peak-season data sites at Aru, the highest value of 59 species was recorded at Aru-long protected site , followed closely by Aru-five-year protected with 58 species, and then by Aru-grazed site with only 14 species. Similar decreasing trend was observed in terms of Shannon's diversity index, with values of 2.5, 2.47 and 1.5, and species evenness values of 0.82, 0.78 and 0.68, in Aru-long protected, Aru-five-year protected, and Aru-grazed sites, respectively. However, in terms of species richness (observed), it was the Aru-five-year protected site which recorded the highest of 24 species, followed closely by Aru-long protected site with 21 species, and by Aru-grazed site with 9 species.

The value of total annual net primary productivity (TNP) in the Dignibal-ungrazed site was 3891.21 $gm^{-2}y^{-1}$, which declined to 2565.04 $gm^{-2}y^{-1}$ in Dignibal-grazed site. More importantly from the forage point of view, the ANP has appreciably higher contribution (*ca.* 50%) to the TNP in the Dignibal-ungrazed site, which showed a drastic decline (*ca.* 20%) in the Dignibal-grazed site. In terms of peak net primary productivity (TNPp), the Dignibal-grazed site showed the highest value of 1329.6 gm^{-2}, followed by Aru-five-year protected and Aru-long protected sites, with values of 1143.4 and 971.8 gm^{-2}, respectively. However, in terms of percentage, the ANPp had relatively higher contribution to the TNPp in Aru-long protected site (*ca.* 30%) and Aru-five-year protected site (*ca.* 20%), while this contribution was the least in Aru-grazed site (*ca.* 5%).

The present study, for the first time, generated reliable empirical data about the floristic diversity and net primary productivity in the Kashmir Himalayan grasslands. This scientific data generated would serve as a key ingredient for formulating an environmentally sound land-use policy, which would take care of the needs of pastorals who are entirely dependent on grasslands for earning their livelihood, and will also help in salvaging the ecological health of these precious ecosystems. The study also emphasizes the need for ecosystem analysis approach to identify and address the ecological and economic issues. For instance, the present study brought into focus the less obvious but critical indirect effects of excessive grazing on species richness, species diversity, and species similarity. This study is hoped to trigger similar studies in other ecologically degraded ecosystems of the region, in particular, and in the entire Himalayan region, in general; the sound field data obtained here would pave way for devising comprehensive management plans, which ultimately would steer the region towards a path of sustainable development.

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