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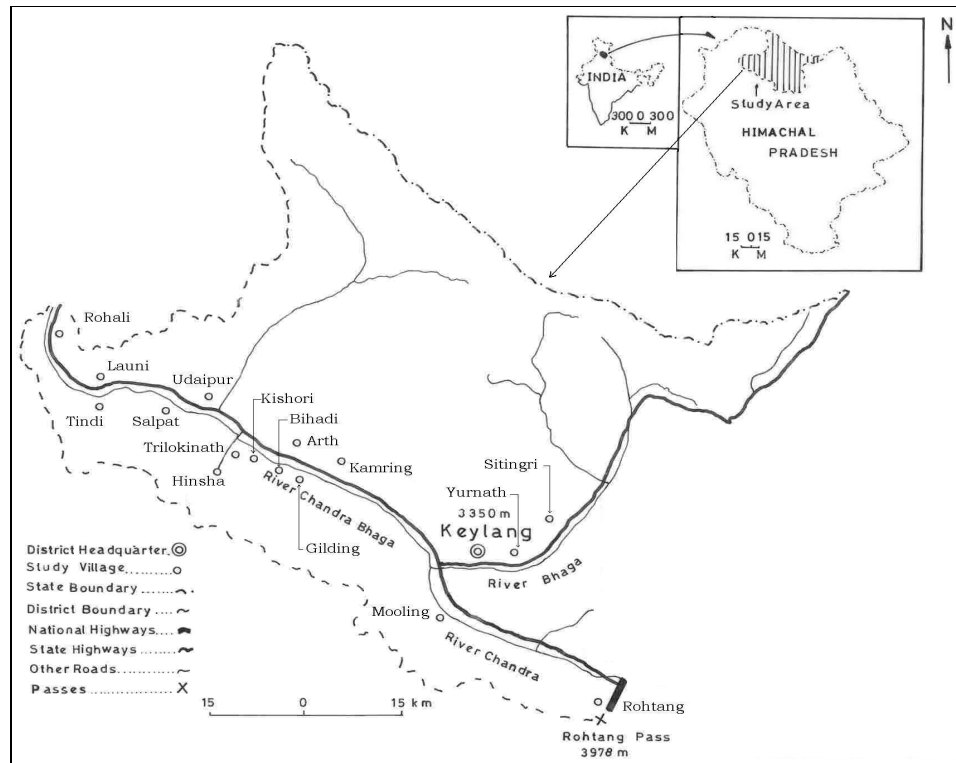
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FOOD AVAILABILITY FOR INSECTIVORES IN GRASSLANDS – ARTHROPOD ABUNDANCE IN PASTURES, MEADOWS AND FALLOW LAND

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Abstract. We studied arthropod occurrence in fallow land, extensively used pastures, extensively used meadows (cut twice or three times a year) and intensively used meadows (cut more than three times a year) in Upper Bavaria. Medium-sized arthropods (5-15mm) were encountered much less frequently on intensively used and fallow land than on pastures and extensively used meadows. Large individuals (>15mm) were observed most frequently on pastures but were hardly found on intensively used meadows. In autumn they occurred almost exclusively on pastures and fallow land. Species richness was the highest on pastures and the lowest on intensively used meadows. The study underscores the depletion of arthropod fauna and in particular the decline of large arthropods on intensively used meadows. It indicates that extensively used meadows and extensively used pastures in particular are the most favourable sources of nutrition for insectivores such as reptiles, amphibians, birds or bats.
Key words: *land use, habitat management, grazing, species diversity, insects*

Introduction

Grasslands are regarded as important foraging areas for many insectivores in Europe, such as birds (Vickery, 2001; Barnet et al., 2004), bats (Güttinger, 1997) or amphibians and reptiles (Langton and Burton 1997). Adequate grassland management is therefore an important tool for the conservation of arthropod eaters (Plantureux et al., 2005). The intensification of grassland usage is supposed to have a negative impact on arthropod diversity (McCracken and Tallowin, 2004). Intensively managed meadows and pastures usually harbour a depleted arthropod fauna (Kruess et al., 2002; Manhart et al., 2004). However, different aspects of grassland management are mainly discussed from the view of insect conservation (e.g. Carvell, 2002; Wetterstein and Schmid, 1999; Vulliamy, 2006; Sjödin, 2007; Radlmair and Dolek, 2002). Intensive application of fertilizers and frequent mowing or heavy grazing have indeed a negative impact on many arthropod species (von Nordheim, 1992; von Wingerden et al., 1992; Benton et al., 2003) and this may have consequences also for species depending on arthropods as food (Vickery, 2001).On the other hand, high biomasses of some arthropods and thus plenty of food may occur at certain times in intensively managed grasslands, because of

the high levels of nutrients in such meadows and pastures (Lauenstein, 1986; Barnett et al., 2004; Atkinson et al., 2005; Plantureux et al., 2005).

In Central Europe extensively used grasslands have become rare. Concerning the food supply for insectivores foraging in open lands, advice for the management of "normal" grasslands on soils of moderate humidity and medium or high nutrient levels is therefore needed. However, the knowledge of the overall arthropod availability in such grasslands is scarce, since many studies about insect populations concentrate on extensive grasslands on poor, dry or wet soils and include only few species or systematic groups (Elligsen et al., 1997; Gibson et al., 1992; Hänsel and Plachter, 2004; Manhart et al., 2004; Kruess and Tscharntke, 2002a, b; Wingerden et al., 1992; Sjödin, 2007 a,b; Perner et al., 2005). Moreover, in temperate regions, data about the arthropod availability in spring and autumn are needed. Climatic conditions reduce the food for arthropod predators during these "critical times" (Zahn et al., 2007a), when body fat storages have to be accumulated (in autumn) or replenished (in spring).

In our study we investigated abundance and body size of arthropods in differently managed types of grasslands with no, moderate or intensive usage.

Study areas

The 5 study areas were located in Upper Bavaria between the valleys of the rivers Isar and Inn:

    a) slopes of the Inn valley near Stampfl (12°19', 48°10'),

    b) a cultivated and drained fen at the river Isen near Walkersaich (12°18'. 48°16'),

    c) moist grasslands at the river Inn near Jettenbach (12°22', 48°09'),

    d) a recultivated gravel pit near Heldenstein (12°23', 48°14'),

    e) a cultivated and drained fen in the valley of the river Isar near Freising (11°40, 48°22').

At these areas we studied grasslands of the following types: fallow land (former usage as meadow or field), extensively used pastures (1.5 – 2.0 animal units/ha), extensively used meadows (cut twice or three times a year), intensively used meadows (cut 4-6 times a year). In each area we chose at least one of each type, except Heldenstein, where only extensively used pastures and intensively used meadow existed. In both Heldenstein and Stampfl, two additional pastures were included. Except Stampfl, where sheep were kept on the pastures, cattle grazed in the other areas. While in Jettenbach cattle stayed on the same parcel from April to November, rotation pastures existed at the other sites.

We excluded grasslands of high botanical value or of usages being unusual nowadays (e.g. cut only once per year) from the study. Additionally we excluded grasslands of very low productivity on very wet, dry or poor soils.

Methods

We took samples from May to October 2005 using a hand net (diameter 40cm, mesh 1mm) between 2 and 7 pm when the weather was dry and warm (> 10 °C). At each site we conducted 100 standardised beats with the net in a minimum distance of 5 m from shrubs and 1m from the grassland edge. The arthropods were stored in a plastic bag and frozen until the analyses. At each site we noticed date, time, weather conditions, mean height of the vegetation and the number of flowering plants per m² (classes: 1 = 0-5

inflorescences/m², 2 = 5-15 inflorescences/m², 3 = 15-30 inflorescences/m², 4 > 30 inflorescences/m²). In August we determined the dominating plant species of the sites and estimated the number of plant species per m² (classes: 1: < 10 species/m², 2 = 10-15 species/m², 3 = 16-20 species/m², 4 > 20 species/m²).

The arthropods were distinguished on the level of the following suborders or families: *Acarina*.

Apoidea, Araneae, Blattodea, Brachychera, Caelifera, Coleoptera, Dermaptera, Ensifera, Formicidae, Heteroptera, Homoptera, Ichneumonidae, Lepidoptera, Mecoptera, Nematocera, Planipennia, Symphyta, Vespoidea. Based on this level we calculated the diversity of each sample (Shannon – Weaver index). As an index of body size we took the body length (including the head). We distinguished between the size classes: < 5mm, 5-15mm, > 15mm. As a rough estimation of differences in the species numbers, we determined the minimum number of species that could be determined by visual inspection according to (Bährmann and Müller, 1995). Since the data were not normally distributed, we used nonparametric statistics. The Mann-Whitney-U-Test was used to test for differences between groups. Relations between independent variables were tested using Spearmans correlations coefficient and Kendall's Tau. If no other explanation is stated, we gave means and standard errors in the result section.

Results

Phenology of arthropods

Heteroptera, *Ensifera*, *Caelifera* and *Coleoptera* were the most abundant from June to August (*Fig. 1a*). *Araneae*, *Hymenoptera* (except ants) and *Homoptera* reached their maximum numbers in late summer and autumn. *Diptera,* which reached the highest numbers of individuals from all groups (*Fig. 1b*), were the most common in spring, whereby very small species (< 3mm) prevailed. Members of other groups occurred only sporadically in the samples. *Acarina* were found mainly in spring, *Formicidae* were more abundant in June, *Lepidoptera, Planipennia* and *Mecoptera* from June to August, *Blattodea* and *Dermaptera* from June to September.
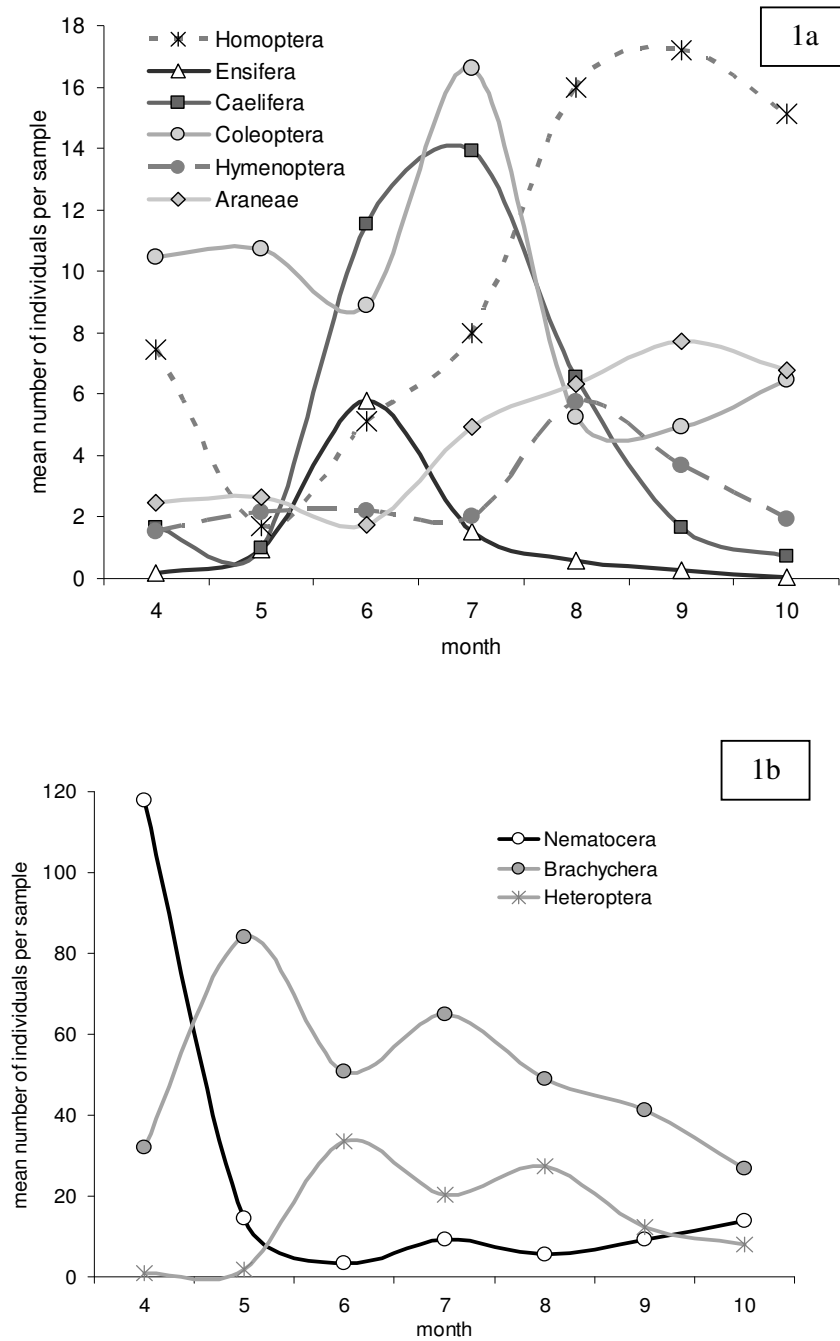
Figure 1. *Phenology of regularly occurring Arthropods.*
Given is the mean number in all samples (n = 15-19) per month (a, b).
Heteroptera, Brachycera and Nematocera are presented separately because of their high
numbers (b)

Effects of grassland management

The arthropod fauna of the studied grassland habitats differed considerably (*Fig. 2*). *Brachycera* were the dominating group in meadows (both extensively and intensively used). In intensively used meadows, most samples consisted mainly of small flies (< 5mm). Besides these, only *Nematocera*, *Heteroptera, Homoptera* and *Coleoptera* occurred regularly in this grassland type.

Hymenoptera (except ants) and *Homoptera* reached the highest numbers in extensively used meadows. *Caelifera, Coleoptera* and *Homoptera* showed their maximum abundances in pastures, *Araneae, Formicidae* and *Ensifera* in fallow land. *Mecoptera* occurred only in pastures and extensively used meadows. In intensively used meadows they were missing, just like *Demaptera* and *Planipennia.*
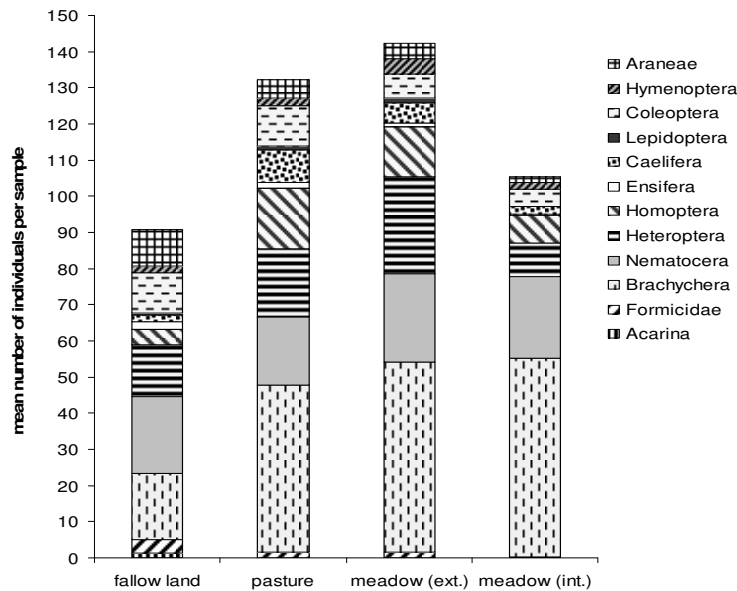

Figure 2. Frequency of different arthropods in samples taken in fallow land (n= 26), pastures (n=60), extensive meadows (n= 26) und intensive meadows (n=33)

Large arthropods (> 15 mm) were the most abundant in pastures and rare in intensively used meadows (*Fig. 3*), where even during summer (June – August) they occurred only in 30% of the samples! On the contrary, 80% of the samples taken in other types of grassland contained large individuals during these months. In autumn, large arthropods were regularly found only in pastures and fallow land: In September and October, they were proven in 75% of the pastures, 62% of fallow lands, 50% of the extensively used meadows and 30% of the intensively used meadows. While in intensively used meadows "large arthropods" consisted only of *Caelifera* and *Nematocera,* we found five groups of arthropods in fallow land, six in extensively used pastures and nine in meadows (*Fig. 4*).

Medium sized arthropods (5-15mm) reached their lowest levels in fallow land and intensively used meadows. They reached their highest numbers in extensively used meadows from May to June (*Fig. 3*). Small arthropods (< 5 mm) occurred mainly in spring and late summer, whereby *Diptera* prevailed. In this size class, the differences

between grassland types were comparatively small. The lowest numbers were found in fallow land (*Fig. 3*).

In most months, the highest diversity (Shannon-Weaver index) was measured in fallow land (*Fig. 5*). Pastures came off second best. Intensively used meadows had the lowest values in summer, but the index was comparatively high in May and did not differ much from extensively used meadows in April, September and October. Over the whole season, the Shannon-Weaver index was the highest in fallow land (1.75 ± 0.01), slightly lower in pastures (1.59 ± 0.05), followed by extensively used meadows (1.44 ± 0.09). The lowest diversity occurred in intensively used meadows (1.2 ± 0.06).
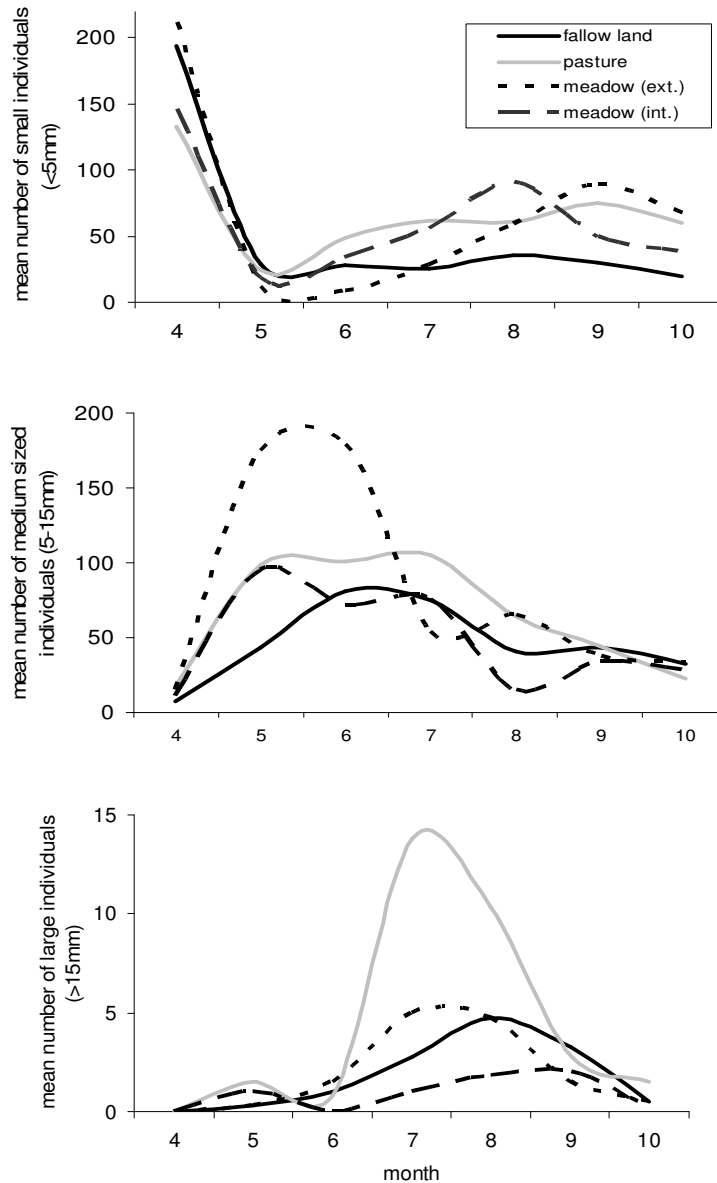

Figure 3. *Frequency of small (< 5 mm), medium sized (5 – 15 mm) and large (> 15 mm) arthropods in different grasslands during the year (means). Fallow land (n= 26), pastures (n=60), extensive meadows (n=26) and intensive meadows (n=33)*

However, in respect of the species richness (variable "minimum species number") the patterns were somewhat different. The most species per sample were found in pastures (23.8 ± 4.2), followed by extensively used meadows (21.5 ± 7.4). In fallow land, the habitat type with the highest diversity, the minimum species numbers were lower than in both extensively used grasslands (20.5 ± 4.1). In the intensively used meadows the value was by far the lowest (13.6 ± 2.3).

The difference between intensively used meadows and the other types of grassland was significant (Mann-Whitney-U-Test; p = 0.002).
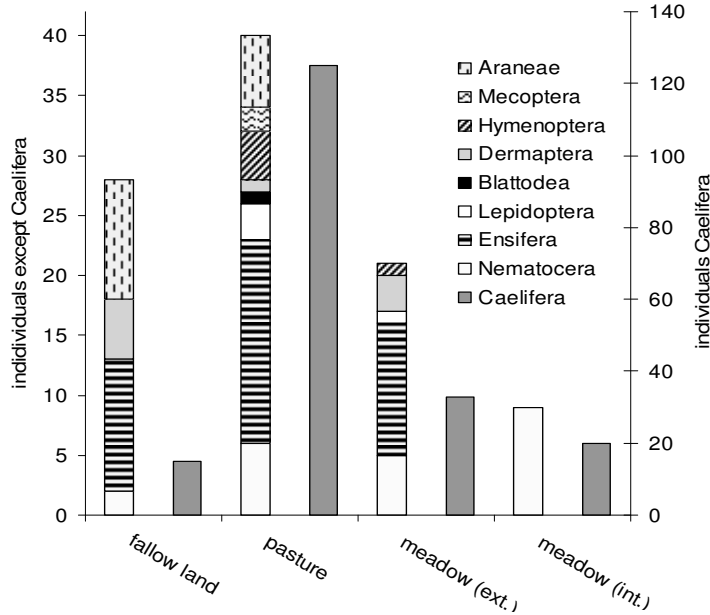

Figure 4. Composition of the group „large arthropods" (> 15 mm). The number of Caelifera is given separately on the right ordinate

Number of plant species, height of the vegetation and density of flowering plants

Intensively used pastures were very poor in plant species. All sample sites belonged to the category 1 (< 10 species /m²). In pastures the highest numbers were reached (two sites belonged to the categories 2 (10-15 species/m²), the other five to the category 3 (15-20 species/m²). In extensively used meadows the category 2 prevailed, only one site belonged to the category 3. Fallow land was poorer in plant species (classes 1 and 2 each at two sites).

The highest numbers of inflorescences in annual average were observed in extensively used meadows and pastures, the lowest numbers in intensively used meadows, where higher flower densities occurred only in short periods (e.g. when *Taraxacum officinale* was blooming in spring). Fallow lands differed to a large extent in flower densities. Some of them were poor in this respect throughout the whole season. It is worth mentioning that in all pastures and to a larger extent even in two extensively used meadows (before the last cut), considerable numbers of inflorescences were observed in autumn (September/October), when flowers were very rare at the other sites.

We pooled all samples to analyse the influence of inflorescence density and vegetation height on the numbers of arthropods. Inflorescence density and vegetation height correlated (r = 0.41, p < 0.01). If partial correlations were calculated, the inflorescence density had a stronger influence on the total individual number (r = 0.33, p < 0.01) and the minimum species number (r = 0.41, p < 0.01) than the height of the vegetation (r = 0.04, ns. and 0.35, p < 0.01, respectively). Consequently we used the inflorescence density to calculate correlations for the different arthropod groups. Except *Nematocera* in all groups the numbers of individuals positively correlated with the density of inflorescences. In *Heteroptera* (r = 0.31), *Caelifera* (r = 0.35), *Ensifera* (r = 0.39), *Lepidoptera* (r = 0.39), *Coleoptera* (r = 0.38) and *Planipennia* (r = 0.27) the relation was significant (p < 0.05, after Bonferroni-correction). Since not all of the groups use flowers directly as a resource, the density of inflorescences must be regarded as an indication of the undisturbed development of the vegetation, because flowers are greatly reduced by mowing, which also reduces the density of many arthropods.
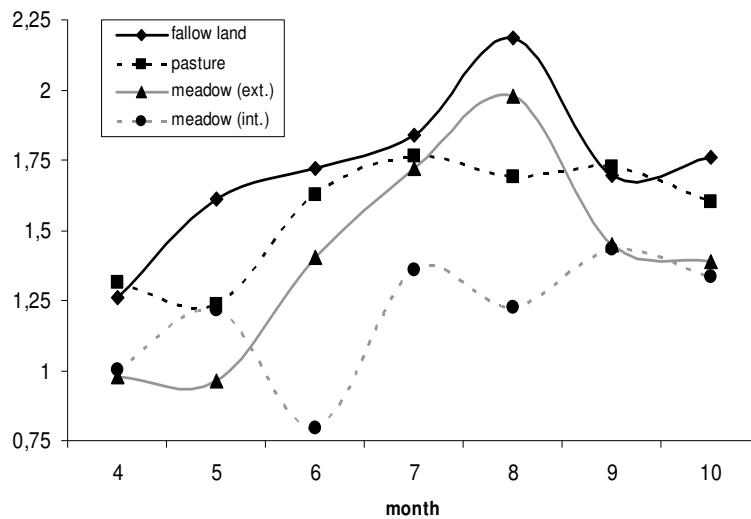

Figure 5. Diversity of species (Shannon-Weaver index) in the studied grasslands throughout the summer season

Discussion

Intensively used meadows

Intensively used meadows were characterised by low numbers of arthropod species, a low diversity of species and, in case of large arthropods, also by low numbers of individuals. While the low species numbers are probably caused by the reduced structural variation and the low diversity of the vegetation (Bornhold, 1991; Fricke and v. Nordheim, 1992), the number of individuals of many arthropod groups is greatly reduced after mowing (Manhart et al., 2004). The highest values in diversity and species richness in intensively used meadows is consequently reached in spring prior to the first cut. In early spring the fast vegetation growth in this type of grassland even favours the arthropod fauna compared to grasslands of lower productivity.

Apart from mowing, the intensive application of fertilizers has a negative impact on some arthropod groups, e.g. on *Caelifera* (von Wingerden et al., 1992; Ingrisch and Köhler, 1998). Similarly Carabid beetles occur in intensively used meadows in reduced densities (Tietze, 1985; Blake et al., 1994). Not only the low abundance but also the low diversity of prey have a negative effect on the predators of arthropods: A higher species diversity results in a more even spatial and temporal distribution of food supply (Plantureux et al., 2005).

However, mass abundance of some arthropods (especially *Diptera*) occurs from time to time in intensively used meadows (Lauenstein, 1986). Such short term food opulence can be exploited by mobile and flexible predators. E.g. mouse-eared bats (*Myotis myotis*) hunt for crane flies (Tipulidea) when they are available in high numbers on freshly mown meadows during summer (Güttinger, 1997). On the contrary, many ground dwelling insectivores like amphibians, reptiles or shrews are not able to settle or forage regularly in intensively used meadows because of the low structural diversity and because of frequent machine processing (Blab, 1984; Langton and Burton, 1997). Overall, intensively used meadows turned out to be by far the most unfavourable type of grassland in respect of the food availability for insectivores.

Fallow land

Fallow lands were inhabited by much more arthropod species than intensively used meadows and resembled in this respect extensively used meadows and pastures. The diversity was even higher than in the other habitats, which corresponds to the findings of Kruess and Tscharntke (2002a,b). However, considering the numbers of medium sized and large arthropods, fallow land performed generally worse than extensively used meadows and pastures. In spring these arthropods were even rarer in fallow land than in intensively used meadows. A main reason might be the fact that the studied types of fallow land offer limited food for herbivore arthropods in this season, because the layer of leaf litter delays the growth of the vegetation. In autumn, on the contrary, fallow lands harboured the highest numbers of large arthropods apart from pastures. Fallow land provides good cover and a high structural diversity at that time, which favours large insects as *Ensifera* and spiders (*Araneae*), which are generally more common in fallow land than in pastures and meadows (Gibson et al., 1992). However, large differences exist between different types of fallow grassland: Older sites, characterised by low plant diversity and a dense layer of leaf litter, often show low species numbers in many groups of arthropods (e.g. Dewenter and Tscharntke, 2001), which can be enhanced by extensive types of usage (Zahn et al., 2007). In contrast, fallow grasslands on poorer soils may harbour rich arthropod communities (Balmer and Erhardt, 2000). In every case, the arthropods in fallow grasslands may be hard to exploit for insectivores, especially bats and many birds, which avoid foraging in dense high vegetation (Atkinson, 2005; Güttinger, 1997; Devereux et al., 2004).

Extensively used meadows

Many authors report a high faunal diversity of extensively used meadows (Ausden and Treweek, 1995; Bornhold, 1991), but most studies refer to meadows of a very moderate cultivation, e.g. only one cut per year. Two or more cuts per year immediately lead to a reduction of arthropod diversity (von Nordheim, 1992; Plantureux et al., 2005). The sites included in our study were mown two or three times a year, and the arthropod

species numbers ranked between fallow land and pastures, whereas the diversity was on average lower than in these habitats. The numbers of individuals were comparatively high, especially in case of medium sized arthropods. However, large arthropods occurred less frequent than in pastures. Probably this was the result of the first cut, which greatly reduces the juvenile stages e.g. of *Caelifera* and *Ensifera* (Manhart et al., 2004). Therefore, these groups make an appearance mainly as medium sized individuals prior to mowing. Similarly, the number of spiders is reduced by mowing in the long term. However, due to the high number of inflorescences, especially flower visiting insects are favoured by this habitat (Sjödin, 2007; Sjödin et al., 2008) if the mowing regime is adjusted to their requirements: mowing of neighbouring areas at different times.

Pastures

Intensively used pastures show a poor arthropod diversity and in case of large arthropod species, the density is also low (v. Nordheim, 1992; Kruess and Tscharntke, 2002a,b; Radlmair and. Dolek, 2002). In case of ration grazing, the high grazing intensity in a short time has similar effects on arthropods as a cut. However, even in heavily grazed grassland dung beetles may represent a considerable amount of food for insectivores (Zahn and Hirschberger, 2001). This resource can be exploited e.g. by birds and bats which are able to forage in extended open areas without cover.

The extensively grazed pastures included in our study harboured the most diverse arthropod fauna of all habitats under cultivation, and the most species of all habitats. In respect of large arthropods, even the highest numbers of individuals within the compared types of grasslands were reached. Especially in autumn (October) pastures offered the best food supply for predators of large arthropods. Extensively used meadows harboured more large individuals on average only in early summer, probably because prior to the first cut the vegetation growth was less disturbed, compared to grazed pastures where the livestock roamed. Additionally the layer of inflorescences was much denser in extensively used meadows than in pastures until the first cut, which favours flower visiting insects (Sjödin, 2007a,b). After the first cut, the studied all-year grazed pastures (without additional mowing!) gained advantage because grazing with low intensities causes less disturbance of the vegetation than cutting, including the inflorescences layer.

Species number and diversity of arthropods are strongly influenced by the type of grazing (all-year grazing, rotation pasture), livestock species, and density and intensity of pasture care. A high grazing pressure may cause considerable damage for the fauna (Dennis et al., 1997; Gibson et al., 1992; Plantureux et al., 2005). A habitat mosaic, which is often a result of low density all-year grazing or pastoralism, is regarded as a key factor for a high diversity of the fauna (Ausden and Treweek, 1995; Riecken et al., 1998; Sjödin et al., 2008; Vulliamy et al., 2006).

Contrary to meadows the spatial structure of the vegetation in areas of low density all-year grazing shows few variations: The extent and the distribution patterns of short lawn-like sites on the one hand, and of taller vegetation on the other hand, fluctuate only gradually over the years (Zahn et al., 2003). This causes fairly constant and predictable conditions for animals and their needs in respect of foraging, shelter, reproduction or hibernation (Zahn et al., 2007b). Moreover, the mosaic structure of the vegetation

allows the occurrence of animals that differ considerably in respect of their habitat requirements (Nyffeler, 1998; Plantureux et al., 2005).

Figure 6. All-year grazed pasture in Jettenbach. The remaining patches of taller vegetation are important microhabitats for arthropods in autumn and winter

Recommendation for habitat management

None of the studied habitat types offered plenty of food all the time, a fact that supports the importance of differences in grassland usage on a local scale, if the arthropod abundance and therefore the food supply for insectivores should be optimised (Söderström, 2001; Wetterstein and Schmid, 1999). Noteworthy the abundance of arthropods, especially of larger individuals, was on average higher in pastures and extensively used meadows than in fallow land and intensively used meadows.

In spite of the fact that our study included only a part of the arthropod fauna (ground dwelling species were excluded), some general differences between the habitat types can be derived:

In grassland on soils of medium or high nutrient levels, the arthropod diversity and abundance (and therefore the food supply for insectivores) is the best in case of low intensity grazing or mowing all over the year. On fallow land of such soil types (with or without mulching), the arthropod availability is lower than in grasslands of extensive usage.

In all-year grazed pastures without additional mowing the left over patches of taller vegetation *(Fig. 6)* are important microhabitats for arthropods in autumn and winter (Hänsel and Plachter, 2004; Zahn et al., 2007b; Plantureux et al., 2005), which are missing on meadows. Large arthropods like grasshoppers survive even the first frosty nights in these shelters (Ingrisch and Köhler, 1998). Especially groups such as orb-weavers or bush-crickets, which are greatly reduced by mowing (Manhart et al., 2004), are favoured by the habitat mosaic of areas with low grazing intensity. The patchy structure of intensively grazed lawn-like sites next to left over taller vegetation created by the livestock is therefore of high value from the viewpoint of arthropod conservation

(Atkinson et al., 2004; Perkins et al., 2000; McCracken and Tallowin, 2004). Additional mowing, which is suggested by agricultural experts (Elsässer 2000), should be avoided.

In case of ration grazing or meadows, where such a habitat mosaic does not exist, stripes of fallow or rarely mown grassland along the edges can have a comparable function for arthropods and may therefore favour the diversity and richness of species.

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CONSERVATION PRIORITIZATION OF HABITATS AND FOREST COMMUNITIES IN THE LAHAUL VALLEY OF PROPOSED COLD DESERT BIOSPHERE RESERVE, NORTH WESTERN HIMALAYA, INDIA

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Abstract. The rapid loss of biodiversity due to habitat degradation and over exploitation has necessitated the conservation prioritization of habitats, species and communities for conservation. The prioritization of habitats and communities through qualitative and quantitative assessment of vegetation is prerequisite for initiating any conservation and management programme. Therefore, an attempt has been made to; (i) identify the habitats and communities; (ii) evaluate habitats and communities for species richness, native, endemic, economically important and threatened species; (iii) prioritize habitats and communities for conservation. Fifteen (15) habitats and fourteen forest (14) communities distributed between 2490-4000m were recorded. Overall, 35.71% communities were broad-leaved, 57.14% coniferous and 7.14% mixed. The habitats and communities have been evaluated for the species richness, native, endemic, economically important and threatened species. Based on these attributes habitats and communities have been prioritized. Among communities *Abies pindrow - Pinus wallichiana* mixed, *Fraxinus xanthoxyloides*, *Picea smithiana - Pinus wallichiana* mixed and *Cedrus deodara -Acer cappadocicum* mixed showed highest CPI and habitats forest, shady moist showed highest CPI value. Regular monitoring of the prioritized habitats and communities has been suggested. Also, mass scale propagation of native, endemic, economically important and threatened species and their plantation in the Lahaul valley have been suggested.
Key words: structure, composition, habitat, community, conservation, prioritization

Introduction

The Indian Himalayan Region (IHR) is very well known for its representative, natural, unique and socio-economically important plant diversity (Samant et al., 1998a). It is designated as one of the Biodiversity Hot Spots (Anonymous, 2007). It supports 18 440 species of plants with 25-30% of endemics (Samant et al., 1998a; Singh and Hajra, 1996). The inhabitants use this rich diversity for their sustenance. The increase in human population has increased the demand of economically important biodiversity elements. This has caused the over exploitation and habitat degradation of many economically important biodiversity elements and led the rapid loss of these elements. In view of the rapid loss of biodiversity elements, ecological and economical evaluation of the habitats and

communities are essentially required. The review of literature indicates that in general, a large number of studies have been carried out on the flora, ecology, ethnobotany and rare endangered plants separately in the IHR (Aswal and Mehrotra, 1994; Chowdhery and Wadhwa, 1984; Dhaliwal and Sharma, 1999; Dhar et al., 1997; Kalakoti et al., 1986; Maity and Chauhan, 2002; Nautiyal et al., 1997; Rawal et al., 1994; Rawal and Pangtey, 1994; Rawat et al., 2001; Rawat et al., 1989; Rikhari et al., 1989; Samant and Joshi, 2004; Samant et al., 2002; Saxena and Singh, 1982; Singh and Singh, 1992; Singh and Rawat, 1999; Singh et al., 1996). In the IHR, a very few studies have been carried out to prioritize potential species and altitudinal zones (Dhar et al., 2000; Dhar and Samant, 1993; Pandey, 2006; Samant et al., 2007a; Samant and Pal, 2003; Samant et al., 2007b; Samant et al., 2007c), and habitats and communities (Arya, 2002; Joshi, 2002; Joshi and Samant, 2004; Pant and Samant, 2007; Samant et al., 2002). However, prioritization of habitats and communities based on species richness, native, endemic, economically important and threatened plants have been carried out by a few workers (Joshi and Samant, 2004; Pant and Samant, 2007). In Himachal Pradesh such integrated studies are not available. Therefore, present attempt has been made to; (i) assess the forest vegetation for community identification; (ii) evaluate the habitats and communities for species richness, native, endemic, economically important and threatened plants; (iii) prioritize habitats and communities for conservation; and (iv) suggest conservation measures.

Materials and methods

Study area

The study has been carried out during 2004-2007 in Lahaul Valley (32°22.517'N – 32° 48.564'N Latitudes and 76°25.017'E – 77°16.636'E Longitudes) of a proposed Cold Desert Biosphere Reserve (CDBR) (*Fig. 1*).

The climate varies from dry temperate to alpine types. The area remains snow covered almost for six months and receives average snowfall 120-400 cm year^{-1}, and average rainfall 10-300 mm year^{-1}. The temperature ranges between -19°-32° C (Sinha and Samant, 2006). The highest mountain peak is Mulkila (6,517 m) and lowest point at Karunallah, (2 400 m) the entrance of Chenab into Pangi valley. The whole area of Lahaul is divided into three valleys namely, Chandra, Bhaga and Chandra-Bhaga. The rock system is very fragile and liable to erosion which is often accentuated by the rigorous of severe winters, avalanches and the strong winds that accompany them. The valley is inhabited by a large number of villages and the inhabitants are largely dependent on biodiversity elements for their sustenance.

Survey, sampling, identification and data analysis

Surveys were conducted to select sites and habitats along the Lahaul valley in forest zone between 2490-4000m amsl. The habitats were identified on the basis of physical characters and dominance of vegetation. Attempts have been made to select sites and habitats on each and every accessible aspects. In each site, a plot of 50 × 50 m was laid.

Trees, saplings and seedlings were sampled by randomly placed 10, 10x10 m quadrats; shrubs by 20, 5 × 5 m quadrats; and herbs by 20, 1 × 1 m quadrats in each plot. For the collection of data from these quadrats standard ecological methods were followed (Curtis and Mc Intosh, 1950; Dhar et al., 1997; Greig-Smith, 1957; Misra, 1968; Mueller-Dombois and Ellenberge, 1974; Samant et al., 2002; Joshi and Samant, 2004). From each site, samples of each species were collected and identified with the help of floras (Aswal and Mehrotra, 1994; Chowdhery and Wadhwa, 1984; Dhaliwal and Sharma, 1999; Polunin and Stainton, 1984; Murti, 2001).
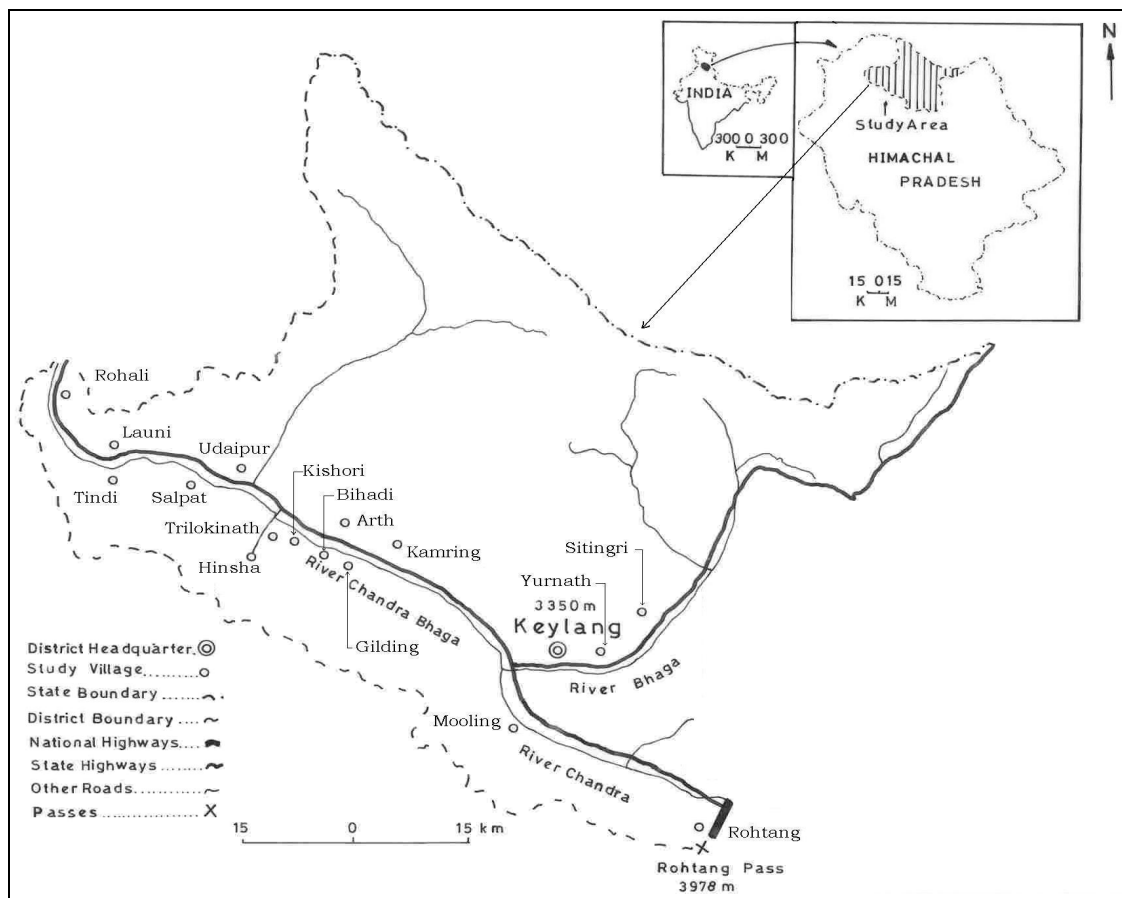

Figure 1. *Study area*

The forest communities were identified on the basis of IVI values of trees. The single tree species representing > 50% of the total IVI was designated as a single species dominated community, whereas two or more species contributing 50 or > 50% of the total IVI were named as a mixed community. Species richness was determined as the number of species.

Identification of native, endemic, economically important and threatened plants

The species with its origin or first record from the Himalayan Region were considered as natives (Samant et al., 1998a; 2002). In case of Pteridophytes the endemic and near-endemic species have been considered as natives to the Himalayan region. The species restricted to IHR have been considered as endemic, whereas those extending their distribution to neighbouring Countries/States were considered as near-endemic (Dhar and Samant, 1993; Samant and Dhar, 1997; Samant et al., 1996a; 1998a).

The information on economically important species was generated through Participatory Rural Appraisal (PRA) (Samant et al., 2002; 2003) and the interviews of the knowledgeable persons including Amchies (local herbal doctors). Among the village experts, one person was hired to survey and collect the economically important species from wild habitats. Fresh samples of the useful species were collected and identified with the help of florulas (Aswal and Mehrotra, 1994; Chowdhery and Wadhwa, 1984; Polunin and Stainton, 1984).

The threatened species were identified based on habitat preference, distribution range, population size, use pattern, extraction trend, nativity and endemism of the species (Samant et al., 1996b; 1998b; 2002).

Prioritization of habitats and communities

The prioritization of habitats and communities has been done using eight parameters like species richness, economically important, native, endemic, threatened plants, altitude, site representation and habitats number following Joshi and Samant (2004); Pant and Samant (2007); Samant et al. (2002) (*Table 1*).

Table 1. *Criteria's used for the prioritization of habitats and communities*

Marks	Richness (%)	EIP (%)	Native (%)	Endemic (%)	Threatened (%)	SR	Altitude (m)	Habitats *
10	>50	>45	>45	>40	>40	1	<200	1
8	46-50	41-45	41-45	36-40	36-40	2	200-400	2
6	41-45	36-40	36-40	31-35	31-35	3	400-600	3
4	36-40	31-35	31-35	26-30	26-30	4	600-800	4
2	<36	<31	<31	<26	<26	>4	>800	>4

Abbreviations: EIP = Economically Important Plants; SR = Site representation; and * = Criteria only applied for the communities

Results

Habitat diversity

Fifteen habitats (15) Rocky, Bouldary, Shady moist, Forest, Parasite, Degraded, Dry, Riverine, Water courses, Grassland, Marshy, Shrubberies, Near-settlements, Camping sites, and Road sides were identified (*Table 2*). The bouldary, dry, rocky, shady moist and grassland habitats showed wide range of distribution. The site representation varied from 3-16, Species richness ranged from 5-259, natives 3-104, endemics 0-87, economical important species 3-202, and threatened species 3-98. Amongst the habitats, maximum

species richness (259), native (104), endemic (87), economically important (202), threatened (98) species were recorded in the forest habitat, followed by shady moist, species richness (235), native (94), endemic (84), economically important (191), threatened (93); dry habitat, species richness (170), native (69), endemic (42), economical important (134), threatened (45); grasslands, species richness (146), native (47), endemic (28), economical important (143), and threatened (29) species were recorded. The remaining habitats showed relatively less number of species (*Table 2*). The notable native, endemic, economically important and threatened species of the prioritized habitats have been presented in *Table 3*.

Table 2. Prioritization of habitats for conservation in Lahaul valley of the proposed CDBR

Habitat Type	SR	AD (m)	SPR	N	En	EIP	TS	CPI
Rocky	15	2500-4000	106	40	30	77	35	14
Bouldary	16	2490-4000	88	24	18	62	28	14
Shady Moist	15	2400-4000	235	94	84	191	93	38
Forest	14	2500-4000	259	104	87	202	98	44
Parasitic	3	2500-3690	5	3	3	3	4	14
Degraded	12	2500-4000	56	12	11	44	10	14
Dry	15	2490-4000	170	69	42	134	45	14
Riverine	14	2500-4000	71	23	19	60	17	14
Water Courses	11	2490-4000	21	9	5	14	6	14
Grassland	15	2400-4000	146	47	28	143	29	14
Marshy	11	2500-3855	17	4	-	11	3	14
Shrubberries	14	2500-4000	45	15	11	38	14	14
Near Settlements	13	2490-4000	52	12	10	50	11	14
Camping Sites	14	2500-4000	51	13	8	49	3	14
Road Sides	12	2500-3900	60	14	11	58	13	14

Abbreviations: SR = Site representation; AD = Altitudinal Distribution; EIP = Economically Important Plants; CPI = Conservation Priority Index; SPR = Species Richness; N = Native; En = Endemic; and TS = Threatened Species

Community diversity, species composition and structural pattern

Overall 96 sites representing 8 aspects and 15 habitats were sampled and 14 tree communities from forests zone were identified (*Table 4*). The identified communities were broad leaved deciduous (i.e.*,* Betula utilis, Hippophae salicifolia, Juglans regia - Ulmus wallichian - Acer acuminatum mixed, Salix daphnoides and Fraxinus xanthoxyloides); evergreen coniferous and deciduous broad leaved mixed (i.e.*,* Cedrus deodara - Acer cappadocicum mixed), and coniferous evergreen (i.e., Abies pindrow, Abies pindrow - Pinus wallichiana mixed, Cedrus deodara, Juniperus polycarpos, Juniperus polycarpos - Cedrus deodara mixed, Picea smithiana, Picea smithiana - Pinus wallichiana mixed and Pinus wallichiana) communities. The communities having relatively wide altitudinal range of distribution were Juniperus polycarpos, Pinus wallichiana, Cedrus deodara and Picea smithiana (*Table 4*).

Table 3. *Some important native, endemic, near-endemic, economically important and threatened species of the main prioritized habitat*

Prioritized habitats	Native	Endemic/Near-Endemic	Economically important	Threatened
Forests	Selinum coniifolium, Campanula aristata, Cyananthus lobatus, Rhododendron campanulatum, Picrorhiza kurrooa	Allium stracheyi, Berberis pseudumbellata, Campanula cashmeriana, Dactylorhiza hatagirea	Carum carvi, Heracleum candicans, Hippophae rhamnoides, Corydalis govaniana, Gentiana coronata, Juglans regia, Rheum webbianum, Abies pindrow	Allium stracheyi, Heracleum thomsonii, Selinum coniifolium, Lonicera spinosa, Dioscorea deltoidea, Gentianella moorcroftiana, Swertia alternifolia, Polygonatum cirrhifolium, Aconitum heterophyllum, Juniperus indica
Shady Moist	Acer acuminatum, Angelica glauca, Bupleurum candollii, Chaerophyllum villosum, Indigofera heterantha	Acer acuminatum, Chaerophyllum villosum, Erigeron bellidioides, Codonopsis clematidea	Allium carolinianum, Angelica glauca, Chaerophyllum reflexum, Corylus jacquemontii, Cedrus deodara, Taxus baccata subsp. wallichiana	Acer acuminatum, Allium victorialis, Bunium persicum, Inula grandiflora, Saussurea glanduligera, Onosma hispida, Lilium polyphyllum
Dry	Bupleurum lanceolatum, Cortia depressa, Ligusticum elatum, Anaphalis busua, Eremurus himalaicus	Bupleurum lanceolatum, Heracleum thomsonii, Berberis jaeschkeana, Cortia depressa, Aster indamellus, Echinops cornigerus	Bunium persicum, Bupleurum falcatum, Ferula jaeschkeana, Caragana versicolor, Ribes alpestre, Ajuga bracteosa	Saussurea deltoidea, Vincetoxicum hirundinaria, Lonicera hypoleuca, Datisca cannabina, Syringa emodi, Hyoscyamus niger, Juniperus polycarpos
Bouldary	Cirsium verutum, Gnaphalium thomsonii, Hackelia uncinata, Rhodiola tibetica, Hedysarum astragaloides, Poa koelzii	Cirsium verutum, Silene moorcroftiana, Rhodiola heterodonta, Oryzopsis lateralis, Physochlaena praealta, Cystopteris montana	Cardamine impatiens, Nepeta eriostachya, Podophyllum hexandrum, Rheum australe, Bergenia stracheyi, Ephedra gerardiana, Adiantum capillus-veneris	Silene moorcroftiana, Hypericum perforatum, Meconopsis aculeata, Podophyllum hexandrum, Bergenia ligulata, Physochlaena praealta, Viola biflora, Athyrium davidii
Degraded	Cousinia thomsonii, Astragalus rhizanthus, Rubus cordifolius, Bupleurum lanceolatum, Ligusticum elatum, Senecio krascheninnikovii	Astragalus bicuspis, Agrostis pilosula, Bupleurum lanceolatum, Cirsium wallichii, Echinops cornigerus	Cousinia thomsonii, Medicago falcata, Melilotus officinalis, Thymus linearis, Plantago depressa, Prunus armeniaca, Verbascum thapsus, Pinus wallichiana	Hyssopus officinalis, Bupleurum lanceolatum, Arctium lappa, Artemisia maritima, Juniperus polycarpos, Solanum nigrum

Amongst the communities, tree density was maximum for Hippophae salicifolia community (1850.00 Ind ha^{-1}), followed by Fraxinus xanthoxyloides (1000.00 Ind ha^{-1}), Juglans regia - Ulmus wallichiana - Acer acuminatum mixed (760.00 Ind ha^{-1}), Abies pindrow - Pinus wallichiana mixed (640.00 Ind ha^{-1}), Juniperus polycarpos - Cedrus deodara mixed (600.00 Ind ha^{-1}). Cedrus deodara - Acer cappadocicum mixed (170.54 m^2 ha^{-1}) community had lowest density. The total basal area (124.89 m^2 ha^{-1}) was maximum for Abies pindrow - Pinus wallichiana mixed community, followed by Juniperus polycarpos - Cedrus deodara mixed (110.64 m^2 ha^{-1}), Picea smithiana (92.26 m^2 ha^{-1}), and Juglans regia - Ulmus wallichiana - Acer acuminatum mixed (91.23 m^2 ha^{-1}). Juniperus polycarpos (13.65 m^2 ha^{-1}) community showed lowest total basal area. Total shrub density was highest in Cedrus deodara (2611.7 Ind ha^{-1}) community, followed by Hippophae salicifolia (2520.00 Ind ha^{-1}), Betula utilis (2230.00 Ind ha^{-1}) and Abies pindrow (1994.00 Ind ha^{-1}). Pinus wallichiana community showed the highest herb density (95.10 Ind m^{-2}), followed by Hippophae salicifolia (63.85 Ind m^{-2}), Juniperus polycarpos (32.77 Ind m^{-2}) and Picea smithiana (32.40 Ind m^{-2}) communities. Fraxinus xanthoxyloides community showed the maximum seedlings density (1300.00 Ind ha^{-1}), followed by Abies pindrow (557.18 Ind ha^{-1}) and Abies pindrow - Pinus wallichiana mixed (390.00 Ind ha^{-1}) communities. Highest saplings density was recorded for Cedrus deodara community (816.42 Ind ha^{-1}), followed by Hippophae salicifolia (790.00 Ind ha^{-1}), Juniperus polycarpos (750.58 Ind ha^{-1}), Juglans regia - Ulmus wallichiana-Acer acuminatum mixed (660.00 Ind ha^{-1}) communities (*Table 4*).

Table 4. *Community types, distribution pattern, structural pattern and major tree associates in Lahaul valley of the Proposed CDBR*

Community types	SR	Altitudinal range (m)	Habitat type (s)	Slope (°)
Juniperus polycarpos	43	2760-3782	A, B, C, D, E, F	35-60
Pinus wallichiana	13	2845-3518	B, C, D, E, F	35-50
Cedrus deodara	11	2550-2830	A, E, C, D, F	20-60
Abies pindrow	6	3220-3440	A, D, F	40-55
Picea smithiana	8	2742-3100	A, B, D, F	40-50
Betula utilis	*5*	*3440-3855*	*A, D, E, F*	*55-70*
Juniperus polycarpos - Cedrus deodara mixed	*2*	*2760-2780*	*A, B*	*35-50*
Hippophae salicifolia	1	3000	L	35
Abies pindrow -Pinus wallichiana mixed	1	3268	D	65
Juglans regia - Ulmus wallichiana - Acer acuminatum mixed	1	2490	D	25
Picea smithiana - Pinus wallichiana mixed	2	2510-2650	D	50-60
Salix daphnoides	1	2560	A	55
Cedrus deodara - Acer cappadocicum mixed	1	2560	A	60
Fraxinus xanthoxyloides	1	2580	A	65

Table 4. cont.

Community types	Density (Ind ha^{-1})					TBA (m^{2}ha^{-1})	Major Associate Species
	Trees	**Seedlings**	**Saplings**	**Shrubs**	**Herbs**		
Juniperus polycarpos	448.55	89.03	750.58	696.24	32.77	13.65	*Picea smithiana & Pinus wallichiana*
Pinus wallichiana	456.01	222.05	362.42	1178.97	95.10	32.46	*Juniperus polycarpos & Betula utilis*
Cedrus deodara	422.12	178.75	816.42	2611.70	28.24	65.14	*Juniperus polycarpos, Robinia pseudoacacia, Pinus wallichiana, Picea smithiana, Acer acuminat. & Celtis australis*
Abies pindrow	475.79	557.18	162.15	1994.00	22.03	61.43	*Pinus wallichiana & Betula utilis*
Picea smithiana	399.76	290.52	337.26	767.20	32.40	92.26	*Pinus wallichiana, Juniperus polycarpos, Taxus baccata subsp. wallichiana & Corylus jacquemontii*
Betula utilis	588.52	136.30	330.00	2230.00	30.48	55.15	*Pinus wallichiana & Abies pindrow*
Juniperus polycarpos - Cedrus deodara mixed	600.00	216.67	566.67	1290.00	26.65	110.64	*Pinus wallichiana, Juglans regia & Pyrus pashia*
Hippophae salicifolia	1850.00	-	790.00	2520.00	63.85	22.07	*Rosa webbiana*
Abies pindrow - Pinus wallichiana mixed	640.00	390.00	310.00	545.00	30.96	124.89	*Picea smithiana*
Juglans regia -Ulmus wallichiana - Acer acuminatum mixed	760.00	-	660.00	490.00	23.77	91.23	*Ulmus villosa, Prunus cornuta, Fraxinus micrantha, Acer cappadoc, Corylus jacquemontii & Pinus wallichiana*
Picea smithiana - Pinus wallichiana mixed	305.56	290.00	310.00	1900.00	20.11	69.18	*Abies pindrow, Picea smithiana, Pinus wallich. Acer acuminatum & Corylus jacquemontii*
Salix daphnoides	220.00	-	500.00	610.00	23.67	25.33	*Cedrus deodara*
Cedrus deodara - Acer cappadocicum mixed	170.00	-	-	690.00	17.18	36.23	*Celtis australis*
Fraxinus xanthoxyloid.	1000.00	1300.00	300.00	480.00	7.83	46.97	*Corylus jacquemontii*

Abbreviations: m = Meters; Ind = Individual; ha^{-1} = Per hectare; A = Bouldary; B = Degraded; C = Dry slopes; D = Shady moist slopes; E = Riverine; F = Rocky; L = Grassland and SR = Site representation

Species richness

The species richness within the identified communities for trees ranged from 1-9, shrubs 3-23, herbs 9-213, seedlings 0-5, and saplings 0-4. The richness of trees was highest in Juglans regia - Ulmus wallichiana - Acer acuminatum mixed community (i.e., 9), followed by Picea smithiana - Pinus wallichiana mixed community (8). The richness of shrubs was highest in Juniperus polycarpos community (23), followed by Cedrus deodara (18) and Pinus wallichiana (15) communities, whereas richness of herbs was again highest in Juniperus polycarpos community (213), followed by Pinus wallichiana (153) and Cedrus deodara (92) communities (*Table 4*).

Native, endemic, economically important and threatened species

The native species ranged from 7-90, endemic species 6-63, economically important species 15-194 and threatened species 6-47 within the communities. Maximum species were recorded in Juniperus polycarpos community (native 90; endemic 63; economically important 194), followed by Pinus wallichiana (native 81; endemic 61; economically important 147), Cedrus deodara (native 52; endemic 30; economically important 101), Picea smithiana (native 40; endemic 16; economically important 69) and Betula utilis (native 30; endemic 26; economically important 52) communities. The remaining communities showed comparatively less native and endemic species (*Table 5*).

Table 5. *Prioritization of forest communities for conservation using different parameters in the Lahaul valley of Proposed CDBR*

Community Type	Habitat(s)	SR	AD (m)	SPR	N	En	EIP	TS
Juniperus polycarpos	6	43	2760-3700	242	90	63	194	47
Pinus wallichiana	5	13	2845-3500	173	81	61	147	46
Cedrus deodara	5	11	2550-2830	117	52	30	101	33
Abies pindrow	3	6	3220-3440	57	30	16	49	15
Picea smithiana	4	8	2742-3100	83	40	16	69	20
Betula utilis	4	5	3440-3855	59	30	26	52	17
Juniperus polycarpos-Cedrus deodara mixed	2	2	2760-2780	39	12	11	36	12
Hippophae salicifolia	1	1	3000-3100	30	13	9	24	7
Abies pindrow-Pinus wallichiana mixed	1	1	3260-3340	24	14	10	21	9
Picea smithiana-Pinus wallichiana mixed	1	2	2490-2600	46	16	18	44	22
Juglans regia-Ulmus wallichiana-Acer acuminatum mixed	1	1	2450-2550	50	15	13	42	23
Cedrus deodara-Acer cappadocicum mixed	1	1	2560-2630	26	9	8	23	13
Salix daphnoides	1	1	2560-2620	28	8	6	24	7
Fraxinus xanthoxyloides	1	1	2500-2650	15	7	6	15	6

Abbreviations: SR = Site Representation; AD = Altitudinal Distribution; EIP = Economically Important Plants; SPR = Species Richness; N = Natives; En = Endemics; and TS = Threatened species

Amongst the communities, threatened species ranged from 6-47 (*Table 5*). It was highest in Juniperus polycarpos community (Endangered 1; Vulnerable 11; Near Threatened 35), followed by Pinus wallichiana (Critically Endangered 2; Endangered 5; Vulnerable 13; Near Threatened 26), Cedrus deodara (Critically Endangered 2; Endangered 2; Vulnerable 7; Near Threatened 22), Juglans regia-Ulmus wallichiana-Acer acuminatum mixed (Critically Endangered 1; Endangered 8; Vulnerable 6; Near Threatened 8), Picea smithiana-Pinus wallichiana mixed (Critically Endangered 1; Endangered 2; Vulnerable 7; Near Threatened 12) and Picea smithiana (Vulnerable 2; Near Threatened 18) communities. The remaining communities showed comparatively less number for threatened species. The notable native, endemic, economically important and threatened species of the prioritized communities have been presented in *Table 6.*

Prioritization of habitats and forest communities

Amongst habitats, the forest habitat showed highest (i.e., 44) Conservation Priority Index (CPI), followed by shady moist (CPI: 38) habitat. The remaining habitats showed CPI 14, each (*Table 2*).

Among the communities, Abies pindrow - Pinus wallichiana mixed community showed highest i.e.*,* 70 Conservation Priority Index (CPI), followed by Fraxinus xanthoxyloides (68) and Picea smithiana - Pinus wallichiana mixed and Cedrus deodara - Acer cappadocicum mixed (62, each) communities. However, lowest CPI (38, each) was recorded for Juniperus polycarpos and Cedrus deodara communities (*Table 1; Figure 2*).
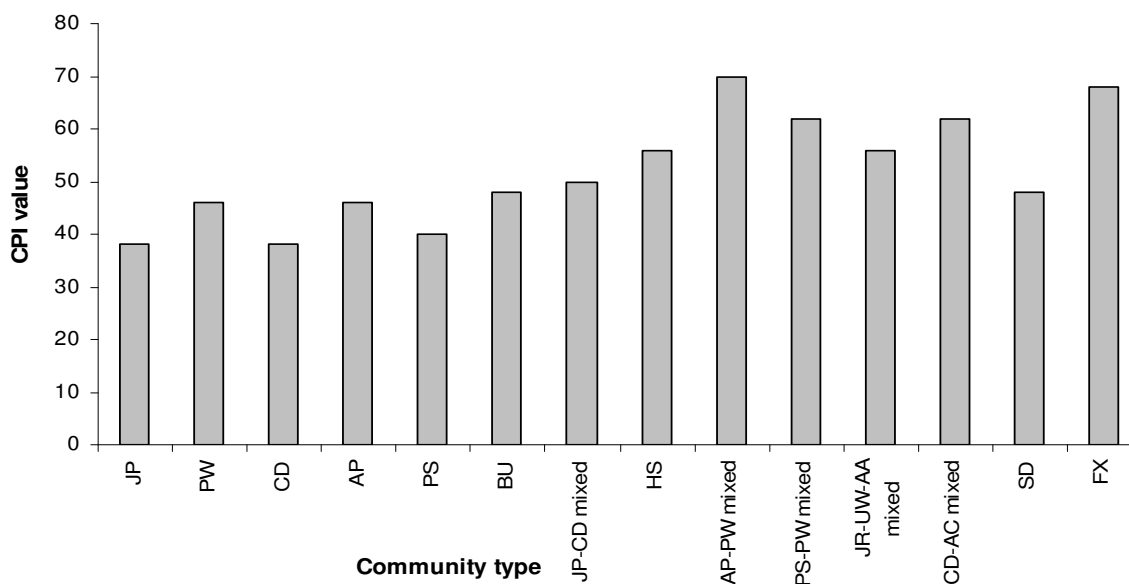

Figure 2. *Conservation Prioritization Index of forest communities in Lahaul valley of the Proposed CDBR*

Table 6. *Some important native, endemic, near-endemic, economically important and threatened species of the prioritized communities*

Prioritized communities	Native	Endemic/near-endemic	Economically important	Threatened
Abies pindrow-Pinus wallichiana mixed	Cousinia thomsonii, Pedicularis porrecta, Plantago himalaica, Lonicera obovata, Rosa webbiana, Rubus foliolosus, Viburnum cotinifolium	Bergenia stracheyi, Galium asperuloides, Lonicera obovata, Pinus wallichiana, Abies pindrow	Viola biflora, Polygonatum multiflorum, Fragaria vesca var. nubicola, Eragrostis minor, Dactylis glomerata, Cousinia thomsonii	Polygonatum multiflorum, Plantago himalaica, Pedicularis porrecta, Bergenia stracheyi
Fraxinus xanthoxyloides	Cremanthodium arnicoides, Stachys melissaefolia, Indigofera hebepetala, Lonicera spinosa, Fraxinus xanthoxyloides	Stachys melissaefolia, Indigofera hebepetala, Lonicera spinosa, Fraxinus xanthoxyloides	Artemisia gmelinii, Carum carvi, Chenopodium hybridum, Galinsoga parviflora, Origanum vulgare, Verbascum thapsus, Sorbaria tomentosa, Rosa webbiana	Carum carvi, Indigofera hebepetala, Lonicera spinosa, Fraxinus micrantha
Picea smithiana-Pinus wallichiana mixed	Chaerophyllum reflexum, Cremanthodium arnicoides, Impatiens glandulifera, Cotoneaster obtusus, Lonicera hypoleuca, Salix denticulata	Bergenia stracheyi, Bilderdykia pterocarpa, Impatiens glandulifera, Rubia cordifolia, Indigofera hebepetala, Syringa emodi, Viburnum cotinifolium, Picea smithiana	Adiantum venustum, Artemisia dracunculus, Asparagus filicinus, Fragaria nubicola, Podophyllum hexandrum, Viola pilosa, Jasminum humile, Ribes alpestre, Corylus jacquemontii	Asparagus filicinus, Bilderdykia pterocarpa, Dioscorea deltoidea, Podophyllum hexandrum, Polygonatum multiflorum, Juniperus polycarpos, Taxus baccata subsp. wallichiana
Cedrus deodara-Acer cappadocicum mixed	Anaphalis busua, Thalictrum cultratum, Plectranthus rugosus, Rosa webbiana, Spiraea canescens	Stachys melissaefolia, Lonicera spinosa, Spiraea canescens, Cedrus deodara, Celtis australis	Artemisia maritima, Carum carvi , Dioscorea deltoidea, Elsholtzia ciliata, Geranium pratense , Sonchus oleraceus, Thymus linearis, Plectranthus rugosus, Sorbaria tomentosa	Viola sylvatica, Lonicera spinosa, Rubus ellipticus, Spiraea canescens, Acer cappadocicum, Celtis australis
Hippophae salicifolia	Aster molliusculus, Astragalus rhizanthus, Heracleum thomsonii, Impatiens tingens, Nepeta nervosa, Nepeta laevigata, Silene edgeworthii, Synotis kunthiana	Astragalus rhizanthus, Heracleum thomsonii, Phlomis bracteosa, Physochlaena praealta, Silene indica, Thalictrum minus, Hippophae salicifolia	Artemisia parviflora, Achillea millefolium, Carum carvi, Chaerophyllum reflexum, Equisetum arvens, Phlomis bracteosa, Rumex acetosa, Silene indica, Sonchus wightianus, Scorzonera virgata, Taraxacum officinale, Urtica dioica, Hippophae rhamnoides	Carum carvi, Heracleum thomsonii, Heracleum candicans, Nepeta nervosa, Physochlaena praealta, Thalictrum minus, Hippophae rhamnoides

Discussion

Conservation prioritization of the habitats, species and communities is pre-requisite for the management planning of the biodiversity in protected and unprotected areas (Joshi and Samant, 2004). Therefore, present attempt has been made to prioritize the habitats and communities of the Lahaul valley in a proposed Cold Desert Biosphere Reserve. The habitat denotes the physical conditions that surround a species, or species population, or assemblage of species, or community (Clements and Shelford, 1939). Today, habitat destruction is a major factor in causing a species population to decrease, eventually leading to its being endangered, or even to its extinction. Due to sparse distribution of forest vegetation in Lahaul valley, the identified forest communities (14) were less than other Biosphere Reserves of the IHR (Samant et al., 2002). This may be due to the severe cold climatic conditions of the area. Mostly coniferous communities with wide range of distribution were recorded. Broad leaved communities were found in few patches and had narrow range of distribution. The loss of biodiversity is a growing concern throughout in the globe. The reasons are over exploitation and habitat destruction of the biodiversity elements. Therefore, protection of natural habitats is essential. Prioritization of habitats and communities supporting high species diversity and native, endemic, economically important and threatened species would help to some extent for the conservation of biodiversity.

Amongst the communities, Juniperus polycarpos, Pinus wallichiana, Cedrus deodara, Picea smithiana, Betula utilis and Abies pindrow showed high species richness, native, endemic, near-endemic, economically important and threatened species.

Significant positive correlations between species richness and native species ($r = 0.98$, $p < 0.01$, $n = 14$) (*Figure 3a*); species richness and endemic species ($r = 0.96$, $p < 0.01$, $n = 14$) (*Figure 3b*); and native and endemic species ($r = 0.97$, $p < 0.01$, $n = 14$) within the identified communities were found (*Figure 3c*). Like other parts of the IHR, in Lahaul valley, the percentage of native and endemic species increased with the altitude and species richness decreased. Regular monitoring of the habitats and populations of the native and endemic species facing high anthropogenic pressure is essentially required, so that adequate planning for their conservation management could be done intime.

In the IHR, most of the studies related to prioritization of species for conservation have been carried out using qualitative attributes/observations, only. Assessment of status of the species for prioritization using qualitative as well as quantitative attributes has been suggested by few workers (Joshi and Samant, 2004; Samant et al., 1996b; 1998a; 2001). Further, assessment status and values of the communities for conservation is urgently required (Joshi and Samant, 2004). In the present study, amongst habitats, forests (44), shady moist (38), respectively, and amongst communities, Abies pindrow - Pinus wallichiana mixed, Fraxinus xanthoxyloides, Picea smithiana - Pinus wallichiana mixed, Cedrus deodara - Acer cappadocicum mixed, Hippophae salicifolia, Juglans regia - Ulmus wallichiana - Acer acuminatum mixed, Juniperus polycarpos - Cedrus deodara mixed, Betula utilis and Salix daphnoides, respectively showed the high CPI, hence prioritized for conservation.
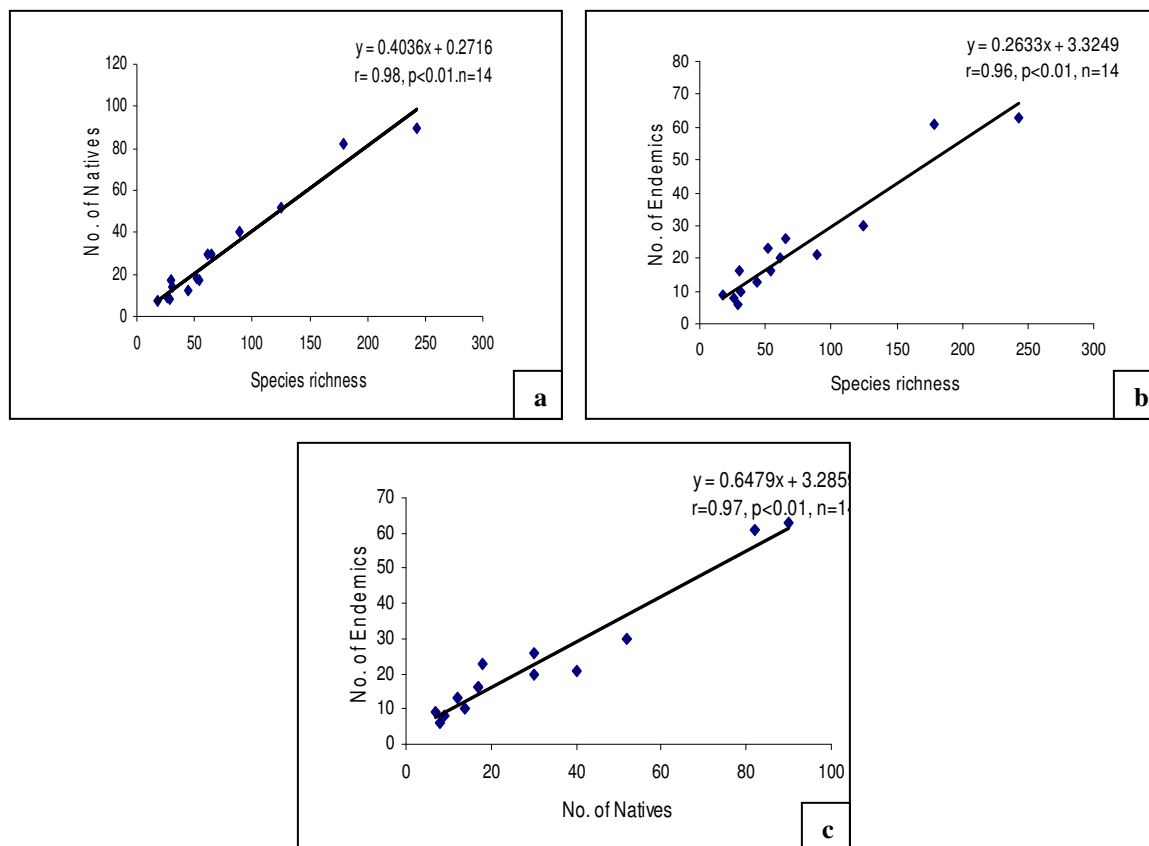
Figure 3. *Correlations between a) Species richness and Naives species; b) Species richness and Endemic species; and c) Native and Endemic species in Lahaul valley of the proposed CDBR*

These habitats and communities, requires regular monitoring, so that adequate management of these habitats and communities could be done intime. Some of the communities, such as Juniperus polycarpos, Pinus wallichiana, Cedrus deodara, Picea smithiana, Betula utilis and Abies pindrow showed wide range of distribution. However, typical topography, severe climatic conditions and high degree of anthropogenic pressure i.e., collection of fuel, fodder, timber, etc. and grazing by the sheeps of the nomadic shepherd (i.e., gaddies, gujjars and locals) have resulted in rapid loss of biodiversity elements of these communities. According to forest policy 1988 of India, the area under forest in hilly region should be 66% of its geographical area. The recorded area under actual forest cover during year 2003 was 180 km^2 in Lahaul and Spiti district out of total 13,835 km^2, which is too less (Gupta, 2007). Promotion of Afforestation programmes through plantation of native species may help in increasing the vegetation cover in the area.

Conclusion

The present study gives comprehensive information for the first time about the status of forests distributed under different habitats and communities in Lahaul valley a part of proposed CDBR. The recorded species richness under forest communities and habitats was

relatively less to other Biosphere Reserves of the IHR, may be due to adverse climatic conditions prevailing in the area. The variation in microclimate of an area showed great variability in species richness among different communities, habitats and aspects. The communities and habitats with wide range of distribution represented high species richness, high number of native, endemic, economically important and threatened species. Most communities showed comparatively less regeneration, indicative of great threat to these forests in future. Also, applicability of sustained forest management principles right from the plantation upto harvesting is lacking. As the area under cold desert is about one fourth part of total geographical area in Himachal Pradesh, there is a need to increase area under plantation. The conservation of broad-leaved communities is important for improving soil fertility status and to maintain the ecosystem conducive for regeneration establishment. The native and introduced fast growing hardy species should be planted through social forestry and agroforestry schemes to revive the habitats. Plantation of native and endemic species like *Abies pindrow*, *Fraxinus xanthoxyloides*, *Hippophae salicifolia*, *Pinus wallichiana*, *Cedrus deodara*, *Picea smithiana*, *Populus ciliata*, *Juglans regia*, *Corylus jacquemontii*, *Acer acuminatum* etc., is important to increase the forest cover of the area. *Hippophae salicifolia,* the most demanded species in market for edible and medicinal values, showed high density with less basal area. The pure stand communities like *Hippophae salicifolia* and *Juniperus polycarpos* need much attention for protection against fire, diseases, high anthropogenic and abiotic pressures. The presence of moderately high species richness, native, endemic and threatened species enhances the conservation value of the area. The regular monitoring of communities and habitats with high CPI is essentially required for their proper management. The communities located near habitations showed high anthropogenic pressure than that of distant communities. The protective measures of key stone species against adverse climate should be encouraged for conservation. In a nutshell, it is suggested that proper strategy and policy dealing with conservation management for prioritized communities and habitats should be formulated so that effective management of forests could be achieved in posterity.

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THE EFFECT OF THE REPRODUCTIVITY'S VELOCITY ON THE BIODIVERSITY OF A THEORETICAL ECOSYSTEM

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Abstract. Climate change has a great impact on the build and the work of natural ecosystems. Disappearance of some population or growth of the number in some species can be already caused by little change in temperature. A Theoretical Ecosystem Growth Model was investigated in order to examine the effects of various climate patterns on the ecological equilibrium. This paper examines the diversity of a theoretical ecosystem on the grounds of changing in temperature-climate patterns.
Simulations were made from the simplest case to the more complex exercise to explain harder questions. Daily temperature values were calculated according to various functions. The effect of existing climate patterns (historical or future daily temperatures) was analysed where temperature values were from various climatic zones. In this paper it is examined what kind of environmental conditions result larger diversity related to the velocity of reproduction.
Keywords: *analogous places; climate change; climate modelling; diversity; ecosystem.*

Introduction

There are several consequences of the decrease in biodiversity. The most scenic is the decrease in the number of species. Secondly the decrease in genetic diversity has to be mentioned, there are a lot of cases where stands of the frequent species decrease. At the third case the contents of ecosystems change also, the various habitats allow of being and maintenance of creatures between different geological and climatic conditions. This kind of role of ecosystems is less known (Nechay, 2002).

At natural ecological systems there is dinamic equilibrium between the climate and ecosystems. If the system is affected a response starts in order to maintain the equilibrium. The degree of this reaction could be a sudden response by leaps or on the other hand gradual. Some variables such as the phenological properties follow the changing climatic conditions simply; in these cases gradual shifts could be expected (Fitter et al., 2002). In case of sudden responses there is a good example in the maritime tidal zone where the community significantly alters under small-scale temperature

increase, which is caused by drastic decrease in the number of dominant predators (Sanford, 1999).

According to the forecasts the probability of extreme weather conditions will increase which increase further the occurence of sudden effects. There are some quick extreme events and the given sudden responses behind the events which seemed to have experienced gradual changes (Easterling et al., 2000).

In case of climate change this is not about the shift of the system being in equilibrium, but the succession could break or unhoped-for steps occur. In a mediterrenean scrub regenerated after a fire the number of species does not change under artificial drought-treating while the number of species increases in the control parcel fluently (Penuelas et al., 2007). The reaction of the run-down, degrading and regenerated communities for the climate change differs significantly from the reaction of natural ecosystems. These processes are important because there are much more of these areas like naturals (Czucz et al., 2007).

The hypothetic-centred approaches of researches there are three types:

- testing simple hypotheses with laboratory or field experiments (e.g. fitotron palnt growing room)
- analysing given ecosystems with tactical models (Sipkay et al., 2008; 2009; Vadadi-Fülöp et al., 2008a; 2008b; Eppich et al., 2009)
- examinations of general questions with strategic modeling (Drégelyi-Kiss and Hufnagel, 2009)

Some results have been reached in analysing the effect of some temperature-climate patterns on the production and community ecological relations in a strongly simplified theoretical model (Drégelyi-Kiss and Hufnagel 2009, 2010). The novelty of this modeling is characterized by a guild-specific approach at first (where competitive relationships can be manifested), on the other hand the population-dinamic model has been connected with the outputs of global circulation models. So this connection enables us to examine directly the effects of climate change. In this paper the various temperature effects are examined under changes in the diversity of a theoretical freshwater ecosystem:

- under constant temperature pattern with random fluctuations,
- under annual temperature fluctuations,
- under real, various climate temperature patterns (for historical and forecasted data).

Materials and methods

Theoretical Ecosystem Growth Model (TEGM)

An algae community consisted of 33 species in a terrestrial freshwater ecosystem is modelled (Drégelyi-Kiss and Hufnagel, 2009). Algae species are characterized by the temperature interval in which they are able to reproduce. There are four types of species based on their sensitivity: super-generalists (SG), generalists (G), transitional species (T) and specialists (S).

The reproductive feature depends only on their temperature sensitivity and a restrictive function has been built which is related to the access of sun (the period of the function is 365.25, the maximum place is on 23rd June and the minimum place is on 22nd December).

The examined temperature range corresponds to the temperature variation in the temperate zone. Thirty-three algae species with various temperature sensitivity can be seen in the *Fig. 1*. The daily reproductive rate of the species can be seen on the vertical axis, which means by how many times the number of specimens can increase on a given temperature. This corresponds to the reproductive ability of freshwater algae in the temperate zone (Reynolds, 2006; Sipkay et al., 2007).
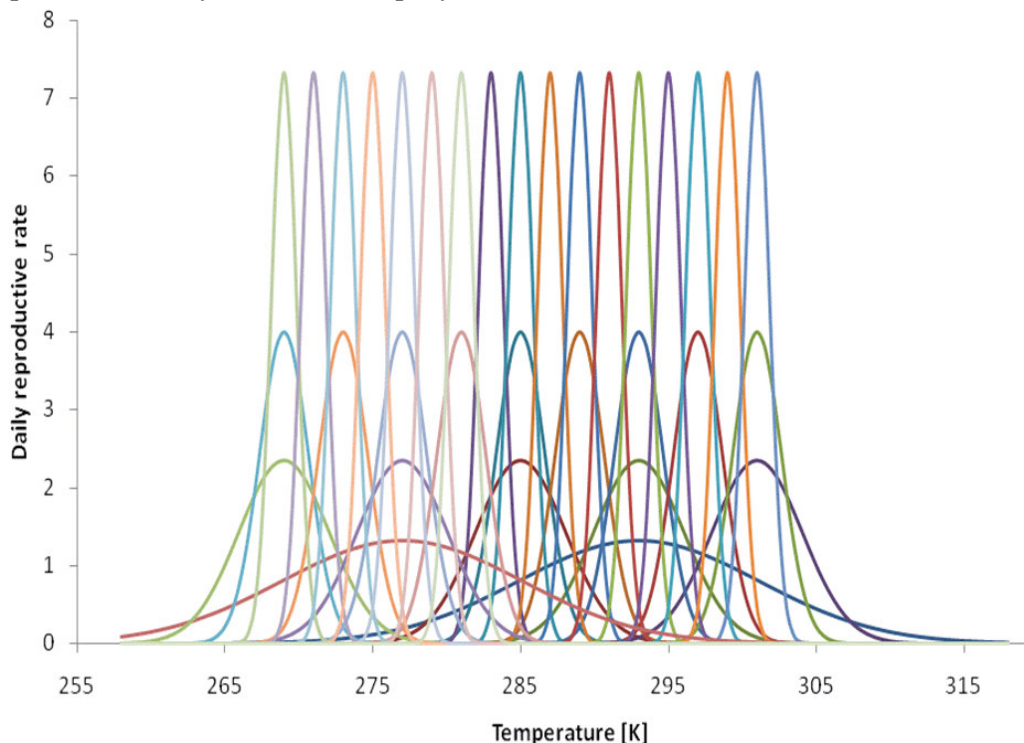

Figure 1. Reproductive temperature pattern of 33 algae species

Among the parameters of the TEGM model a velocity parameter exists, the velocity of the reproductivity of the ecosystem can be set up. During our experiments this value has been examined in two level (r = 1 in case of faster, r = 0.1 in case of lower reproducibility).

In our experiments the distribution of the algae community of a theoretical freshwater ecosystem has been examined by changing the temperature. Competition begins among the species when the temperature changes. In each temperature interval there are dominant species, which win the competition. The temperature was changed systematically in order to be able to estimate the various effects separately. The duration of the simulation was usually 10 years in the experiments. Measuring the diversity the Shannon diversity function was used (Hammer et al., 2009; Jost, 2006; Tóthmérész, 1997; 2001).

Functions of temperature patterns

The daily values of the temperature were determined in several ways. At first the effect of the constant temperature pattern is examined on the composition of the theoretical ecosystem, then the consequences of the annual temperature fluctuation modelled by sine function. Finally, the effects of existing climate patterns is calculated.

The stream of the daily temperature average values is cyclic therefore the used temperature data series is described by appropriate functions.

Constant temperature

Simulation experiments were made on 293 K, 294 K and 295 K with the two velocity parameters (r = 1 and 0.1). Fluctuation was added as ±1…±11 K random numbers.

Sine temperature pattern in a year

Annual average temperature of Budapest is about 283-284 K (between 1960 and 1990), the range of temperature data is between 30 K and 45 K. The following function describes this temperature pattern (with a day period of 365.25):

$$T = s_1 \cdot \sin(s_2 \cdot t + s_3) + s_4 \qquad (Eq.1)$$

where s_2=0.0172, s_3=-1.4045. During the experiments the height (s_1) and the place (s_4) of the sine function can be modified. The s_1 value is between 15 K and 22.5 K and s_4 is equal to 284 K according to the Budapest historical data.

Existing climate patterns (historical and future)

One of the parameters used for describing the climate is the daily average temperature value. This means a great deal of numbers for long term (30 years) examinations: 30x365 values. In order to simplify the description of the climate and to ignore the fluctuation between years a function has been fitted to averages and dispersion of the 30 years' data according to the least squares method. It is found that the daily average temperature from various climate zones fluctuates according to the following function (deterministic term):

$$T(t) = a_1 \sin(a_2 \cdot t + a_3) + a_4 + a_5 \sin(a_6 \cdot t + a_7) \qquad (Eq.2)$$

The standard deviation of the daily average temperature during the 30 years can be described by the next function (stochastic term):

$$s(t) = b_1 \sin(b_2 \cdot t + b_3) + b_4 \qquad (Eq.3)$$

The daily average temperature values can be classified according to their origin: historical or future. For these data series daily temperature values can be generated with the following expression:

$$Generated(t) = average(t) + (3,5 \cdot Rnd(\ ) - 1,75) \cdot s(t) \qquad (Eq.4)$$

where t is the number of the year, *Rnd()* is a random value between 0 and 1.

The classification of the used existing climate patterns is the following:
- Historical temperature values from 1960 to 1990 in Hungary (Budapest)
- Historical international data from various climate zones (according to Köppen-Geiger classification (Peel et al., 2007))
  Daily average temperature data were collected with the help of web database (Klein Tank et al., 2002) from the following cities all over the world. (In some meteorological stations there were not any available historical data for 30 years, only from 1995 till 2008)
  - i. Tropical climate (Bangui 4°22' N 18°35' E, Central African Republic)
  - ii. Dry climate (Taskent 35° 16' 15" N, 33° 23' 30" E, Cyprus)
  - iii. Temperate climate (Den Helder 51° 25' 60" N, 4° 31' 60" E, The Netherlands)
  - iv. Continental climate (Ulan Bator 47° 55' 0" N, 106° 55' 0" E, Mongolia)
  - v. Polar climate (Sodankyla, 67°25'00" N, 26°35'35" E, Finland)

A place has been chosen from the main climate zones and the change in the composition of our theoretical ecosystem has been examined.

- Future temperature patterns from 2070 to 2100 in Hungary
  - i.   Hadley Center adhfa (Regional Model 3, SRES A2)
  - ii.   Hadley Center adhfd (Regional Model 3, SRES B2)
  - iii. MPI 3009
- Analogous places in Hungary by 2100 (Hufnagel et al., 2008)

  It is predicted that the climate in Hungary will become the same which is the present day climate on the border of Romania and Bulgaria or near Thessaloniki. According to the worst prediction the climate will be like the current North-African climate. Therefore the following places have been examined:
  - i.   Turnu Magurele, Romania (43.75° N, 24.88° E, 31.0 m)
  - ii.   Cairo, Egypt (30.058° N, 31.229° E)

  Scenarios of analog places are simple because temperature is the only factor which has been considered.

Results

Constant temperature

There is larger change in diversity in case of lower velocity parameter while smaller change in the other case which has higher velocity parameter (*Table 1*). The diversity value of the ecosystem is strongly affected by the setting temperature value and the random fluctuation. The largest diversity value has been reached when the temperature is not an optimum of a genus for reproduction (294 K).

With the increase of fluctuation the diversity value becomes a low value, but there is one order of magnitude difference in diversity between the processes come from the slower and the faster reproduce-able ecosystems.

In case of largest fluctuation (± 11K) the degree of the diversity increases significantly.

Table 1. *Shannon diversity values in case of constant temperature patterns and various random fluctuations*

Fluctuation [K]	r = 1			r = 0.1		
	293 K	294 K	295 K	293 K	294 K	295 K
0	0.0244	0.8860	0.0002	0.0569	1.0080	0.0010
±1	0.0429	0.0001	0.0001	0.1180	0.0285	0.0003
±2	0.0001	0.0000	0.0000	0.0302	0.4224	0.0034
±4	0.0000	0.0000	0.0000	0.0000	0.0000	0.6575
±7	0.0000	0.0000	0.0000	0.0000	0.0000	0.0000
±11	1.4920	1.6670	1.2210	0.9243	1.1530	2.7620

Sine temperature pattern in a year

The experiments – in case of $r = 1$ – were run until that degree of fluctuation where the annual average use of resources was higher than 0.1 %. With the increase of random fluctuation the annual total number of specimens and the composition of species in consecutive years better and better differ from each other.

In case of $s_1 = 0$ with $r = 0.1$ velocity factor (which is equal to the 284K constant temperature) the species have already disappeared at ± 7K random fluctuation. This was

not observed under 293-295 K constant temperature, where this has occurred at ±11K random fluctuation.

It can be seen in *Fig. 2* and *3* how the annual total number of specimens changes in function of daily random (0…±7K) and the annual fluctuation (s_1, the amplitude of sine function). The two fluctuations result that the number of specimens and the use of resources is low already at ±7K random fluctuation.

The maximum value in annual total number of specimens is reached in case of $r = 1$ velocity parameter and $s_1 > 15$. The maximum annual total number of specimens related to the $r = 0.1$ process is reached at low annual fluctuation ($s_1 = 0, 2$).

Regarding the diversity of the ecosystem there are opposite trends in case of ecosystems which have different velocity parameters. It can be seen in Fig. 4 and 5 that the ecosystem which has faster reproduction has low diversity values at constant temperature and with $s_1 = 2$ parameter except for the case which has no daily random fluctuation. The values of Shannon diversity in case of greater annual random fluctuation (with parameters $s_1 > 5$) are in inverse proportion with the increase of daily random fluctuation.

In case of ecosystems which have slower reproduction the diversity is in direct proportion with the function of random daily fluctuation after a short decrease. The diversity values are smaller in case of ecosystems which have $r = 1$ parameter.
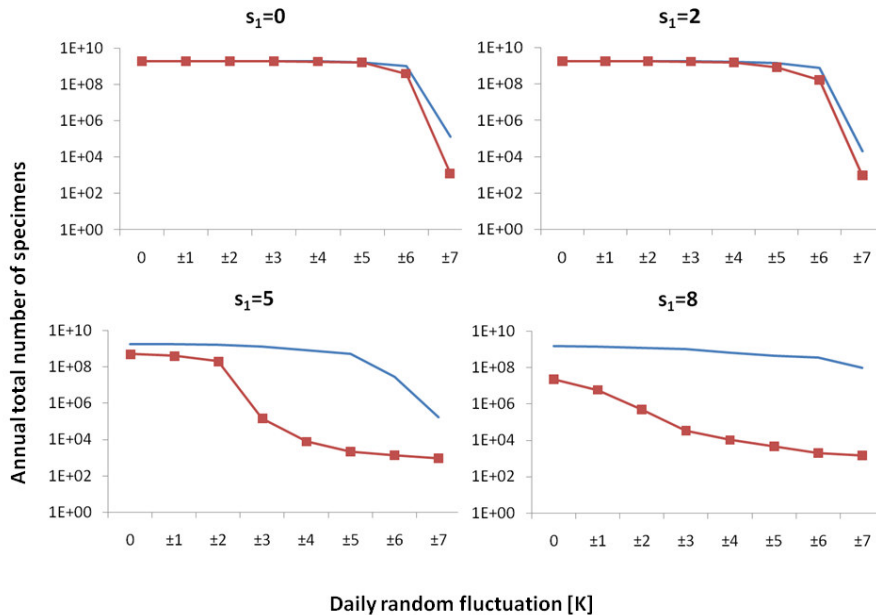

Figure 2. Annual total number of specimens versus the daily random fluctuation in sine temperature environment for <u>low</u> s_1 parameters (The square-signed data series show the processes with r=0.1 velocity parameter and the non-signed is related to the other setting, r=1)

Figure 3. *Annual total number of specimens versus the daily random fluctuation in sine temperature environment for <u>high</u> s₁ parameters (The square-signed data series show the processes with r=0.1 velocity parameter and the non-signed is related to the other setting, r=1)*
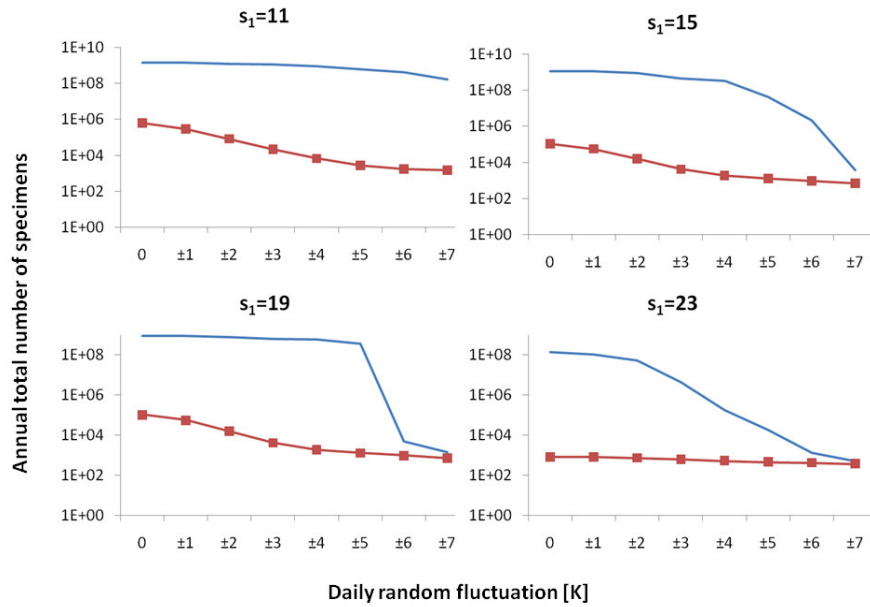

Figure 4. *Shannon diversity values versus the daily random fluctuation in sine temperature environment for <u>low</u> s₁ parameters (The square-signed data series show the processes with r=0.1 velocity parameter and the non-signed is related to the other setting, r=1)*

Figure 5. *Shannon diversity values versus the daily random fluctuation in sine temperature environment for* <u>high</u> *s_1 parameters (The square-signed data series show the processes with r=0.1 velocity parameter and the non-signed is related to the other setting, r=1)*
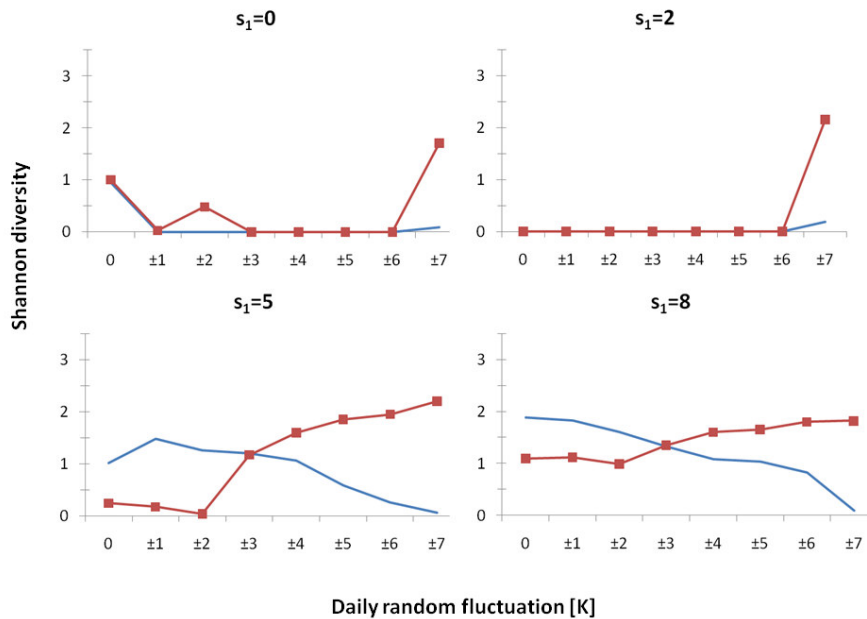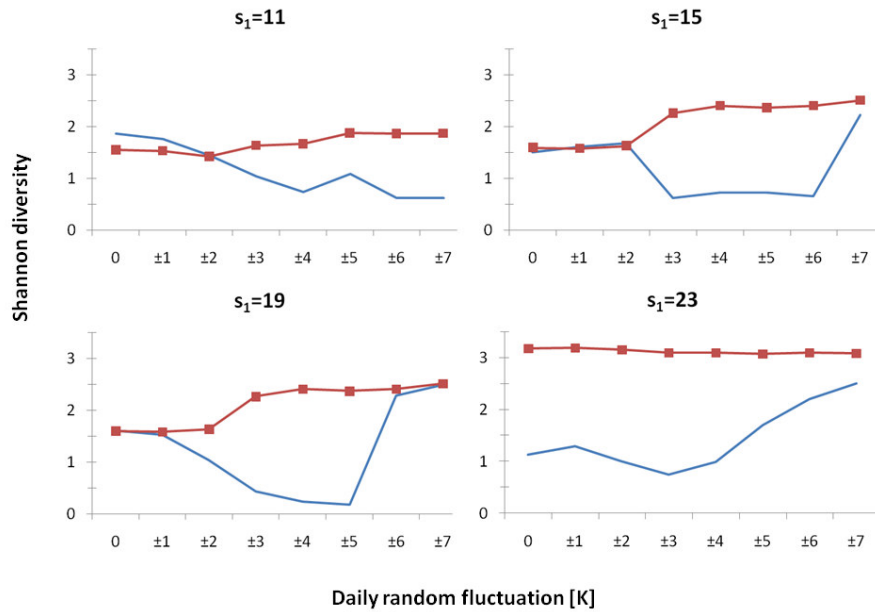
Existing climate patterns

The annual total number of specimens and the diversity values for various climate conditions are presented in *Fig. 6*. The simulation has been repeated 10 times, therefore the averages of the characteristics and the calculated deviation for the diversity values can be seen in graph.

Regarding the annual total number of specimens it is stated that the species are in maximum numbers in case of ecosystems which has faster reproduction. There is only one climate condition (Bangui) where the maximum value in the number of specimens has been reached in case of slower reproduction. The difference between the annual total number of specimens in cases $r = 1$ and $r = 0.1$ is the largest at climate pattern of Ulan Bator.

The reproduction of the simulated experiments is sufficient in most cases.

The diversities are between 1.5 and 2.5 in case of $r = 1$ velocity parameter except for the case of Bangui which has 0 diversity value.
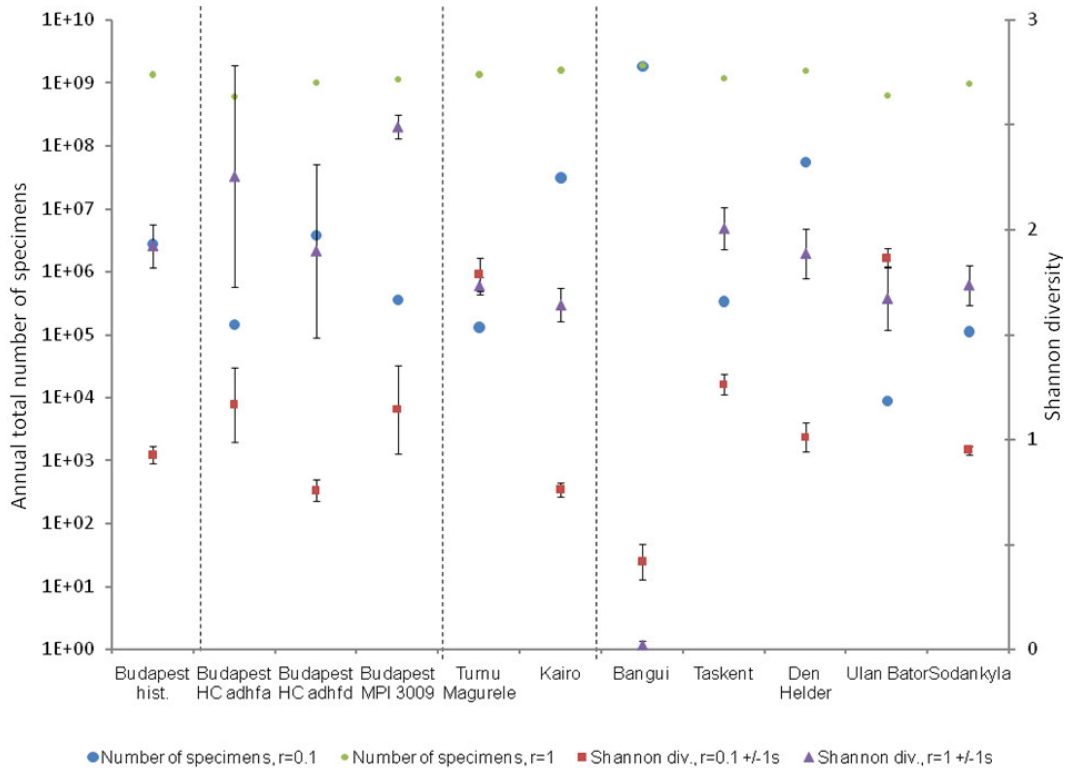
Figure 6. *The average of annual total number of specimens and diversity values in case of various climate conditions (10 repeated simulations, ±1s are shown regarding the diversity values)*

Discussion

Initially the community tries to adapt to its environment and many genre compete for the environmental elements with large number of specimens. In case of large noise the diversity is large and the annual total number of specimens is low, because a few genres could adapt to the environmental conditions.

Analysing peaks evolved at constant temperature patterns and with $r = 0.1$ velocity parameter it is stated, that the largest biodiversity values exist in case of medium disturbance, the Intermediate Disturbance Hypothesis (Connell, 1978; Elliott et al., 2001) could be observed.

In case of sine temperature patterns there is a strong connection between the number of specimens and the velocity parameters hence $r = 0.1$-characterized ecosystems have low number of specimens. There is huge difference between the number of specimens values for cases $r = 1$ and $r = 0.1$. In ecosystems having $r = 0.1$ velocity parameter there are larger diversity and smaller number of specimens, namely the specimens of the community belong to different genus.

Irrespectively of velocity parameter the largest number of specimens is in case of the smallest amplitude ($s_1 = 2$) and the diversity is small, namely a few genre occur with large number of specimens. This could be observed similarly in tropical wet climate, Bangui, since the temperature fluctuation is small at the given climate. The genres which could adapted to the environment suppress the new arrivals, the principle of

competitive exclusion prevails. Increasing the noise the biodiversity begins to grow, because the above-mentioned principle could not prevail here, since new genres occur continually.

Regarding the number of specimens the change is smaller in cases with $r = 0.1$ velocity parameter – except for the temperature of Bangui. In diversity there are higher values in case of Ulan Bator and Turnu Magurele with $r = 1$ than in case of lower reproduction.

In point of diversity functions there are lower values in case if $r = 0.1$ parameter, while there are higher values with faster reproduction. There are a few exceptions such as Bangui (tropical wet climate), Ulan Bator (continental climate) and Turnu Magurele (analogous place with Hungary, Rumanian lowland).

Examining the responses of the real temperature patterns it is stated, that Bangui composes a separate group, it does not show any similarity with the other climatic environments. This is because Bangui lays on tropics, where nearly same temperature exists through the year, so the dominant species are able to reproduce.

During the various climate conditions the decrease in the number of specimens is reached earlier in case of slower reproduction than in the other case, and there are larger changes in diversity values. Generally it could be said that an ecosystem evolves finally with low number of specimens.

By the real climate functions it is stated, that from the predicted analogous places (Turnu Magurele, Romania; Cairo, Egypt, (Hufnagel et al., 2008)) Budapest shows similarity with Turnu Magurele in number of specimens and diversity values.

The specimens of the ecosystems do not only suffer the change in climate but they could affect the equilibrium of the biosphere or the composition of the air through the biogeochemical cycles. There is an opportunity to examine the controlling ability of temperature-climate with the theoretical ecosystem.

In our further research we would like to examine the effect of the ecosystem back to the climate (Dregelyi-Kiss et al., 2008). These temperature feedbacks have got a great emphasis related to DGVM models with large computation skills (Friedlingstein et al., 2006), but the feedbacks are not estimated directly. We would like to examine the process of the feedback with PC calculations to answer easy questions.

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Koutika – Rainey: *Chromolaena odorata* in different ecosystems: weed or fallow plant?

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CHROMOLAENA ODORATA IN DIFFERENT ECOSYSTEMS: WEED OR FALLOW PLANT?

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Abstract. To understand the use of *Chromolaena odorata* in different agricultural systems and ecosystems, findings of several scientific studies conducted in different areas have been assessed in this review paper. Some authors considered *C. odorata* as a serious weed because of its ability: to regenerate and colonize uninvaded areas; to be a threat to some ecosystems and environment; to reduce the biodiversity of grasslands, savannahs and forests; and to be a considerable problem in commercial tree plantations as it suppresses the growth of young pine and eucalypt trees. Others argued that the species may be considered as a beneficial fallow plant rather than a weed, because it may be considered as a welcome plant rather than a weed in some agricultural systems, when considering the expected properties of species for fallow improvement. The following are the main reasons why *C. odorata* is considered as a fallow because of it ability: to be a nutrient sink and its potential benefit to the crop as regular source of organic matter and nutrients after slashing; to have a beneficial effect on exchangeable K concentration; to be used as green manure; to be better adapted as a fallow plant on acidic soils than some leguminous.
Keywords: *tropical and subtropical ecosystems, invasive plant.*

Introduction

Chromolaena odorata (L), King & Robinson (*Asteraceae, Eupatorieae*) is a perennial shrub forming dense tangled bushes 1.5-2.0 m in height, occasionally reaching 6 m as a scrambler up trees. *C. odorata* originated in subtropical and tropical America, and had a wide native distribution, from the southern United States to northern Argentina, and exhibits considerable variation throughout its distribution (Holm et al., 1977). MacFadyen Cruttwell (1996) argued that the source of the original introduction is likely to have been Jamaïca in the West Indies. In the Neotropics *C. odorata* is always found in competition or association with a complex of closely related species not present in Asia or Africa (MacFadyen Cruttwell, 1991). In the new world, *C. odorata* is common in most habitats except undisturbed rainforest. It is typically a plant of secondary succession, rapidly invading clearings and persisting until shaded out by the overgrowth of forest trees (MacFadyen Cruttwell, 1991). Almeida-Neto and Lewinsohn (2004) investigated the spatial distribution of *C. odorata*, in its native habitat in the Reserva Biologica e Estaçao Experimental de Mogi-Guaçu, Sao Paulo State, Brazil. The authors concluded that the spatial autocorrelation in phenologial traits between *C. odorata* neighbours may be driven by genetic similarity among neighbouring plants and/or spatial structuring of environmental factors.

Nowadays, this shrub is widespread in subtropical and tropical areas all over the world due to its fast invasion or colonisation and its facility to regenerate; *C. odorata* regenerates and colonises equally as well through its roots or by high seed production

and it is also well dispersed by wind. *C. odorata* is present in different agricultural systems of its native continent (MacFadyen Cruttwell and Skarratt, 1996), but also of colonized continents such as Asia (Nemoto et al., 1983 ; Roder et al., 1995), Africa, mainly in slash-and-burn systems (Olaoye, 1986; Rouw de, 1991; Obatulu and Agboola, 1993) and Oceania (Michael, 1989; MacFadyen Cruttwell, 1989). *C. odorata* was introduced into Asia in the 1840s, probably through the Serampore Botanical Gardens at Calcutta (MacFadyen Cruttwell, 1989; MacFadyen Cruttwell and Skarratt, 1996). From Calcutta, *C. odorata* spread east into Burma, and then progressively through Indonesia and Indochina (MacFadyen Cruttwell, 1989). *C. odorata* presence was reported in the late 1920s in Laos (Vidal, 1960) and in 1922 in Thailand (Kerr, 1932), and is wide spread in other Asian countries (Nakano, 1978, Roder et al., 1995). *C. odorata* was introduced into West Africa in 1936-1937 as seeds mixed with seeds of *Gmelina arborea* in Enugu (Nigeria) from Sri Lanka (Odukwe, 1965). Using the enzymatic diversity in *C. odorata*, Lanaud et al. (1991) supported the theory that *C. odorata* was introduced to Africa from Asia. From West Africa, C. *odorata* has since spread to the Central Africa, i.e., Cameroon, Central African Republic, Democratic Republic of Congo and Republic of Congo in the northern and southern part of the country (Prasad et al., 1996). MacFadyen Cruttwell and Skarratt (1996) have established a dynamic simulation model to estimate of *C. odorata's* geographical distribution and relative abundance as determined by climate, and found that the potential distribution coincided with the area invaded and it has reached its apparent climatic limits in West Africa. *C. odorata* was introduced in 1940 in South Africa, and has naturalized and invaded most of tropical and sub-tropical regions of the country (Delfino Abeya, 2002). *C. odorata* spread rapidly towards Australia and the Pacific region. In 1985, Australian weed scientists started a campaign as the species was identified as the greatest weed threat to the northern part of country, because of its rapid spread and its high potential to damage agriculture and the environment (Michael, 1989). Using climatic comparisons, MacFadyen Cruttwell (1989) found that much of the tropical and subtropical coastal land from the Kimberley's in Western Australia to Brisbane, was suitable for invasion by this species.

As shown in *Table 1*, the use of *C. odorata* is somewhere controversial (Roder et al., 1995; Weise and Tchamou, 1999). Some researchers considered *C. odorata* as a weed (Olaoye, 1986; Rouw de, 1995; Waterhouse, 1994; MacFadyen Cruttwell, 1996) or a beneficial fallow plant (Ekeleme et al., 2004; Norgrove et al., 2000; Koutika et al., 2005a). Therefore, the aim of this review paper is to present selected research and findings which considered *C. odorata* as a weed or as a beneficial fallow plant and to show how would its use affect any decision on classical biological control and environment management. On balance, *C. odorata* is more a weed or a beneficial plant?

C. odorata as a weed

In this part of the review we make a summary of the factors that would allow a farmer to decide to eliminate the species, according to crop, soil or climate. *Chromolaena odorata* is an aggressive pioneer shrub species, which is regarded as a very serious threat to agriculture and the environment in most invaded countries. The high productivity of light seeds allows the species to invade disturbed sites in a short period of time (Holm et al., 1977; Swaine et al., 1997). The rapid spread of the weed is due to extensive seed production which is estimated to be 93,000-160,000 seeds/plant

(Wilson, 1995), and also through wind dispersal of seeds (MacDonald and Frame, 1988). There are other specific characteristics such as quick germination of plants, which grow to 2 m and form a dense woven canopy that is almost impossible for tree seedling to grow through (Riddock et al., 1991); the understory only receives about 70% of full sunlight (Honu and Dang, 2000). All these characteristics have caused many researchers to recognize *C. odorata* as a serious weed in countries where it grows (Olaoye, 1986; Michael, 1989; Roder et al., 1994; MacFadyen Cruttwell and Skarratt, 1996).

In addition, *C. odorata* is considered as an alien invasive species which negatively impacts the forest in economic, ecological and environmental and social and health terms (Holm et al., 1977; Moore, 2004). It is also considered as a considerable threat to conservation and ecotourism, as it has first invaded natural area, reducing the biodiversity of grasslands, savannahs and forests (MacDonald, 1983; Goodall and Erasmus, 1996; Matthews and Brand, 2004). *C. odorata* has been found to threaten the breeding habitat of the Nile crocodile *Crocodylus niloticus* in South Africa, by decreasing the temperature of nesting sites by shading and crowding and this can induce female biased sex ratios or may even prevent embryotic development altogether (Leslie and Spolita, 2001). *C. odorata* is a considerable problem in commercial tree plantations as it suppresses the growth of young pine and eucalypt trees and allows fire and to penetrate deeper into plantations (Matthews, 2004; Matthews and Brand, 2004). It can also promote wildland fires (Moore, 2004). *C. odorata* may also cause skin problems and asthma in allergy-prone people.

Kriticos et al. (2005) revised a climate model of the estimated potential distribution of *C. odorata*. The authors argued that the revised model reduces the estimated potential distribution of *C. odorata*, particularly in terms of the poleward and inland extents of suitable climates. This model shows that Mediterranean, semi-arid and temperate climates are predicted to be unsuitable, and supports the previous conclusions that much of tropical Africa, the north-eastern coast of Australia and most Pacific islands are at risks of invasion. Joshi et al. (2006) investigated the possibility of indirect mapping techniques to localize areas, where *C. odorata* was capable of expressing its invasive traits. The authors found that *C. odorata* was frequently observed in forest too shaded for seed production and argued that the presence of *C. odorata* under conditions unfavourable for seed production implies continuous re-colonization of such sites from adjacent seed production populations.

In South Africa, Kluge and Zachariades (2006) found *Lixus aemulus*, a stem-boring weevil imported from Brazil caused 66.6% mortality of the stems and reduced the dry mass of infested *C. odorata* stems by 46.6% in a damage trial in pots. Van Gils et al. (2005) argued that promoting a closed tree canopy cover itself will not be sufficient to control *C. odorata* at Mtubatuba municipability of 20 km (west to east) and 10 km (north to south) in South Africa. While studied allelopathic effects of *C. odorata* (R.M. King and Robinson) toxin on tomatoes (*Lycopersicum esculentum* Mill), Onwugbuta-Enyi (2001) found significant growth reduction of the latter species with *C. odorata* aqueous-leaf extract at concentrations as low as 1g fresh weight in 40 ml of water. The author concluded that the presence of *C. odorata* in a field may result in the accumulation of leachates and residues that are toxic to crops resulting in yield losses.

Similarly, Ambica and Jayachandra (1980) argued that *C. odorata* possesses allelopathic potentialities and growth inhibitors. Prashanti and Kulkarni (2005) proposed that *C. odorata* is a threat to agriculture and environment. They reported that

there is an urgent need to manage weed growth and its spread so as to maintain ecological integrity of habitats. It was found that *C. odorata* could be controlled by the introduction of natural enemies. The genetically homogeneous population structure in weeds makes it easier to match a biological control agent to host genotypes and makes this species particularly vulnerable to biological enemies (Van Driesche and Bellows, 1996; Muniappan and Marutani, 1991) argued that mechanical control is labor intensive and not long lasting; cultural control is long lasting, however either mechanical or chemical control program has to be carried out initially to implement the cultural control; while chemical is effective, but expensive, and poses some environmental problems. Ye et al. (2004) studied a genetic structure of the invasive *C. odorata* in China. The authors argued that because the genetic variation of *C. odorata* is very low, if a suitable biological enemy was identified effective for control, the various populations of *C. odorata* in China could have a uniform response to this biological control.

In the upland rice fields of some areas of Laos, *C. odorata* is the most important weed. However, the interviewed elderly persons could not recollect the dominant weed species before the *C. odorata* invasion (Roder et al., 1994). In Ghana, poor natural forest regeneration is due to the invasion of deforested areas by *C. odorata* (Honu and Dang, 2000). However, Honu and Dang (2000) argued that tree seedlings were observed under *C. odorata* canopy and these seedlings positively responded to the removal of the shrub. These findings indicated that release treatment may be an effective way to restore forest to sites previously occupied by *C. odorata*. Chandrasekaram and Swamy (2002) estimated the biomass, litter production and aboveground net production of herbaceous community and analysed vegetation composition in natural and man-modified ecosystems. The authors found that exotic plant invasions such as those by *C. odorata* influence ecosystem structure, species composition and ANPP, and argued that the sustainable use of these weeds (extracted from plantations) as an organic mulch in the establishment of young banana plantation, needs to be encouraged to manage its effectiveness. In Southern China, establishment of signal grass (*Brachiaria decumbens* Stapf*)* in pasture has effectively prevented germination of the seeds and growth of the seedlings of *C. odorata* (Renrun and Xuejun, 1991). They found that a few seedlings of *C. odorata* may grow in the first two years after establishment of *Brachiaria decumbens*. However in the third year virtually no C. odorata is noted in these fields.

Some farmers did not consider *C. odorata* as an aggressive shrub species. Thus, some farmers of Laos found it relatively easy to remove *C. odorata* from the young plants when it regrew from rootstock after burning. They said that *C. odorata* growing from seeds had a comparatively low initial growth phase and was not a big problem in the initial growth stage of the rice plant (Roder et al., 1995). Furthermore, even though *C. odorata* accounted for 48% of total biomass after the first year of fallow against 37% fallow tree and bamboo species, plant density and percent contribution declined in the second year of fallow in northern Laos (Roder et al., 1995). Similarly, *C. odorata* contributed 80% of the canopy cover and was the dominant fallow species (Roder et al., 1995). All the findings cited have shown that *C. odorata* is a weed and that classical biological control is used to its iradication.

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C. odorata as a beneficial fallow plant

In this second section, we made a summary of the factors that would allow a farmer to decide to grow the species, according to crop, soil or climate. The fallow period is used to limit the development of weeds because they threaten the re-use of the land (Rouw de, 1995). When considering the expected properties of species for fallow improvement such as ease of establishment, large biomass, fast decomposition rate, weed suppression, *C. odorata* may be considered as a welcome plant rather than a weed in some slash-and-burn systems (Dove, 1986; Ikuenobe and Analiefo, 2003; Koutika et al., 2005a; Norgrove, 2007). Ikuenobe and Analiefo (2003) argued that infestation of weeds was lower in plots cropped after *C. odorata* than in the modified natural bush fallow in Nigeria. *C. odorata* grows dense canopy cover in a short space of time and was able to suppress other plant growth. Moreover, no correlation has been found between period and selected soil fertility parameters and frequency of *C. odorata* (Roder et al., 1995). Nevertheless 85% of farmers appreciated *C. odorata* and qualified it as a 'good fallow plants' or plant they like to have in their fallow fields in northern Laos (Roder et al., 1995).

These authors argued that various explanations might be given for the preference of *C. odorata*: absence of negative effects on rice yield; relative ease of control by hand weeding in the rice crop; and fast growth and large biomass production. Some farmers suggested that soil structure improved where *C. odorata* is dominant compared to the fields with dominant bamboo species (Roder et al., 1994). Ngobo et al. (2004) argued that frequently farmed *C. odorata*-dominated short fallows were also characterized by higher litter and crown cover but lower tree basal area and canopy height, compared with bush fallow that had been a forest in the precedent cropping cycle. In addition, Ngobo et al. (2004) found that weed infestations were less rampant on plots planted following a *C. odorata* dominated fallow than on fields established following a short fallow not dominated by *C. odorata*. Being a dominant weed and fallow species in slash-and-burn agriculture of humid forest zone in Cameroon, Ngobo et al. (2004) argued that, as for the findings of Roder et al. (1995) and Ikuenobe and Analiefo (2003), some of the properties that render *C. odorata* as a 'good' fallow plant: namely fast development during the cropping phase, thus provide protective cover and allow better weed suppression than fallow systems not dominated by *C. odorata*.

Koutika et al. (2002) argued that *C. odorata* performed better than *Pueraria* leguminous fallow in acidic soils of southern Cameroon. Furthermore, Koutika et al. (2005a) found that *C. odorata* is more adapted to acidic soils than both *Pueraria* and *Calliandra calothyrsus* in central southern Cameroon, *C. calothyrsus* more adapted to nonacidic soils. However, in some situations of nonacidic soils, leguminous *Pueraria* may perform better or similar to *C. odorata*. Koutika et al. (2005b) argued that *C. odorata* is a good fallow option in the humid forest zone according to soil acidity and nutrient concentrations, while Kanmegne et al. (1999) also argued that *C. odorata* lead to an improvement of soil properties. In central-southern Cameroon, Kanmegne et al. (1999) showed that *C. odorata* had a beneficial effect on exchangeable K concentration on a sandy soil developed on granites and on a sandy clayey soil developed on gneiss, by comparing natural fallow dominanted by *C. odorata* and fallow where *C. odorata* had been removed by hand.

Table 1. *Chromolaena odorata as a weed or a fallow plant according to different researches*

References	Localities	As a weed	As a fallow
[1] Almeida-Neto & Lewinsohn (2004)	Brazil	*	
[2] Ambica & Jayachandra (1980)	-	*	
[3] Chandrasekaram & Swamy (2002)	South India	*	
[4] Delfino Abeya (2002)	South Africa	*	
[5] Dove (1986)	Indonesia	*	
[6] Ekeleme et al. (2004)	Nigeria	*	
[7] Fuambeng Yonghachea et al. (2005)	Cameroon	*	
[8] Goodall & Erasmus (1996)	South Africa	*	
[9] Holm et al. (1977)	Tropic&Subtropic	*	
[10] Honu & Dang (2000)	Ghana	*	
[11] Ikuenobe & Analiefo (2003)	Nigeria		*
[12] Joshi et al. (2006)	Central Nepal	*	
[13] Kerr (1932)	India, Birmania	*	
[14] Kanmegne et al. (1999)	Cameroon		*
[15] Kluge & Zachariades (2006)	South Africa	*	
[16] Koutika et al. (2001)	Cameroon		*
[17] Koutika et al. (2002)	Cameroon		*
[18] Koutika et al. (2004)	Cameroon		*
[19] Koutika et al. (2005a)	Cameroon		*
[20] Koutika et al. (2005b)	Cameroon		*
[21] Koutika & Meuteum-Kamga (2006)	Cameroon		*
[22] Kriticos et al. (2005)	Tropic&Subtropic	*	
[23] Lanaud et al. (1991)	Africa	*	
[24] Litzenburgen & Ho Tong Lip (1961)	Cambodia		*
[25] Leslie and Spotila (2001)	South Africa	*	
[26] MacDonald (1983)	South Africa	*	
[27] MacDonald & Frame (1988)	South Africa	*	
[28] Matthews (2004)	Asian countries	*	
[29] Matthews & Brand (2004)	African countries	*	
[30] McFadyen Cruttwell (1989)	Australia	*	
[31] McFadyen Cruttwell (1991)	Australia	*	
[32] McFadyen Cruttwell (1996)	Australia	*	
[33] McFadyen Cruttwell & Skarratt (1996)	Tropic&Subtropic	*	
[34] Michael (1989)	Australia	*	
[35] Moore (2005)	Tropic&Subtropic	*	
[36] Muniappan & Marutani (1991)	Tropic&Subtropic	*	
[37] Nakano (1978)	Thailand	*	
[38] Nemoto et al. (1983)	Northeast Thailand	*	
[39] Ngobo et al. (2004)	Cameroon		*
[40] Norgrove et al. (2000)	Cameroon		*
[41] Norgrove et al. (2007)	Cameroon		*
[42] Obatolu & Agboola (1993)	Nigeria		*
[43] Odukwe (1965)	Nigeria	*	
[44] Olaoye Soa (1986)	Nigeria	*	
[45] Onwugbuta- Enyi (2001)	Nigeria	*	
[46] Prashanthi & Kulkarni (2005)	India	*	
[47] Prasad et al. (1996)	India	*	
[48] Renrun & Xuejun (1991)	China	*	
[49] Riddock et al. (1991)	Nigeria	*	
[50] Roder et al. (1994)	Laos	*	
[51] Roder et al. (1995	Northern Laos	*	
[52] Rouw de (1991)	Ivory Coast		*
[53] Rouw de (1995)	Tropic&Subtropic		*
[54] Swaine et al. (1997)	Ghana	*	
[55] Tian et al. (1998)	Nigeria		*
[56] Tian et al. (2005)	Nigeria		*
[57] Vidal (1960)	Laos	*	
[58] Van Driesche & Bellows (1996)	Tropic&Subtropic		*
[59] Van Gils et al. (2005)	South Africa	*	
[60] Waterhouse (1994)	Southeast Asia	*	
[61] Weise & Tchamou (1999)	Cameroon	*	*
[62] Wilson (1995)	South Africa	*	
[63] Ye et al. (2004)	China	*	

Furthermore, NH4^{+}-N mineralization was smaller in leguminous fallow than in *C. odorata* fallow and depended on intrinsic soil properties (Koutika et al., 2004). *C. odorata* has a fast decomposition rate and might improve SOM quantity and quality (Norgrove, 2007). In fact, *C. odorata* decomposed fast with only 36% of its original weight remaining after 14 weeks (Kanmegne et al., 1999). Litzenburgen and Tong (1961) argued that *C. odorata* might be used as green manure with good results for low land rice. On an isohyperthermic *Typic Kandiudult* (USDA), Koutika et al. (2001) found that biological and chemical fertility under leguminous fallow (*Pueraria phaseoloides* and *Mucuna pruriens*) were higher than under natural regrowth (mainly composed of *C. odorata, Paspalum conjugatum* and *Ageratum conizoides*). The authors found that the beneficial effect of the legumes on N content in particulate organic matter (POM), one of active form of SOM, was small, but yet statistically significant after only 9 months of fallow.

POM and cumulative mineralized C values under *C. odorata* were better under fallow where *C. odorata* had been removed, underlining the possible management of this fallow type even though because of reduced weed control, removing *C. odorata* may bring more problems to farmers (Kanmegne et al., 1999). In fact *C. odorata* management may promote the sustainability of a cropping system in the area, since in some cases soil characteristics under natural fallow with dominantly *C. odorata* had closer beneficial relationships with crop yields (maize, groundnut and cassava) than leguminous *Pueraria* (Kanmegne et al., 1999). Moreover, Tian et al. (1998) found higher performance of maize after cover crop-fallow relative to the natural fallow with dominantly *C. odorata*, and argued that *Pueraria* could be a better alternative to the natural fallow with dominant *C. odorata* in sustaining soil productivity over a shortened period of fallow. In addition, Tian et al. (2005) showed that *Pueraria* cover fallow with 1 year fallow length could be a better alternative to natural *C. odorata* regrowth fallow for higher maize yield, and *Leucaena* alley cropping with 1 year fallow length could be a better alternative to natural regrowth fallow for maintenance of soil organic matter.

Nevertheless, one study conducted in low acidic soil of central Cameroon, found that it was not necessary to replace *C. odorata* with planted leguminous fallow since it performed better or equal to planted fallow, and *C. odorata* could be considered a beneficial fallow plant rather than be considered a weed (Koutika et al., 2005b). Studies conducted in central-southern Cameroon have shown that *C. odorata* is better adapted as a fallow plant on acidic soils (Kanmegne et al., 1999, Koutika et al., 2005a) and higher level of POM, an active part of SOM, might be observed in the acidic *Typic Kandiudox* compared to the *Typic Kandiudult* (Koutika et al., 2006). However, the quality of POM (C and N contents in fractions) remains lower indicating that sustainable crop production might be good after chemical constraints are alleviated. A survey conducted in the North West, South West and Littoral provinces of Cameroon, has shown that all farmers perceived *C. odorata*-dominated fallow to indicate softer and more fertile soils than *Imperata cylindrica*; the soils in which the latter species is found, require very high labour input for land preparation (Fuambeng Yonghachea et al., 2005). *C. odorata* was most effective in weed suppression and in fields where it was the fallow plant there was the lowest weed biomass compared to bamboo or other fallow species (Roder et al., 1995). While evaluating the size and composition of weed communities under different planted fallow in a rotational hedgerow intercropping system in the forest/savannah transition zone in Nigeria, Ekeleme et al. (2004) considered *C. odorata* as a fallow plant rather than a weed. The authors considered *C.*

odorata as a natural fallow plant, which was better at reducing weed growth than the planted fallow *Leucaena leucocephala*. Therefore several findings have shown that most farmer of appropriate region (part of Africa and Asia) may decide to grow the species as fallow according to its response to crop productivity and soil characteristics improvement.

Conclusions

It is apparent that the use of *C. odorata* is controversial as found by some authors (Roder et al., 1995; Weise and Tchamou, 1999). *C. odorata* may be considered either as a weed, i.e., an invasive plant species or as a beneficial fallow plant necessary to sustain agriculture in some part of the world (*Table 1*.). Thus, in some invaded countries as Nigeria, Australia, South Africa and Laos, *C. odorata* is considered as an aggressive invasive species and a very serious threat to agriculture and the environment (Olaoye, 1986; MacFadyen Cruttwell, 1989; Michael, 1989; Roder et al., 1994).

In fact, because of its negative impact on the forest in terms of economy, ecology and environment and social and health aspects, *C. odorata* is considered as an alien invasive species (Moore, 2004). *C. odorata* is a considerable threat to conservation and ecotourism since it invades primarily invaded natural areas and reduces the biodiversity of grasslands, savannahs and forests (MacDonald, 1983; Goodall and Erasmus, 1996; Matthews and Brand, 2004). *C. odorata* suppresses the growth of young pine and eucalypt in commercial tree plantations and allows fire and to penetrate deeper into plantations (Matthews, 2004; Matthews and Brand, 2004). It is a threat for the breeding habitat of the Nile crocodile in South Africa, as a decrease in temperature due to shading and crowding of nesting sites, can induce female based sex ratios or may even prevent embryotic development altogether (Leslie and Spolita, 2001; Matthews and Brand, 2004) *C. odorata* can also promote wildland fires (Moore, 2004), and may also cause skin problems and asthma in allergy-prone people.

However, in other part of the world or in other findings, *C. odorata* was not considered as a real threat to agriculture and the environment. Thus, Chandrasekaram and Swamy (2002) estimated that *C. odorata* was a good influence on ecosystem structure, species composition and aboveground net production, and stated that its sustainable use as an organic mulch in the establishment of young plantation should be encouraged to improve its effectiveness. Norgrove et al. (2000) argued that in low input systems, particularly those containing crops with a low initial uptake capacity for nutrients, weed management needs to find a balance between the competitive role of *C. odorata* as a nutrient sink and its potential benefit to the crop as regular source of organic matter and nutrients after slashing.

Even though it is often suggested that leguminous fallow would perform better than *C. odorata* and that the plant species has serious adverse effects on agriculture productivity (Weise and Tchamou, 1999), *C. odorata* has advantages in some agricultural systems. Thus, in slash-and-burn agriculture and on acidic soils of southern Cameroon, *C. odorata* performed better than *Pueraria* leguminous fallow (Kanmegne et al., 1999) and *Calliandra calothyrsus* (Koutika et al., 2005a). Roder et al. (1995) argued that with a progressive change to shorter fallow periods, the ability of *C. odorata* to suppress weeds and nematodes could become greater in Laos. However, the authors also argued that properties which are considered as advantages might become serious

constraints in the gradual change of the slash-and-burn system that integrate grazed fallow, crop rotation, fruit or/and timber production.

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A REVIEW OF THE POTENTIAL CLIMATE CHANGE IMPACT ON INSECT POPULATIONS – GENERAL AND AGRICULTURAL ASPECTS

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Abstract. Considering insect populations, we can see that climate change affects in many ways: it can cause a shift in geographical spread (Porter et al., 1991; Ward and Masters, 2007), abundance (Ayres and Lombardero, 2000; Olfert and Weiss, 2006) or diversity (Conrad et al., 2002; Feehan et al., 2009; Sharon et al., 2001), it can change the location, the timing and the magnitude of outbreaks of pests (Volney and Fleming, 2000), and it can define the phenological or even the genetic properties of the species (Gordo and Sanz, 2006; Klok and Chown, 2001; Parmesan, 2007). Long-time investigations of special insect populations, simulation models and scenario studies give us very important information about the response of the insects far away and near to our century. Getting to know the potential responses of insect populations to climate change makes us possible to evaluate the adaptation of pest management alternatives as well as to formulate our future management policy.
Keywords: *climate change, insects, pest management, simulation, agriculture*

Introduction

In our century increasing societal, environmental, and economic pressures force us to develop new agricultural pest management strategies (Lima and Berryman, 2006; Lima et al., 2008; 2009). Interdisciplinary approaches have the aim to find the way, how the environmental degradation caused by the use of chemicals can be decreased, how the productivity can be increased by reducing insect and disease damage to cultivated plants, and how the competition with weeds can be reduced. Crop and forestry population system models are useful tools to examine the interrelationships among plants, pests and the environment. With simulation models we can find optimal strategies that meet individual and societal goals (Tang and Cheke, 2008).

Improved techniques for managing pests require weather and insect data from thoroughly maintained monitoring as well as climate information and forecast to determine their suitability. Climatic change, including global warming and increased variability require improved analyses that can be used to assess the risk of the existing and the newly developed pest management strategies and techniques, and to define the impact of these techniques on environment, productivity and profitability (Lee et al., 2009a; 2009b). Each technique has to be evaluated whether and how it is suitable in the farming system where they are to be applied.

Nowadays, several studies are investigating the impact of climatic change on insect populations. Some of the effects can be discovered in laboratories, only (eg. the effect of humidity (Buxton, 2004)), some of them need field observations maintenance (Andrew and Hughes, 2007). The impact of climate change, moreover, alters from region to region, from species to species. Quite a lot of new methods from different disciplines are used to detect the most important effects. Therefore, the studies of climate change effects are considering different aspects, such as palaeontological, agricultural (Spencer et al., 2009), medical (Kearney et al., 2009; Kiritani, 2006; Takken and Knolsm, 2007), geological (Uniyal and Uniyal, 2009), biological as well as the aspects of forestry management. This widespread research work requires interdisciplinary cooperation of researchers from several fields (Hilker and Westerhoff, 2007; Strand, 2000). We give a short review of the main approaches.

The most common methods

Palaeontology

One of the most important ways to find out what climate change can bring us is to look back into the very past. A few thousand years ago some regions were characterized by such kind of vegetations that existed within warming thermal conditions analogous to those today. For example, the transition from parkland vegetation and insects to the one of coniferous forest of south-western Ontario region indicates that the climate continued gradually warm through the mid-Holocene (Schwert et al., 1985).

The Lateglacial-Holocen transition is characterized by major changes in the insect fauna, too, reflecting an extremely rapid climate change in South-Sweden, as well as in Swiss-Alps. In these regions the cold-adapted species assemblage was immediately replaced by temperate species (Lemdahl, 1991; 2000). During the same time period most of temperate species of Chihuahuan Desert (Texas) were replaced either by desert species or more cosmopolitan taxa (Elias and Devender, 1990). The above studies pointed out the dependency of the changes in climate and fauna. The question of how these kinds of responses proceed was studied by (Ammann, 2000).

Models, simulations and scenarios

Developing ecological and simulation models is a very useful tool to find out the response of a system to an event or a series of events (Estay et al., 2009; Gillman, 2009). Ecological or meteorological models describe biological or climate properties mathematically, while simulations make a computer based models system supplied with a great amount of empirical data (Musolin, 2007).

To reach his above mentioned palaeontological results in Swiss-Alps, (Lemdahl, 2000) applied a so-called climatic reconstruction (MCR) method that simulates realistic climate data in the past. Simulated weather data, however, are most commonly used to examine the potential future effects. These approaches are called scenario studies.

The main problems that have to precede scenario studies are, nevertheless, the evaluation, the validation and verification of the applied models. Though several models have been developed e.g. for the carbon budget of boreal forests, enormous problems remain in incorporating pest effects in these models. These problems have their origins, partly in scaling. The common problems of verification and validation of model results

are particularly troublesome in projecting future productivity (Volney and Fleming, 2000).

A main point of scenario studies is, therefore, how the applied model should be scaled. Hanson and Weltzin (2000) noticed, that although early model predictions of climate change impacts suggested extensive forest dieback and species migration, more recent analyses suggest that catastrophic dieback will be a local phenomenon, and changes in forest composition will be a relatively gradual process. Better climate predictions at regional scales, with a higher temporal resolution (months to days), coupled with carefully designed, field-based experiments that incorporate multiple driving variables (e.g. temperature and CO_2), will advance our ability to predict the response to climate change.

Time-dependent models developed at fine spatial resolution of experimental studies are widely used to forecast how plant – insect populations will react over large spatial extents. Usually the best data available for constructing such models comes from intensive, detailed field studies. Models are then scaled-up to coarser resolution for management decision-making. Scaling-up, however, can affect model predictions and dynamical behaviour which can result misinterpretation of model output. The potential negative consequences of scaling-up deserve consideration whenever data measured at different spatial resolutions are integrated during model development, as often happens in climate change research (Fleming et al., 2002).

Chen et al. (2000) investigate the integrated effects of insect infections, management practices, carbon cycle and climatic factors both at regional and global scales.

To see that there can be great difference between the responses of even similar species, we refer to Conrad et al. (2002). They examined the garden tiger moth (*Arctia caja*) that was widespread and common in the UK in the last century, but its abundance fell rapidly and suddenly after 1984. The most UK butterflies are expected to increase under UK climate change scenarios of global warming. Contrary to them, garden tiger is predicted to decrease further because of warm wet winters and springs, to which it is very sensitive (Conrad et al., 2002).

Ecological models serving climate change studies

We give a short list of the most widely applied ecological models focused to insect populations.

The Forest Vegetation Simulator (FVS) is a distance-independent, geographic region dependent individual-tree forest growth model that has been widely used in the United States for about 30 years to support management decision making. It has been continuously extended, improved and adapted to further management tasks like prediction of climate change effects. Component models predict the growth and mortality of individual trees, and extensions to the base model represent disturbance agents including insects, pathogens, and fire. The geographic regions are represented by regionally specific model variants. The differences are due to data availability and the applicability of existing models. The model supports specification of management rules in the input (Crookston and Dixon, 2005; Dyck, 1999).

The Phenology and PopulatIoN SIM (INSIM) is an age – structured model that needs biological information on the insect species and gives calculations on the number of individuals and the development of the population. It involves a complex pest – natural enemies model, as well (Mols, 1990; 1992).

Agro – ECOsystem ManagemenT and OPtimization Model (ECOTOPE) is a typical simulation model, which describes processes of an agricultural ecosystem for crop growth, nitrogen dynamics in soil and pest population. It is used to derive optimum management strategies (Seppelt, 1999; 2000; 2001).

Boundary LAYER Model (BLAYER) simulates atmospheric flows and it has been adapted to forecast the timing and location of insect pest migrations into the United States corn belt. It is very useful to study the possible changes in pest populations like migration or dispersal patterns resulted by climate change (Paegle and McLawhorn, 1983).

Boll Weevil DISPersal Model (BWDISP) is a stochastic simulation model that predicts the spread of boll weevil populations on cotton. Because the development and dispersal of this insect is sensitive to temperature, it is important to understand how this insect will potentially respond to climate change. In addition, without proper management of this pest, other secondary pests may attack the crop (McKibben et al., 1991).

Northern Corn ROOTWORM Model (ROOTWORM) is a process – oriented simulation model that examines the population dynamics of corn – rootworm in the northern United States. The rootworm attacks both the roots and tassels of corn, decreasing yields. The model examines how planting date affects the population dynamics of the insects. It gives information on phenology and the number of individuals in each growth state of corn. The model can analyse global change impact on the population levels and distribution of the insects, as well as the potential economic impacts (Norango and Sawyer, 1989).

Potential responses of insects to climate change

Climate and weather can substantially influence the development and distribution of insects. Current estimates of changes in climate indicate an increase in global mean annual temperatures of 1°C by 2025 and 3°C by the end of the next century. Such increases in temperature have a number of implications for temperature-dependent insects, especially in the region of Middle-Europe. Changes in climate may result changes in geographical distribution, increased overwintering, changes in population growth rates, increases in the number of generations, extension of the development season, changes in crop-pest synchrony of phenology, changes in interspecific interactions and increased risk of invasion by migrant pests (Memmott et al., 2007; Parmesan, 2007; Porter et al., 1991).

Under the climatic changes projected by the Goddard Institute for Space Studies general circulation model, northward shifts in the potential distribution of the European corn borer of up to 1220 km are estimated to occur, with an additional generation found in nearly all regions where it is currently known to occur (Porter et al., 1991).

Several results on the effect of climate change on insects were published in the field of forestry sciences, since insects cause considerable loss of wood that has an adverse effect on the balance of carbon sequestered by forests. Volney and Fleming (2000) state that pests are major, but consistently overlooked forest ecosystem components that have manifold consequences to the structure and functions of future forests. Global change will have demonstrable changes in the frequency and intensity of pest outbreaks, particularly at the margins of host ranges.

Ayres and Lombardero (2000) have shown that climate change has direct effects on the development and survival of herbivores and pathogens; physiological changes in tree defences; and indirect effects from changes in the abundance of natural enemies (e.g. parasitoids of insect herbivores), mutualists (e.g. insect vectors of tree pathogens), and competitors.

Because of the short life cycles of insects, mobility, reproductive potential, and physiological sensitivity to temperature, even modest climate change will have rapid impacts on the distribution and abundance of many kinds of insects. To consider scenario studies, some of them predict negative, but many forecast positive effects on insects. E.g. global warming accelerates insect development rate and facilitate range expansions of pests, moreover, climate change tends to increase the vulnerability of plants to herbivores. One alarming scenario is that climate warming may increase insect outbreaks in boreal forests, which would tend to increase forest fires and exacerbate further climate warming by releasing carbon stores from boreal ecosystems (Ayres and Lombardero, 2000).

Hanson and Weltzin (2000) studied especially the drought disturbances caused by climate change. They showed that severe or prolonged drought may render trees more susceptible to insects.

Climate variability at decadal scales influences the timing and severity of insect outbreaks that may alter species distributions. Coops et al. (2005) have presented a spatial modelling technique to infer how a sustained change in climate might alter the geographic distribution of the species. Using simulations they produced a series of maps that display predicted shifts of zones where the species they examined might expand its range if modelled climatic conditions at annual and decadal intervals were sustained.

The connection between temperature tolerance and phenology of insects was investigated by Klok and Chown (2001). They defined how current climate change like increased temperature and decreased rainfall affect on physiological regulation and susceptibility.

Powell and Logan (2005) have reviewed the mathematical relationship between environmental temperatures and developmental timing and analysed circle maps from yearly oviposition dates and temperatures to oviposition dates for subsequent generations. Applying scenarios for global warming they proved that adaptive seasonality may break down with little warning with constantly increasing (and also decreasing) temperature.

Forecasted increases in atmospheric CO_2 and global mean temperature are likely to influence insect – plant interactions. Plant traits important to insect herbivores, such as nitrogen content, may be directly affected by elevated CO_2 and temperature, while insect herbivores are likely to be directly affected only by temperature. Flynn et al. (2005) stated that insect populations did not change significantly under elevated CO_2, but tended to increase slightly. Average weight decreased at high temperatures. Plant height and biomass were not significantly affected by the CO_2 treatment, but growth rates before infestation were enhanced by elevated CO_2. These results indicate that the combined effects of both elevated CO_2 and temperature may exacerbate pest damage to certain plants, particularly to plants which respond weakly to increases in atmospheric CO_2.

Up to this time, as we have seen, mainly two climatic factors – temperature and humidity have been investigated. Though, it is possible that some parts of solar radiation have at least the same importance in controlling insect populations (Buxton, 2004).

Last, but not least, changes in climate increases the likelihood of insect transport from regions to regions, as well Whinam et al. (2005).

Special agricultural aspects of climate change effect on insects

Global climate change impact on plant - pest populations depends on the combined effects of climate (temperature, precipitation, humidity) and other components like soil moisture, atmospheric CO_2 and tropospheric ozone (O_3). Changes in agricultural productivity can be the result of direct effects of these factors at the plant level, or indirect effects at the system level, for instance, through shifts in insect pest occurrence. With respect to crops, the data suggest that elevated CO_2 may have many positive effects, including yield stimulation, improved resource - use efficiency, more successful competition with weeds, reduced O_3 toxicity, and in some cases better pest and disease resistance. However, many of these beneficial effects may be lost — at least to some extent — in a warmer climate. Warming accelerates plant development and reduces grain-fill, reduces nutrient-use efficiency, increases crop water consumption, and favours weeds over crops. Also, the rate of development of insects may be increased. A major effect of climate warming in the temperate zone could be a change in winter survival of insect pests, whereas at more northern latitudes shifts in phenology in terms of growth and reproduction, may be of special importance. However, climate warming disturbs the synchrony between temperature and photoperiod; because insect and host plant species show individualistic responses to temperature, CO_2 and photoperiod, it is expected that climate change will affect the temporal and spatial association between species interacting at different trophic levels. Although predictions are difficult, it seems reasonable to assume that agro – ecosystem responses will be dominated by those caused directly or indirectly by shifts in climate, associated with altered weather patterns, and not by elevated CO_2 per se. Overall, intensive agriculture may have the potential to adapt to changing conditions, in contrast to extensive agricultural systems or low - input systems which may be affected more seriously (Fuhrer, 2003).

Crop protection in Europe became strongly chemically oriented in the middle of the last century. An excellent climate for fast reproduction of pests and diseases demanded high spray frequencies and, thus, resulted in quick development of resistance against pesticides. This initiated a search for alternatives of chemical pesticides, like natural enemies for control of pests. A change from chemical control to very advanced integrated pest management programs (IPM) in European greenhouses took place at the end of the last century (Kogan and Jepson, 2007; Lenteren, 2000). For the main greenhouse vegetable crops in northern Europe, most insect problems can now be solved without the use of insecticides. IPM without conventional chemical pesticides is a goal that will be realised for most of the important vegetables in Europe, not limited to greenhouse vegetables. At the same time, however, climate change affects the distribution, the phenology, the susceptibility and the interrelationship of insects drastically, which emphasize the risk of sustainable crop protection by loosing the control on pests – natural enemies populations.

Summary

Based on results of factorial experiments under a range of experimental conditions, it is difficult to draw generalized conclusions. Climate change on insect populations, however, forces us to assess the ecological, economical, and social risk of biotic disturbances (Merrill et al., 2008). There are a number of priorities for future research such as examination of the influence of climatic variables, long-term monitoring and modelling of insect population levels and insect behaviour, identification of potential migrants and consideration of possible changes in pest management systems in agriculture and forestry.

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REMOVAL OF METHYLENE BLUE FROM AQUEOUS SOLUTION USING GHASSOUL, A LOW-COST ADSORBENT

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Abstract. The present study examined the use of ghassoul, natural clay available in large quantities under the Atlas Mountains in Morocco, for the removal of the basic dye, methylene blue, from aqueous solutions. The effect of initial dye concentrations, agitation time, pH and temperature on adsorption capacities of methylene blue was investigated. The initial pH of the aqueous solution and the change of temperature (25-55 °C) were found to have little effect on the adsorption process. The adsorption was a rapid with 90-99% of the dye removed within the first 10-20 min. The adsorption kinetics are described successfully using a pseudo-second order rate equation and the rate constant decreases with increasing the initial concentration of MB. Experimental and calculated kinetic data for equilibrium are well expressed by Langmuir isotherm. The equilibrium adsorption capacity of ghassoul was determined with the Langmuir equation as well as the pseudo-second-order rate equation and found to be >290 mg dye per gram of the adsorbent. The results indicate that ghassoul could be employed as a low cost alternative to commercial activated carbon in wastewater treatment for the removal of colour and dyes.
Keywords: natural clay, basic dye, adsorption kinetics

Introduction

Coloured dye wastewater arises as a direct result of the production of dye and also as a consequence of its use in the textile and others industries (Allen and Koumanova, 2005). Considering both volume-discharged and effluent combustion, the wastewater from the textile industry is rated as the most polluting among all industrial sectors (McKay et al., 1981). Their presence in water, even at very low concentrations, is highly visible and undesirable and may significantly affect photosynthetic activity in aquatic life due to reduced light penetration (Aslıhan Gücek et al., 2005). As dyes are toxic to micro-organisms, stable to light irradiation and heat, they cannot be easily removed by conventional wastewater treatment processes due to their complex structure and synthetic origins (Pagga and Taeger, 1994).

In our day various physical–chemical techniques have been studied to assess their applicability for the treatment of this type of industrial discharge. Among these processes may be included coagulation, adsorption, precipitation, flocculation, ozonation (Stephenson and Sheldon, 1996; Churcley, 1998; Nyholm and al., 1992; Ogutveren and Kaparal, 1994; Coro and Laha, 2001; Kang and al., 2000). Of the numerous techniques mentioned, the adsorption process is one of the effective techniques that have been successfully employed for colour removal from wastewater.

Many adsorbents have been tested to reduce dye concentrations from aqueous solutions. Currently, the most common procedure involves the use of activated carbons (Walker and Weatherley, 1998; Rao and Ashutosh, 1994; Al-Degs et al., 2000) as adsorbents because of their higher adsorption capacities. Despite the prolific use of activated carbon for wastewater treatment, carbon adsorption remains an expensive process, and this fact has recently prompted growing research interest into the production of low-cost alternatives. Various workers have exploited substances such as fly ash (Voudrias et al., 2002; Albanis et al., 2000; Rachakornkij et al., 2004; Mohan et al., 2002), activated carbon prepared from agricultural waste material (Erhan Demirbasa et al., 2004; Kunwar et al., 2003), perlite (Dogan and Alkan, 2003; Dogan et al., 2004), pyrophyllite (Aslıhan Gücek et al., 2005), sepiolite (Alkan et al., 2004; Armagan, 2003), diatomite (Erdem et al., 2005), bentonite (Ozcan et al., 2004; Hu et al., 2006), actived charcoal (Daniela Suteu and Doina Bilba, 2005), chitosan (Ming-Shen Chiou et al., 2003), baggase (Raghuvanshi et al., 2004), alumina (Moreira et al., 1998).

The material used in the present study was prepared with a purified Moroccan clay mineral from mid Moulouya known as a ghassoul or rhassoul. The ghassoul is capable of adsorbing guest cations according to the magnitude of the cation exchange capacity (CEC) (Elmchaouri and Mahboub, 2005; Faust et al., 1959). Considered for its astringent virtues, absorbing and lenitive, ghassoul, clay has a very long history of use in beauty care. Even today it remains a favourite with upscale spas and resorts worldwide. Ghassoul is inexpensive and easily available, this could make it a viable candidate as an economical adsorbent for removing unwanted hazardous components from contaminated water.

The present work aims to study a convenient and economic method for basic dye (Methylene Blue) removal from water by adsorption on an abundantly available adsorbent. Batch studies were carried out involving process parameters such as initial dye concentration, pH, contact time and temperature. Equilibrium and kinetic analysis were conducted to determine the factors controlling the rate of adsorption, the optimization of various parameters in dye recovery and to find out the possibility of using this material as low-cost adsorbent for dye removal.

Materials and methods

Preparation of adsorbents

The clay sample used in this investigation was taken from deposits in the careers of Tamdalet in the province of Boulemane, Morocco. The ghassoul sample was treated before using in the experiments as follows: a distilled water suspension of the clay was dispersed for approximately 4 h and then cleaned several times with de-ionized water. The fine fraction of ghassoul clay was collected by repeated dispersion, sedimentation and siphoning techniques. The solid sample was dried at 105 °C for 24 h, ground then sieved by 140 µm sieve. The sample was analysed for its chemical composition and found to contain SiO_2 58.16 %, Al_2O_3 4.48%, MgO 27.44%, CaO 1.88%, Fe_2O_3 1.92%, K_2O 1.05%, Na_2O 0.17% and loss on ignition 19.5%. The presence of a significant amount of silicium and a minimum aluminium quantity lead to the supposition that this sample can classified with the silicate family. On another hand it contains a large quantity of magnesium, thus we can consider it as a magnesian smectite.

Dye and chemicals

All chemicals used were of analytical reagent grade and supplied by Merck Germany. Stock solutions of the test reagents were made by dissolving Methylene Blue {3,9-bis dimethyl- aminophenazo thionium chloride}, in doubly distilled water. The structure of this dye is shown in *Fig. 1*.
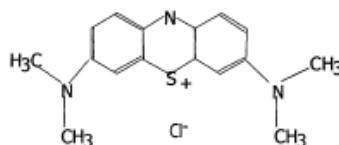
Figure 1. *The structure of methylene blue*

All colour measurements were made on a Shimadzu 160 UV-visible spectrophotometer in the visible range in absorbance mode. Absorbance values were recorded at the wavelength of maximum absorbance (λmax = 664.5 nm). Absorbance was found to vary linearly with concentration, and dilutions were made when the absorbance exceeded 0.8.

Adsorption studies

Adsorption studies were performed by the batch technique to obtain rate and equilibrium data. Various Methylene Blue solutions with different initial concentrations, in the range of 100 - 600 mg L^{-1}, were prepared by diluting stock dye solution (1g L^{-1}).

Equilibrium experiments, to determine the adsorption capacity of ghassoul, were conducted using 125 mL centrifuge tubes where 0.2 g of ghassoul and 100 ml of the above dye solution were added and shaken for 2 h at 25 °C. Kinetic experiments demonstrate that adsorption equilibrium was reached within 1-2 h.

For the experiments of adsorption kinetics, 1g ghassoul sample was added into a litre of dye solution at desired concentration. Under stirring, samples were extracted at selected time intervals (from 10 to 210 min) and centrifuged for 10 min at 3000 rpm and then the concentration of the residual dye (Ce), was determined. The amounts of dyes adsorbed were calculated from the concentrations in solutions before and after adsorption according to the equation:

$$qe = V.10^{-3} \times \frac{C_0 - Ce}{m} \qquad (Eq.1)$$

where: qe is the equilibrium dye concentration on adsorbent (mg of dye / g of sorbent), C_0 and Ce are respectively the initial and the equilibrium concentration of dye in solution (mg L^{-1}), m is the mass of dry adsorbent used (g) and V is the volume of dye solution (mL).

The effect of pH was observed by studying the adsorption of dye over a pH range of 3 - 11. The sorption kinetic studies were also carried out at different temperatures, i.e., 25, 40, and 55 °C, to determine the effect of temperature and to evaluate the sorption thermodynamic parameters. The concentration of ghassoul studied was 0.2 g per 100 mL of dye solution with the dye concentration of 500 mg L^{-1}. Each experimental point was an average of three independent adsorption tests.

Results and discussion

Effect of pH

The adsorption capacity of Methylene Blue onto ghassoul at various pHs (3, 5, 7, 9 and 11) are shown in *Fig. 2*. It is seen that the adsorption capacities was affected very slightly by the pH. Since the pH of freshly prepared dye solution was about 4.5 to 5, no initial pH adjustment on the dye solutions was made for subsequent studies.
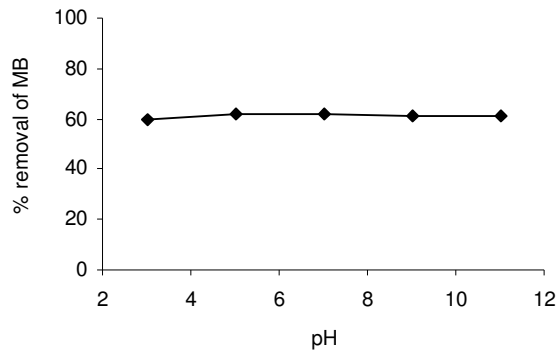

Figure 2. *The effect of pH on the adsorption of MB ont ghassoul*

Effect of temperature

The adsorption isotherms at different temperatures (25, 40 and 55 °C) are shown in *Fig. 3*. The results obtained indicate that the change in temperature in the specified range has almost no effect on the adsorption capacity of Methylene Blue. For this reason, a room temperature of 25 °C was used throughout this work.
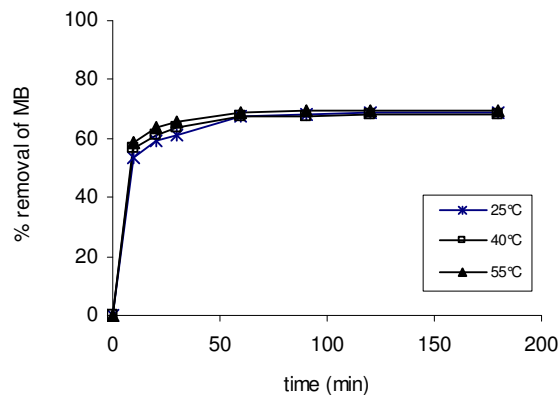

Figure 3. *The effect of temperature on the adsorption of MB onto ghassoul*

Adsorption isotherms

Adsorption equilibrium

The equilibrium adsorption of Methylene Blue was studied as a function of concentration. The amount of dye adsorbed *qe*, plotted against the equilibrium concentration *Ce* for MB, is given in *Fig. 4*. The equilibrium adsorption of MB increases with the increase of initial dye concentration, showing the adsorption process to be dependent on the initial concentration. Initially, the adsorption isotherms of dye molecules show a steeply rising part, suggesting a strong affinity of the dye molecules for the surface sites on ghassoul. Then the amount of adsorption reaches a limiting value of around 300 mg g^{-1}. As also seen in *Fig. 4*, adsorption of Methylene Blue onto ghassoul forms a typical Langmuir-type isotherm, which indicates that dye molecules outcompete water molecules for the sites available on the surface of ghassoul.
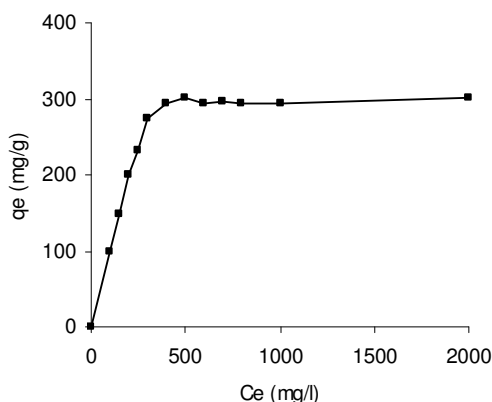

Figure 4. Equilibrium isotherms for MB adsorption onto ghassoul

Isotherm analysis

Adsorption isotherms are critical in optimizing the use of adsorbents and describe how adsorbate interacts with adsorbent. The analysis of the isotherm data with either theoretical or empirical equations is important to develop an equation which accurately represents the results and which could be used for design purposes (McKay et al., 1985). Several isotherm equations are available. Two of them have been selected in this study: Langmuir and Freundlich isotherms.

The Langmuir isotherm assumes that all the adsorption sites are equivalent and that there is no interaction between adsorbed species (Gupta and Ali, 2001). The Freundlich model assumes a heterogeneous adsorption surface and active sites with different energy (Lagergren, 1998).

The applicability of the Freundlich adsorption isotherm is analyzed by plotting log(*qe*) versus log(*Ce*). Furthermore, the low correlation coefficient ($R^2 = 0.7378$) show the poor agreement of the Freundlich isotherm with the experimental data.

The Langmuir isotherm has been widely used to describe single-solute systems. It is based on the assumption that intermolecular forces decrease rapidly with distance and consequently it predicts monolayer coverage of the adsorbate on the outer surface of the

adsorbent. The isotherm equation further assumes that adsorption takes place at specific homogeneous sites within the adsorbent and there is no significant interaction among adsorbed species. Theoretically, the adsorbent has a finite capacity for the adsorbate. Once a dye molecule occupies a site, no further adsorption can take place at that site. The rate of sorption to the surface should be proportional to a driving force which times an area. The driving force is the concentration in the solution, and the area is the amount of bare surface.

The linear form of the Langmuir equation can be written as follow:

$$\frac{Ce}{qe} = \frac{1}{qm \times K_L} + \frac{Ce}{qm} \qquad \text{(Eq.2)}$$

where qe is solid-phase adsorbate concentration at equilibrium (mg g^{-1}), Ce is aqueous-phase adsorbate concentration at equilibrium (mg L^{-1}), qm (mg g^{-1}) is the maximum amount of dye per unit weight of ghassoul to form a complete monolayer coverage on the surface, K_L is the Langmuir constant related to the affinity of binding sites (L mg^{-1}). K_L and qm are computed from the slope and intercept of the straight line of the plot of (Ce/qe) vs. Ce.

Adsorption isotherms were obtained in terms of equations (2) by using experimental adsorption results in this equation. Values for qm and K_L are summarised in *Table 1*. The isotherm data were calculated from the least square method and the related correlation coefficients (R^2 values) are given in the same table. As it cab be seen in *Table 1*, the Langmuir equation represents the adsorption process very well; the R^2 value was higher than 0.999, indicating a very good mathematical fit. The fact that the Langmuir isotherm fits the experimental data very well may be due to homogenous distribution of active sites on the ghassoul surface.

Table 1. *Values of the constants in Langmuir and Freundlich models*

qm mg g^{-1}	K$_L$ L m g^{-1}	R^2
294	0.59	0.9998

The essential characteristic of the Langmuir isotherm may be expressed in terms of the dimensionless separation parameter R (Weber and Chackravorti, 1974), which is indicative of the isotherm shape that predicts whether an adsorption system is 'favourable' or 'unfavourable'. R is defined as:

$$R = \frac{1}{(1 + K_L \times C_0)} \qquad \text{(Eq.3)}$$

where: K_L is a Langmuir constant and C_0 is the initial concentration of dye.

According to the value of *R* the isotherm shape may be interpreted as follows:

Value of *R*	Type of adsorption
R>1.0	unfavourable
R = 1.0	linear
0<*R*<1.0	favourable
R = 0	irreversible

The initial dye concentration C_0 used in the adsorption isotherm studies was in the range of 100 to 600 mg L^{-1} and the corresponding values of parameter *R* are in the range 0.0164 - 0.0028. The fact that all the *R*-values for the adsorption of MB on ghassoul are found to be less than 1 and greater than 0, shows that the adsorption process is favourable.

R values showed that the adsorption of MB was more favourable for the higher concentration than the lower one, which is due to the effect of the pore diffusion sorption phenomenon.

Adsorption kinetics

Effect of the initial dye concentration

The results of methylene blue adsorption with increasing contact time are presented in *Fig. 5*. On changing the initial concentration of MB solution from 100 to 600 mg L^{-1}, it was found that the MB adsorption increased with increasing contact time. The curves in *Fig. 5* show that the adsorption is a very fast process with about 90-99% of the ultimate adsorption was uptaken by ghassoul within the first 10-20 minutes, indicating high affinity between ghassoul and methylene blue. Then the amount of adsorption reaches a limiting value, beyond which no further adsorption can take place. The necessary time for the equilibrium to be reached was about 90 min. Based on these results, ninety minutes was taken as the equilibrium time in kinetic adsorption experiments.
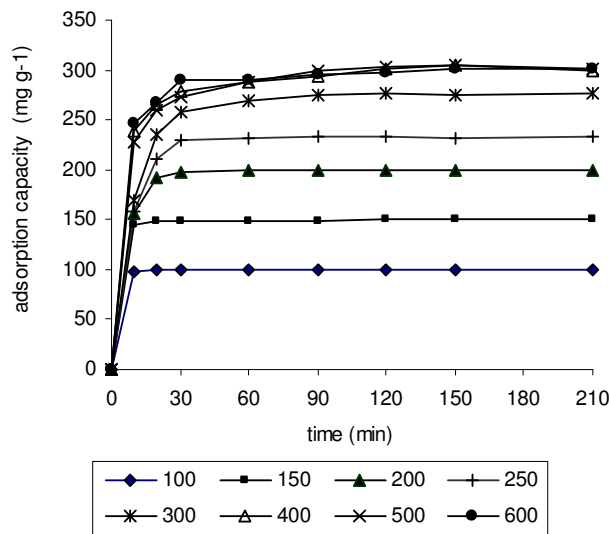

Figure 5. *Effect of initial concentration on the adsorption of MB onto ghassoul*

Kinetic studies

It is important to be able to predict the rate at which contamination is removed from aqueous solutions in order to design an adsorption treatment plant. In order to investigate the mechanism of adsorption and potential rate controlling steps such as mass transfer and chemical reaction, the kinetics of MB sorption onto ghassoul was investigated using two different models: the pseudo-first-order (Lagergren, 1998), and the pseudo second-order (Ho et al., 2000) kinetic models. The conformity between experimental data and the model predicted values was expressed by the correlation coefficients (R^2, values close or equal to 1). A relatively high R^2 value indicates that the model successfully describes the kinetics of MB adsorption.

The pseudo first-order and the pseudo second-order equations are generally expressed as follows:

$$\log(qe - qt) = \log qe - \frac{K_1}{2.303} \times t \tag{Eq.3}$$

$$\frac{t}{qt} = \frac{1}{K_2 \times qe^2} + \frac{t}{qt} \tag{Eq.4}$$

where:
- qe and qt are the adsorption capacity at equilibrium and at time t, respectively (mg g^{-1}),
- K_1 is the rate constant of pseudo first-order adsorption (L min^{-1}),
- K_2 is the rate constant of pseudo second-order adsorption (g mg^{-1} min^{-1}).

The plot of log ($qe - qt$) vs. t should give a linear relationship from which K_1 and qe can be determined from the slope and intercept of the plot, respectively.
The plot of (t/qt) vs. t should give a linear relationship from which qe and K_2 can be determined from the slope and intercept of the plot, respectively.

Table 2. *Comparison of first-Order and second-Order adsorption rate constants, calculated qe and experimental q_{exp} values for different initial dye concentrations*

C_0 mg L^{-1}	q_{exp} mg g^{-1}	First-Order Kinetic Model			Second-Order Kinetic Model		
		qe mg g^{-1}	K_1 min^{-1}	R^2	qe mg g^{-1}	K_2 g mg^{-1} min^{-1}	R^2
100	100	3.78	0.96 E-2	0.2084	100.00	66.6 E-3	1
150	149	6.31	1.31 E-2	0.3172	149.25	22.4 E-3	1
200	199	14.06	1.86 E-2	0.4389	200.00	6.09 E-3	0.9999
250	233	34.32	2.25 E-2	0.6320	238.09	2.23 E-3	0.9996
300	275	63.25	2.60 E-2	0,7381	277.77	1.25 E-3	0.9994
400	294	71.99	2.00 E-2	0.7044	303.03	1.55 E-3	0.9996
500	300	76.93	2.39 E-2	0.7342	303.03	1.31 E-3	0.9996
600	295	64.96	2.34 E-2	0.8394	303.03	1.81 E-3	0.9998

Table 2 lists the calculated adsorption constants at different initial dye concentrations by the pseudo first-order and pseudo second-order models. The correlation coefficients, R^2, of the pseudo first-order model at different initial concentrations (100 - 600 mg L^{-1}) did not exceed the values of 0.8342. The calculated qe values are too low compared with experimental q_{exp} values. This shows that the adsorption of MB by ghassoul did not

follow the pseudo first-order. The correlation coefficient R^2 for the pseudo second-order adsorption model has an extremely high value (greater than 0.999), and the calculated qe values also agree very well with the experimental data. These results suggest that the overall rate of the dye adsorption process appears to be controlled by the chemical sorption or chemisorptions process (Aslıhan Gücek et al., 2005; Ho and Mckay, 1998). The similar phenomena have also been observed in the adsorption of MB onto pyrophyllite (Aslıhan Gücek et al., 2005) and perlite (Dogan et al., 2004).

According to the pseudo-second order model, the adsorption rate dq/dt is proportional to the second order of ($qe - q_t$). Since the ghassoul in our experiments had very high equilibrium adsorption capacity qe, the adsorption rates became very fast and the equilibrium times were short (Ming-Shen Chiou et al., 2003) Such short equilibrium times coupled with high adsorption capacity indicate a high degree of affinity between the dyes and the ghassoul (Aksu and Tezer, 2000).

Conclusion

The study of ghassoul as a natural clay adsorbent to remove dye, Methylene Blue, from aqueous solution has been investigated. Ghassoul was shown to be an excellent adsorbent. The results demonstrate that the MB removal is a very fast process, indicating that the adsorption is a surface phenomenon and that the surfaces are readily accessible to the ions in solution. The contact time required for maximum MB adsorption onto ghassoul was found to be nearly 90 min. The data showed that the second-order equation provide the best correlation for the experimental data which indicate the chemical sorption process. The Langmuir model can be used to fit the data and estimate model parameters with adsorption capacity of 290 mg g^{-1}. The adsorption capacity of other adsorbents for MB obtained by some other investigators are presented in *Table 3*. A comparison of these values with the one obtained in this study showed that ghassoul used in this research exhibited a higher capacity for MB adsorption from aqueous solutions.

Table 3. *Adsorption capacity of methylene blue on various adsorbents*

Sorbent	Sorption capacity (mg g^{-1})	Reference
Activated carbon (coconut shell fibbers)	19.59	19
Activated carbon (olive stones)	303	43
Cotton waste	240	45
Date pits	80.3	42
Fly ash	53.84	36
Perlite	5.6 – 9.08	18
Perlite	162.3	41
Perlite (EP)	17.4 – 31.7	44
Pyrophyllite	70.42	3
Zeolite	53.1	41

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