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PhD. Thesis

The impact of different management treatments on upland meadow characteristics

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I declare, that I worked this study out myself using only the cited literature.

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Content

1. Introduction

1.1 The cessation of grassland management and its consequences

Large boundary regions with former German settlement were depopulated after World War II in the Czech Republic and vast landscape areas were ceased from agricultural management. The postwar changes in agriculture are characterized by large area management devaluating old peasant experience, consolidation of land, artificial fertlizers and pesticides usage, often in overabundance, centrally controlled landscape desiccation and the destruction of traditional farming formely practised by small-scale farmers, e.g. extensive grazing. The consequences for vegetation are especially the degradation and gradual destruction of plant communities related to the traditional types of management and, on the contrary, the development of ruderal vegetation with high proportion of neophytes (Sádlo 2007).

Most forest-free natural reserves in the Czech Republic are endangered by successional processes (Prach 1994).

The main aim of protected areas is the conservation of different types of cultural landscape with high ecological and aesthetic potential and with high proportion of forest and meadow ecosystems (Šrámek et al. 2001).

The vegetation is generally very good indicator of environmental conditions and it is possible to use it indirectly for environmental monitoring. All meadow communities has evolved secondarily on place of woods except of grasslands above timber-line, some peat-bogs and fragments of steppe vegetation. Hence, they are continually subjected to successional press of wood species whenever their periodic management abates. The greatest threat of very valuable grasslands presents expansion of wood species and also enhanced nutrients input and artficial wetland drainage. Whatever change of management ways mostly reveal itself very quickly in botanical composition and vitality of individual populations of grassland species. It is possible to suggest an optimal management based on monitoring of successional changes and after introduction of this management to evaluate its fruitfulness (Prach 1994).

The elaboration and above all the implementation of appropriate management is the presumption of grassland maintenance (Mrkvička & Veselá 2001). Properly chosen monitoring is essentially the sole way how to find out, whether the management was successful or not (Prach 1994).

Long-term experiments are valuable for examining the effect of disturbance on species richness because long-term disturbance may result in changes in the environment such as vegetation structure (Knapp & Seastedt 1986) and soil organic matter and nitrogen availability (Fynn et al. 2004), which in turn, may affect species richness. In contrast, short-term experiments may result in spurious conclusions over the impact of disturbance on species richness because the initial response of species richness to disturbance may only be transient (Wilson & Tilman 2002).

Results of Prach & Řehounková (2006) imply, that surrounding vegetation, macroclimate, soil moisture, amount of nitrogen and soil texture appeared to have the highest influence on the course of succession. Less influential were the size of a disturbed site, pH, organic matter and phosphorus content. Surrounding vegetation exhibited a significant effect in all cases where this was considered. These results imply that succession cannot be studied without the landscape context (Prach & Řehounková 2006).

Meadow phytocoenoses belong to the richest communities: they are absolutely the richest ones on small-scale areas. There are more mechanisms known, which maintain the species richness. The most likely mechanism is the fact, that meadow communities are strongly disequilibrium systems with la emphasized by rge spatial heterogeneity in individual processes, different time responses to control mechanisms (e.g. different climatic effects), but also with very slight differences in management (Krahulec et al. 1994). The botanical composition depend upon the topography, bedrock, climate, soil and biotic factors including anthropic influences (Sádlo 2007). The replacement of individual species is strongly dependent on species composition: e.g. in species-poor upland matgrass swards *Festuca rubra* ang *Anthoxanthum* are those species, which increase after fertilization, whereas in more productive types they are the first species decreasing after the same treatment (Krahulec et al. 1994).

Meadow ecosystems represent the highest diversity of both species and biotopes. Semi-natural meadows require considerable amount of human energy for maintainance, namely by sheep and cattle grazing (Šrámek et al. 2001). In many species-rich hay meadows in Central Europe, the traditional extensive (low input, low output) management is no longer economical and meadows are either fertilized or abandoned. Both these practices lead to changes in species composition and usually to a loss in species diversity (Lepš 1999). This development is described also from other countries, e.g. from Italy (Tasser & Tappeiner 2002). Herb-rich upland meadows are result of sensitive farming of our ancestors. Whenever this farming is interrupted or neglected, quick changes leading to their botanical and agricultural degradation occur (Hadincová et al. 1997).

Abandonment of semi-natural grassland from agricultural use is a major risk for grassland diversity. Many investigations have shown that almost independently of the vegetation type, cessation of grassland management leads to successional change and to a loss of plant species diversity (Isselstein et al. 2005). Species diversity has declined in ecosystems worldwide as a result of habitat fragmentation, eutrophication, and land-use change (Collins 1998). Current land use is mainly controlled by the degree of accessibility for vehicles. Accessible areas are being used more and more intensively, while poorly accessible areas are being abandoned or used as pasture (Tasser & Tappeiner 2002).

The extent of species-rich semi-natural grasslands has decreased considerably throughout Europe during recent decades (Blackstock et al. 1999). It is very well confirmed both theoretically and via experiments, that the cessation of mowing of species-rich meadows leads to the species composition changes and to the decrease of species diversity. The strong competitors exceed, such as tall grasses and forbs, to the prejudice of ground-layer species which are cut-tolerant (Zeller et al. 2000). Abandoned meadows usually become covered with monotonous species-poor vegetation, in which a single grass species tends to dominate (Klimeš & Klimešová 2002). High levels of primary production following cessation of mowing result in large surface accumulation of litter which limit grasses but favour the proliferation of forbs (Dickinson & Polwart 1982). The cessation of grassland management results in gradual changes of soil conditions, e.g. the enhancement of organic matter and nutrients, mainly nitrogen, which gradually negatively affects the diversity of semi-natural meadows (Zobel et al. 1996). The soil pH decreases following abandonment of traditionally managed cut subalpine meadows (Zeller et al. 2000).

Although the total species richness generally decrease quickly during the first ten years after abandonment, the rare species are able to survive in abandoned meadows for a long time, but in very little abundance (Pykälä 2003). These species inevitably vanish, whenever the restoration does not take place. Management programmes resulting in the regeneration and spreading of rare species therefore should be performed while these species are still present in local flora. The development of phytocenoses on abandoned meadows is often characterized by an enhanced litter accumulation and by the dominance of perennial grass species, which could have a strong negative effect on recruitment and survival of both rare and less prominent species (Symstad & Tilman 2001).

1.2 Management treatments

In recent decades, several countries have applied intensive management programmes to preserve the remaining semi-natural areas and to restore the former species-rich communities. The primary management tool is the reintroduction of former management practices, such as mowing, grazing and sod cutting (removal of vegetation and topsoil layer). In wet heathlands and matgrass swards, the restoration of the original hydrology, for example by blocking drainage ditches, is also applied (Dorland et al. 2004).

The influence of cutting on botanical composition of meadow communities is discussed by many authors, e.g. Jakrlová (1997), Lepš (1999), Hansson & Fogelfors (2000), Bakker et al. (2002), Wahlman & Milberg (2002), Marriott et al. (2003), Fynn et al. (2004) and others; the impact of cutting on botanical composition of prairies studied for example Collins (1998), Maron & Jefferies (2001) and others. However, findings have not been consistent across the various studies. It is necessary to keep in mind that plant species are mostly site specific and, therefore, general trends in the effects of low-impact management of grasslands are still difficult to identify (Louault et al. 2005). Available field data suggest large differences in regrowth of individual plants after mowing (Klimeš & Klimešová 2002). The year-to-year variation of climatic parametres may also be an important driving factor in community dynamics (Herben et al. 1993).

The species capable of quick regeneration after cutting, e.g. grasses, are always favoured. In contrast, the competitiveness of slow-growing species which flower and ripen in late summer gradually decreases, resulting in the decrease of their vitality and in their slow withdrawal from the stand (Rychnovská et al. 1985).

Small herbs should increase slightly but steadily in mown plots, whereas they should first decrease and thereafter remain unchanged in unmown plots. Tall herbs should slowly increase in cover, especially on unmown plots, but they cannot completely exclude grass invasion (Huhta et al. 2001). By mowing, larger plants lose a higher proportion of their above-ground biomass than small plants (Klimeš & Klimešová 2002). Intensive mowing leads to increased investment in below-ground components - roots, rhizomes and detached roots in the second year following the commencement of mowing (Dickinson & Polwart 1982).

The biomass removal following cutting and also burning are essential and they have two reasons: the decrease of nutrient content over time and an immediate change of light pentration into the stand. Both these factors enhance the species richness (Schaffers 2002).

The results of Dorland et al. (2004 and 2005) show, that the combination of sod cutting and liming would create suitable soil conditions for the germination and establishment of endangered plant species of dry and wet heathlands. The success of restoration projects of these areas can thus be increased. Sod cutting is commonly used as an effective measure to remove excess nutrients (Dorland et al. 2004).

Antonsen & Olsson (2005) performed a field experiment with mowing and spring burning of a boreal former hayfield in Norway. Mowing increased the plant species richness and diversity, mainly by enhancing the number and frequency of forb species. Small-statured forb species were promoted by mowing, whereas tall leafy grasses declined. Mowing is therefore the most efficient way of enhancing biodiversity in this case. The results also show that mowing-mediated changes in above-ground plant communities may stimulate below-ground symbiotic micro-organisms, potentially resulting in a positive feedback on ecosystem development (Antonsen & Olsson 2005).

Huhta et al. (2001) surveyed the changes in plant cover and species composition of a formerly grazed meadow (recently abandoned) in northern Finland. The cover of grasses decreased and herb cover remained unchanged in both unmown plots and plots mown and raked annually in August. Mowing and raking significantly reduced litter accumulation and increased the number of ground layer species. During a simulation period of 30 years, the abandonment led to a decrease in the cover of small herbs, to a slight increase in the cover of tall herbs and to a slight decrease in the cover of grasses. In contrast, the cover of small herbs on the mown plots remained unchanged or slightly increased during the course of simulation. These results suggest that mowing late in the season is primarily a management tool for the maintenance of the existing species diversity and composition. However, it may not be an effective restorative tool to induce overall changes in the resident vegetation of abandoned grass-dominated meadows. Grazing or mowing early in the season may be more effective in this respect. (Huhta et al. 2001).

Pavlů et al. (2007) observed the increase in total plant species richness in grazing and cutting treatments in Jizerské Mts. after seven years, whereas the species number was reduced in untreated plots at the end of the experiment. Tall forbs (*Aegopodium podagraria*, *Galium album*, *Anthriscus sylvestris*, *Cirsium arvense*) as well as tall grasses (*Elytrigia repens* and *Alopecurus pratensis*) were more abundant in the control. The abundance of tall grasses and tall forbs reflected the intensity of the management in the order control > extensive grazing, first cut followed by extensive continuous grazing > intensive grazing > first cut followed by intensive continuous grazing. Prostrate forbs, on the other hand, increased their cover with increasing intensity: first cut followed by intensive continuous grazing > intensive grazing > first cut followed by extensive continuous grazing > extensive grazing. This study has shown that the restoration of grazing management on abandoned mesic grassland altered the plant species composition toward increased proportions of short grasses and prostrate forbs. (Pavlů et al. 2007).

Mašková (2008) found out, that the applied treatments did not appear to affect the number of vascular plant species during the first six seasons. Starting from the seventh season, the total number of species tended to differentiate according to the treatment. Shannon diversity showed the greatest increase in the mulched treatment during the first four seasons. The lowest diversity values were consistently found in the mown treatments compared to both the mulched and fallow plots. The mulched treatment showed a significantly higher proportion of forbs and a lower proportion of grasses than the mown and fallow treatments. The lowest Shannon diversity in mown plots does not correspond with those obtainned in published studies (e.g. Moog et al. 2002) and documented an increase in species diversity, especially among the dicotyledonous plant species, in regularly mown meadows. In experiment of Mašková (2008) the differences between the treatments have not been significant, thus the confirmation of their assumptions requires a longer period.

Maron & Jefferies (2001) performed a five-year experiment in enriched coastal prairie in California. They combined the effects of mowing and biomass removal on the total soil nitrogen, net rates of mineralization, nitrogen retention, and the species richness and biomass. The mowing and biomass removal resulted in a dramatic change in the species assemblage, from exotic annual grasses to a mixed exotic/native forb community composed primarily of perennials. The species richness was significantly greater in the treated plots than in the control. Weedy exotic grasses diminished in abundance and both native and exotic forb species increased. In mowed vs. control plots, there was significantly less mean aboveground biomass, but significantly greater belowground biomass (Maron & Jefferies 2001).

Collins (1998) performed long-term field experiments in North American prairies to assess the effects of fire, nitrogen addition, and grazing or mowing on plant species diversity. In one experiment, richness declined on burned and fertilized plots, whereas mowing maintained diversity under these conditions. In the second experiment, loss of species diversity due to frequent burning was reversed by bison, a keystone herbivore in North American grasslands. Thus, mowing or the reestablishment of grazing in anthropogenically stressed grasslands enhanced biodiversity. Furthermore, different type and timing of disturbances may influence the abundance of dominant species (Collins et al. 1998).

Fynn et al (2004) evaluated more than fifty year experiment in subtropical mesic grassland in South Africa. Forb species richness declined by 25% in sites mown twice in summer relative to sites mown in early summer only. The disturbance was necessary to achieve the maximum grass species richness presumably by removing the litter and increasing the availability of light. The interaction of time of mowing in summer (early versus late) and time of burning during the dormant period (spring versus winter) had the most dramatic effect on species richness. Time of burning had no effect on richness in sites mown in early summer, but winter burning resulted in a dramatic decline (27 - 42%) in richness in sites mown in late summer. (Fynn et al. 2004).

It is often stated, that the production increases and the the diversity decreases after long-term high fertilization (Elberse et al. 1983). However, if we evaluate the genic richness of meadows, the reliance only on species diversity could be misleading in cases. The high number of common species could be present in fertilised meadows with high species diversity, whereas unfertilised meadows with low species diversity could be represented by the rare species (Losvik 1993).

Competition seems likely to contribute to the productivity–diversity relationship because the productivity interacts with resource levels. Increased soil resources lead to increased productivity, but as plants grow larger, shading reduces light availability. Competition for both resources therefore changes with productivity (Rajaniemi 2002). With increased nutrient availability, nutrient limitation weakens and competition for light becomes the decisive factor. Competition for light appears to be more asymmetric than competition for nutrients, and consequently, it is more likely to drive inferior species to extinction. This is, together with seedling recruitment limitations, the most important cause of a decrease in species richness under high nutrient levels (Lepš 1999).

Species-rich grasslands that become enriched with nitrogen often suffer decreases in species richness, increases in plant biomass, and invasion by weedy exotic species. Suitable techniques to restore enriched grasslands and reestablish native communities are increasingly needed (Maron & Jefferies 2001). Smits et al. (2008) performed the fertlisation experiment in calcareous grasslands in The Netherlands. The general trend is a clear increase in number of plant species after the fertilization ceased (at most after about 10-15 years), followed by a slow decrease. The Npk-, and nPk-treatments (considerable differences in N and P ratio) in their experiment showed the same trend after the termination of the experimental additions which took 9 years, but the number of species was lower compared to unfertilized control. The recover of the same number of plant species as in the other plots took 8 years in the case of Npk-treatment and even 12 years in the case of nPk-treatment. In experiment of Zechmeister et al. (2003) there were significant negative correlations between plant species richness and mowing intensity and intensity of fertiliser application. Vascular plants showed the highest species richness at an intermediate nitrogen supply. The total plant species richness decreased with increasing nitrogen supply (Zechmeister et al. 2003). In experiment of Soudzilovskaia & Onipchenko (2005) the biodiversity estimated by the Shannon-Wiener index decreased under N + P treatment in alpine lichen heath plant community in northwestern Caucasus in Russia. The total numbers of forb plants per plot were not influenced by treatments, while the total number of graminoids increased by 20% in response to P treatment and threefold in response to N + P treatment (Soudzilovskaia & Onipchenko 2005).

On the other hand, the opposite results are mentioned in some papers. Neither Hadincová et al (1997) nor Štursová (1974 and 1985) found any changes in species composition in species-poor matgrass swards above timberline in Krkonoše Mts. after NPK application. Only the quantitative ratio of present species changed and therefore the total stand production changed also (Hadincová et al. 1997). As well as Roem et al. (2002) observed that species richness was not significantly influenced by the nutrient availability treatments (N, P, NP).

The effect of fertilization on the biomass production is not consistent as well as the effect of mowing. Willems et al. (1993) observed different responses of above-ground production and species number following fertilization in chalk grassland. They used different combinations of nutrients (N, P, K, M - micronutrients). Addition of N and P doubled the above-ground production, and the species richness dropped cca 50 % compared with the control. But the increase of the biomass does not necessarily mean the decrease of the species number in some combinations of fertiliser, e.g. the more productive PM plots (phosphorus plus micronutrients) retained significantly more species than the less productive NM plots (nitrogen plus micronutrient) (Willems et al 1993). The additions of nitrogen fertilizer to experimental plots in successional grasslands in Southwest Michigan (Foster & Gross 1998) over two growing seasons increased the plant production as indicated by increases in both living plant biomass and the litter biomass. Increased productivity reduced species richness by effectively preventing the seedling establishment of the subordinate forb species. The densities of biennial and perennial forbs were reduced significantly by nitrogen and litter. Litter significantly reduced species richness by the same amount in fertilized and unfertilized plots (Foster & Gross 1998). Øien (2004) performed full-factorial NPK fertlization in rich-fen vegetation in Central Norway. The above-ground biomass differed from site to site after two years of fertilization. At the community level, the results indicate multiple limitation by N and

P in the two least productive rich-fen communities: one characterized by small sedges and herbs, and the other by high abundance of *Menyanthes trifoliata* and tall sedges. Increased nutrient availability had no effect on a more highly productive, tall-growing, spring-influenced community, indicating no nutrient limitation. Thus, the greatest increase in biomass was observed at the least productive community, whereas at the most productive meadow fertilization did not lead to any increase in either biomass production or shoot density. Generally, adding N or P alone had very little effect on the biomass production, but when N was added together with P, biomass production increased (Øien 2004).

van der Hoek et al. (2004) studied the biomass production and species composition changes in a species-rich fen meadow (*Cirsio dissecti-Molinietum*) in central Netherlands. At one site (where the most abundant species are *Molinia caerulea* and *Cirsium dissectum* and small sedges such as *Carex panicea* and *C. hostiana*) N addition boosted biomass production, but only during one year. The species composition was not changed. P fertilization increased the biomass production in the second season and changed the species composition from a vegetation dominated by *Carex panicea* to a grassland community with abundant *Holcus lanatus*. At second, little drained margin site, neither the N- nor the P-supply affected total biomass production in the first and subsequent years. Although P-addition did not change total biomass, it increased the biomass of *H. lanatus*, whereas the biomass of formerly dominant *C. panicea* decreased (van der Hoek et al. 2004).

In species-poor grassland of association *Sileno-Nardetum plaerozietosum* (experiment of Hadincová et al. 1997) the production fluctuated within several years, namely in the first years of the monitoring. Later the fluctuation declined. The biomass of both fertilised and nonfertlised plots decreased (both variants were cutted in conjunction), but the decrease in nonfertilised plots was more intensive. The stand increased the production not until the third year following the fertilization. The comparison of maximum production in fertilised and nonfertilised plots indicates, that the stand responded to fertilizing above all in combination with favourable year (Hadincová et al. 1997).

The species *Bistorta major* becomes an expansive species in higher altitudes of Krkonoše Mts. thanks to the abandonment of traditional management and gradually it completely breaks up the original connected sward. In the experiments of Hadincová et al. (1997) *Bistorta major*, which had a high production at the beginning of the monitoring in species-poor meadows, quickly receded being intensively cutted and fertilised, whereas it did not respond to management changes in species-rich meadows, where it figured only as a minor species. Dominant species always answered much faster to management changes than other species, because the former conditions were optimal for them. Thus, dominant species responded to the changes of these optimal conditions very sensitively and quickly (Hadincová et al. 1997).

Eight years after the start of annual mowing in unfertilised plots in calcareous grassland in The Netherlands (Willems 1985) the rosette plants decreased suddenly in cover percentage, whereas the solitary forbs increased. After a period of three years the opposite situation could be seen. NPK application in yearly mown plots resulted, within four years, in a decrease of all growth forms except of tall graminoids and solitary forbs. In plot with high amount of N-fertilizer the tall graminoids increased enormously in cover, from 30 to 70%. Solitary forbs reached 40% in the fourth year. Fertlizer treatment ceased 9th year of the experiment, but tall graminoids reached the highest cover percentage (96%) one year later. From this year onwards the cover percentage of tall graminoids decreased dramatically to less than 50% during next four years. The dominance (cover percentage more than 50%) of tall graminoids, climbing forbs and tall woody shrubs is clearly related to a decreasing species diversity, whereas the dominance of rosette plants with leafless flowering stalk show no negative effect on species richness (Willems 1985).

Fourteen years after cessation of fertilizer with continual cutting did Olff & Bakker (1991) observe a strong decrease in standing crop in two fields on peaty soil, but in a field on sandy soil the initial production did not decrease. The species composition changed gradually in all the fields. The species-richness increased significantly over the study period (Olff & Bakker 1991).

Bakker et al. (2002) examined *Nardo-Galion saxatilis* communities (characteristic of oligotrophic soils) in The Netherlands under low-intensity farming. The various cutting regimes revealed different effects 25 years after cessation of fertilization. The study site still harbours several species characteristic of eutrophic soil and few species characteristic of oligotrophic soil after 25 years of annual cutting and removal of the swath. The soil seed bank harbours only few target species. The abandonment resulted in the establishment of shrubs, although it takes more than 20 years to completely cover the site. Mulching resulted in tall forb communities. Different periods of annual haymaking regimes did not reveal different plant communities. Haymaking twice a year resulted in the largest removal of nutrients and a decrease in species number after an initial strong increase (Bakker et al. 2002).

1.3 The forest-free areas of Šumava Mts.

The anthropogenically dependent forest-free areas in Šumava Mts. are very important, valuable and inseparable from this region. These communities concern above all wet, waterlogged and peaty meadows or meadow mires, mesic meadows and pastures, and semixerophilous grasslands and heathlands. These forest-free areas are not original on the whole, but they often have considerably natural character (Zelenková 2000).

Whereas the primary forest-free areas are usually considered as something statically unchangeable and functioning as a refugium of relict biota, at secondary forest-free areas is emphasized the human impact, inconstancy, dynamics and the role of secondary succession. Otherwise, this strict classification well describes extremes (rock versus agricultural enclave), but most types of forest-free areas in Šumava Mts. are transitions. Their character is more likely covered by this classification. These zones mostly are not wide. They arose after the disturbation of wood (so by the secondary succession), but since that time they were protected against the spread of wood by climatic effects, and thereby they became an important refugium of paleochoric types of vegetation. These areas presumably expire over time (hundreds of years), but meanwhile new localities arise in the neighbourhood and enable spatially discontinuous, but chronologically continual survival of paleochoric vegetation. The traditional classification of vegetation phenomena into original and derived, natural and man-made, primary and secondary, could become more likely an aggravation in practice (Sádlo 2001).

Forest-free areas arose by deforestation when Šumava Mts. was colonized. The traditional agricultural management of meadows and pastures throughout the entire border region of the Šumava Mts. stopped shortly after the withdrawal of the German populations at the end of World War II, leading to a radical change in the land use. Grasslands degraded as a consequence of the absence of appropriate farm management, or, conversely, due to devastation by intensive agricultural practices (Zelenková 2000).

Nowadays, still more often it is possible to see grazing cattle on large meadows and pastures, which lay waste from the postwar period until recently, eventually were used as a military areas (Matějková 2001).

1.4 The description of plant communities

1.4.1 Intermittently wet *Molinia* meadows

The alliance *Molinion caeruleae* Koch 1926 - intermittently wet unfertlized meadows with strongly fluctuating groundwater-level, belongs to the order *Molinietalia caeruleae* Koch 1926 - grasslands of wet habitats, which is included in the class *Molinio-Arrhenatheretea* Tüxen 1937 - agriculturally managed meadows and pastures on wet or mid-wet sites (Slavík et al. 1988). Chytrý et al. (2007) distinguished two associations within the alliance *Molinion*: 1) *Molinietum caeruleae* Koch 1926 occurs on wet soils from lowlands to submontane areas which regularly contain calcium carbonate and 2) *Junco effusi-Molinietum caeruleae* Tüxen 1954 occurs on more oligotrophic and acidic soils in higher colline and submontane altitudinal belts. This second association is the case of the studied community.

The cover of *Junco effusi-Molinietum caeruleae* is relatively constant and it fluctuates around 90%. The most frequent dominants or subdominants are *Molinia arundinacea* and *M. caerulea*, *Sanguisorba officinalis*, several species of narrow-leaved fescue grasses, e.g. *Festuca rubra* agg. and *F. ovina* and the matgrass (*Nardus stricta*). The species richness of vascular plants mostly ranges between 35-50 species on the area of 16-25 m^2. The cover of mosses balances between 0 and 90%. Its most frequent dominants are *Aulacomnium palustre*, *Climacium dendroides* and *Rhytidiadelphus squarrosus* (Chytrý et al. 2007). This association represents extensively managed, intermittently wet unfertilised meadows on acidic soils with strongly fluctuating groundwater-level. The threat of these communities consists in eutrophication (fertilization), drainage, abandonment and subsequent forest invasion (Chytrý et al. 2001). The groundwater-level typically occurs near the soil surface for most of the year and it desclines under 50 cm in late summer. The surface flooding never occurs during the year (Stanová & Valachovič 2002). These meadows used to be extensively cut (usually once a year in late summer) or grazed, but they have never been fertilized. The management is often missing recently, thus meadows are overgrown, especially with *Molinia*, and the number of species declines (Chytrý et al. 2007).

Abandonment or fertilization leads to a loss of species diversity and to an enhancement of single clonal species (e.g. Lepš 1999, Lepš 2004, Klimeš & Klimešová 2002). *Molinia* meadows have above all a conservational importance. Their productivity is low, so their former utilisation as an extensive meadows and pastures is not profitable nowadays. The community *Junco effusi-Molinietum caeruleae* is rare in the Czech Republic and it occurs only in fragments. This habitat is occupied by many endangered species. It is threatened mainly by abandonment, eutrophication and drainage. Hence, conservation measures are necessary (Chytrý et al. 2007).

In contrast, the strong expansion of *Molinia caerulea* especially in western and northwestern Europe represents a threat to biodiversity of heathlands, ombrotrophic bogs and other habitats. Accordingly, many papers are focussed on the suppression of this species to encourage the development of dwarf shrub vegetation (*Calluna*). *Molinia* was found to be highly competitive in many different situations compared to other species (Sansen & Koedam 1996).

In both areas of Europe mentioned above, the focus of research has been on the behaviour of the *Molinia* species dominant in different grassland types under different management treatments.

The revegetation process and behavior of *Molinia* studied Sansen & Koedam (1996) in Belgium in sod-cut wet heathlands where *Molinia* dominates. *Molinia* tended to dominate again within a few years. Whereas the other species were affected in their development by either groundwater regime, soil acidity, nutrient availability or cut depth, *Molinia caerulea* was highly competitive in all situations. In contrast to *Erica tetralix*, the recovery of *Molinia* was generally less influenced by the soil variables. Only initial P-availability influenced its competitive ability (Sansen & Koedam 1996).

The decrease of *Molinia* cover following cutting has been frequently discussed (e.g. Milligan et al. 2004). The only treatment that had consistent effects (Milligan et al. 2004) in the univariate analysis of variance was cutting, where there was increased bare ground, reduced vegetation height, increased species diversity and reduced *Molinia* cover. Cutting three times had the greatest effect, maintaining a reduced *Molinia* cover over four years. Grazing generally produced vegetation which had a greater moorland species complement. Where grazing was restricted the vegetation had a greater component of *Molinia* and other acid grassland species. The most effective treatment was grazing in conjunction with cutting thrice, which maintained a low *Molinia* cover for longest and had less variation in moorland species in the fourth year (Milligan et al. 2004).

The reduction mainly occurs following early cutting (Rychnovská 1985, Blažková & Řehořek 1999). Similarly, the decrease of biomass after cutting has been observed many times (e.g. Klimeš & Klimešová 2002, Hakrová & Wotavová 2004, Milligan et al. 2004). The frequency and severity of defoliation appeared to be more important than the timing (Grant et al. 1996, Jakrlová 1997, Milligan et al. 2004). In contrast, Hansson & Fogelfors (2000) observed an increase of *Molinia* density following cutting. The species composition, species richness and diversity are mainly affected positively by mowing (Rowell et al. 1985, Jongepierová et al. 1994, Lepš 1999). Nevertheless, Jakrlová (1997) observed the reduction in species number after cutting three times per year.

The *Molinia* biomass increased to varying degrees after the application of fertiliser (Aerts & Caluwe 1989, Lepš 1999, Klimeš & Klimešová 2002, Tomassen 2003) and the different effects of the NPK components and of varying nitrogen : phosphorus ratios were also recorded (Roem et al. 2002, Güsewell 2005). In experiment of Klimeš & Klimešová (2002) the fertilization did not affect plant growth. However, shoots of fertilized *Molinia* developed larger storage organs containing a higher amount of storage carbohydrates. Grazing often has an important effect on *Molinia* cover and community composition (Grant et al. 1996, Marrs et al. 2004). Other management treatments tested in western Europe have had intermittent or marginal success in *Molinia* control and in species richness enhancement; e.g. graminicide application (Milligan et al. 2004), different light conditions (Güsewell 2005), liming, acidification and glucose application (Roem et al. 2002), burning, burning followed by cutting, burning followed by herbicide application (Ross et al. 2003), removal of *Molinia* litter by raking or application of *Calluna* seed (Marrs et al. 2004), controlled conditions of soil aeration and ground-water movement (by Webster 1962).

1.4.2 Species-poor grasslands dominated by *Carex brizoides*

Carex brizoides L. (family *Cyperaceae*) is the species of wet forests, mainly alder carrs, the edges of brooks and uncut meadows from lowlands to montane areas. It is abundant in Bohemia and northern Moravia, but locally it is rare or absent (Kubát et al. 2002). In terms of phytocoenology, *Carex brizoides* occurs in many vegetation units, above all in forest communities: it is both the diagnostic and dominant species in hardwood forests of lowland rivers (the alliance *Alnion incanae*) and in acidophilous beech forests (the alliance *Luzulo-Fagion*), dominant in wet acidophilous oak forests (the alliance *Genisto germanicae-Quercion*), further in willow carrs (the alliance *Salicion cinereae*), in Polonian oak-hornbeam forests (the alliance *Carpinion*) and in birch mire forests (the alliance *Betulion pubescentis*). *C. brizoides* often grows in wet *Cirsium* meadows (the alliance *Calthion*), which is divided into several associations (Chytrý et al. 2001).

The studied *C. brizoides* community corresponds in all parameters to the association *Scirpo sylvatici-Caricetum brizoidis* Kučera et al. 1994 (wet heath meadows with *Carex brizoides*). It is characterized by the only dominating species - strongly clonal grass determining the physiognomy of the stand. *Carex brizoides* forms connected characteristically curly stands, which are only rarely overgrown by some accompanying dicotyledonous species. The number of vascular plant species on the area of 16-25 m^2 ranges between 20-35 depending on the age of the stand. The moss layer is mostly missing due to strong litter accumulation, which is a characteristic feature, preventing the establishment of most species and delaying succession of woody vegetation. Species-poor grasslands dominated by *Carex brizoides* develop from abandoned wet meadows on acidic, nutrient-poor soils and represent blocked successional stage of the evolution from abandoned meadows to alder forests. Some of these stands remain in a relatively stable state for several decades. In the entire border regions, this habitat occurs on farmlands abandoned more than 50 years. Nowadays, it has no practical usage and it is rather valuable for the soil protection function. The importance for the nature protection consists above all in blocking the succession toward bushy and tree formations and in the maintenance of forest-free vegetation. However, *Carex brizoides* is an expansive species, positively responding to eutrophicaton of the environment and it spreads in woods during last decades (Chytrý et al. 2007).

Vacková (1997) and Hakrová & Wotavová (2004) based on field experiments with mowing of *Carex brizoides* confirmed, that both total biomass and the biomass of *Carex brizoides* decrease after mowing. Nevertheless, the biomass of *Carex* markedly increased in one of four monitoring plots of Hakrová & Wotavová (2004). The botanical composition of those stands did not change so much. A few of new species appeared that were present in the vicinity, such as *Equisetum sylvaticum* and *Rubus idaeus*. Blažková & Hruška (1999) assumed a slow recovery of this long-term successional stadium toward original meadow communities after the restoration of cutting. The achieving of effectual restoration requires the combination of early summer cutting and the additional autumnal fertilizing with mulch, eventually the burning of the litter. These treatments should encourage the spread of accompanying species and suppress vegetatively spreading *Carex brizoides*.

Matějková et al. (2003) studied what kind of vegetation community is preferred by the cattle on the semi-natural species-poor pasture in Šumava Mts. *Carex brizoides* swards were grazed only sparsely, at the beginning of the season. Cattle dung was found most frequently in the *Nardus stricta* and *Carex brizoides* swards, which were preferred by the cattle as resting areas (Matějková et al. 2003).

1.4.3 Species-poor grasslands dominated by *Calamagrostis villosa*

Calamagrostis villosa (Chaix) J. F. Gmelin (family *Poaceae*) is the species of acidophilous woods, tall-forb grasslands and forest clearings from submontane to subalpine areas. It is common in higher altitudes and rare or even absent in lowlands and colline landscape (Kubát et al. 2002). *Calamagrostis villosa* is an expansive perennial grass, which occurs naturally in many vegetation units. However, its occurrence in forest habitats was not the point of interest, thus only the description of nonforest habitats follows. *C. villosa* is the diagnostic, dominant and constant species of the alliance *Calamagrostion villosae* Pawłowski et al. 1928 (subalpine tall-stalk grasslands), which occurs mainly around and above the timberline in Krkonoše Mts., Králický Sněžník and Hrubý Jeseník Mts. Fragmentary distributed species-poor stands without diagnostic species were also recorded in several places in Šumava Mts.

The alliance *Calamagrostion villosae* is devided into 3 associations accordig to the dominant species, which is either *Molinia caerulea*, *Calamagrostis villosa* or *Deschampsia cespitosa.* The second of mentioned associations is *Crepido conyzifoliae-Calamagrostietum villosae* (Zlatník 1925) Jeník 1961 - subalpine grasslands with *Calamagrostis villosa*. Representative stands of this association are relatively species-rich and mostly quite connected. The typical habitat of these *C. villosa* stands are leeward sides of slopes 20-30(-45)$^{\circ}$, most frequently in altitudes of 1200-1400 m., eastward and southeast orientated, generally on the upper slopes of glacial cirques, where pronounced snow cover accumulates in winter and protects plants against frost. The species richness of vascular plants mostly ranges between 10-25 species on the area of 16-25 m^2. The moss layer is developed only slightly due to high cover of the herb layer. The cover of mosses reaches only about 3% or it is quite missing.

Nowadays, an agronomical importance of species-poor grasslands with dominating *Calamagrostis villosa* is negligible. They were utilized as pastures or extensive meadows in the past, but nowadays, they have the importance mainly for the soil protection, water utilization and for the protection of rare species, e.g. *Anemone narcissiflora*, *Crepis conyzifolia*, *Pulsatilla alpina* subsp. *austriaca* and the endemit of Krkonoše Mts. *Sorbus sudetica*. Considering the strict protection of all localities, the association *Crepido conyzifoliae-Calamagrostietum villosae* is probably not threatened by direct impacts. However, the eutrophication together with acid rains seems to reduce the diversity of dicotyledonous species and the gradual prevail of *Calamagrostis villosa* takes place. Species-rich stands are rare and they probably degrade toward species-poor types. In recent decades, *Calamagrostis villosa* expands into low-rise, species-poor, oligotrophic habitats as a result of the nitrogen availability, acid rains and the cessation of management (Chytrý et al. 2007).

Chytrý et al. (2007) do not distinguish the association *Sileno vulgaris-Calamagrostietum villosae* Jeník et al. 1980 anymore (due to the absence of diagnostic species) which was described by its authors as a species-poor association vicariant to species-rich association *Crepido conyzifoliae-Calamagrostietum villosae* mentioned above. Species-poor stands occur on similar habitats as species-rich communities, but they occupy considerably larger areas. Besides the mountains mentioned above, they occur in a very depleted form in the Šumava Mts. and they represent the transition to the vegetation of the forest clearings (Chytrý et al. 2007). The studied species-poor community is probably closest to this association, but classification is problematic in this case.

In central Europe, forest decline and subsequent timber extraction due to atmospheric $S0_2$ pollution have lead to the creation of new habitats available for colonization. Some species have adapted to the changing environment better than the others; among the most successful is *Calamagrostis villosa* (Pyšek 1990). *C. villosa* is a strong competitor under such extreme ecological conditions due to (1) easy distance spread due to low caryopsis weight, (2) capability of producing large amount of biomass, (3) the accumulation of the litter on the soil surface which contributes to suppression of other species, (4) rapid vegetative spread by extensive rhizome system, (5) low nutrient requirements, and (6) tolerance to air pollution (Pyšek 1994). Expansion of *C. villosa* in mountain regions of the Czech Republic causes considerable difficulties to the forest management. The negative effects may be divided in several groups (Pyšek 1993):

1. Replanting efforts

Competition from grasses may be a cause of reduction in tree or shrub saplings' cover. Lokvenc (1971) estimated that the 50% cover of replanted *Pinus mugo* stands above the timberline was reduced to 8.7% by the presence of *C. villosa* stands. Consequently, 50% of costs spent on replantation were lost. Furthermore, dense *C. villosa* stands provide suitable habitats for small mammals, namely field vole (*Microtus agrestis*) and common shrew (*Sorex araneus*) which can damage tree saplings considerably by browsing.

2. Decrease in species diversity

Although the number of plant species present in deforested sites does not substantially differ from the species richness of original forests, species diversity is reduced due to the dominance of *C. villosa*. Natural revegetation of woody species is prevented and succession to forest communities is extremely slow or even impossible (Pyšek 1994).

3. Hydrological consequences

After timber extraction, the deforested sites become wet or locally waterlogged. On the other hand, during the spring snow melting, most of the water flows on the settled-down surface of C. *villosa*; consequently, infiltration into soil is lowered and a large proportion of precipitation is lost from the site. Due to the uniformity of vast areas, the period of melting is getting shorter which may increase the danger of flooding.

4. Recreational aspects

Obviously, vast uniform areas of deforested sites are less attractive for leisure activities; their value decreases due to aesthetic reasons, mechanical barriers restricting access and making walking difficult (logs, fallen trunks, depressions), and the sites are more exposed to bad weather.

Positive effects of the presence of C. *villosa* stands are (a) prevention from soil erosion, and (b) closed nutrient cycling maintaining the soil chemicals in the site (Pyšek 1993).

In terms of Grime's strategy scheme, *C. villosa* is considered to be a C-strategist in its natural undisturbed habitats. In secondary habitats, the CSR-strategy would describe better the real species behaviour (Pyšek 1994). Plant mass production is higher in deforested sites than in forest habitats (Pyšek 1991), due to the increase in available light (Pyšek 1993). Higher species richness was found in the moist meadow habitats invaded by *C. villosa* than in the deforested sites completely dominated by this species (Pyšek 1990). Pyšek (1994) also proved the decrease of both tiller height and thickness of the litter layer with soil moisture. Species diversity H' (expressed as Shannon index) was not related to the moisture level but it was shown to be correlated positively with available light and negatively with soil acidity; these two predictors explained 60.9% of variance in the data set where H' was concerned and 76.3% where the number of species was taken into account. Nitrogen level had not a significant effect on the community composition. (Pyšek 1994).

Hejcman et al. (2009) analysed plant species composition of *C. villosa* community by RDA. The species composition was significantly influenced by cutting but not by fertilization. Cutting reduced the cover, biomass, sward height and tiller density of *C. villosa*. Panicles of *C. villosa* were recorded only in plots with cutting management. The introduction of regular cutting management had a decisive effect on plant species composition as the interaction of year with cut was significant. High year-to-year variability in plant species composition was also revealed; the effect of year was significant. Morávková-Lipnická (1991) also revealed reduction both total above-ground plant mass and root mass, but only in the year of treatment.

This result, however, strongly depends on at which time of the growing period the perturbation is made. In contrast to cut, fertilization in experiment of Hejcman et al. (2009) had a negligible effect on plant species composition. Biomass production of *C. villosa* was relatively stable in the control and fertilised plots. Thus, *C. villosa* was recognised as a defoliation-sensitive species and this sensitivity cannot be overcome by an increase in N supply. Recent expansion of *C. villosa* in sub-alpine grassland can be attributed to the long-term succession that has followed the cessation of agricultural management and to an increase in the nitrogen availability in recent decades (Hejcman et al. 2009).

As shown by Morávková-Lipnická (1991), liming encouraged C. *villosa* by an increase in living above-ground plant mass in two years following its application; among underground organs, the increase in root mass was balanced by a decrease in rhizome mass. Herbicide applications reduced above-ground or both above- and underground production. Moreover, the treatment affected the spatial structure of C. *villosa* population and resulted in dense clustering of tillers which was reflecting reduced penetration of rhizomes into the surrounding toxic environment (Morávková-Lipnická 1991).

Pyšek (1990) observed, that the number and total biomass of the other species decrease with the biomass of *C. villosa*. On the contrary, evenness J' of the other species increases with the *C. villosa* total biomass. The decrease in species number is compensated by the increase of species evenness. This is reflected by the pattern of species diversity *H'* which does not show any significant relation to the *C. villosa* productivity. Both the number of the other species and their total biomass are more closely related to the amount of *C. villosa* litter ($P < 0.01$) than to its living biomass ($P < 0.05$).

Success of other species in *C. villosa* stands therefore appears to be conditioned by (a) plant height and growth rate - potentially tall, tough forbs more readily penetrate the litter layer and then escape from its influence by rapid growth in height, (b) capability to spread vegetatively and produce their own litter, and (c) local site conditions (Pyšek 1990).

1.4.4 Wet *Filiendula* grasslands

The alliance *Calthion palustris* Tüxen 1937 - wet *Cirsium* meadows, represents the substitutional vegetation after alder carrs (the alliance *Alnion glutinosae*) and ash-alder alluvial forests (the alliance *Alnion incanae*). *Calthion* was traditionally devided into suballiance *Calthenion* (representing species-rich polydominant meadows with the occurrence of all diagnostic species) and *Filipendulenion* (representing species-poor monodominant stands with the dominance of vigorous species, namely *Cirsium oleraceum*, *Filipendula ulmaria*, *Geranium palustre* and *Scirpus sylvaticus*) (Chytrý et al. 2007). However, Chytrý et al. (2007) do not distinguish both suballiances in new interpretation anymore and the alliance *Calthion* is devided recently into 14 associations. One of them is the case of the studied community: *Lysimachio vulgaris-Filipenduletum ulmariae* Balátová-Tuláčková 1978 (wet *Filipendula* grasslands with *Lysimachia vulgaris*).

These stands with the dominant species *Filipendula ulmaria* are dense, high 1.5-2 m, often species-poor, markedly monodominant and without subdominants in lower herb layer. Other species could be locally present with higher cover, e.g. tall sedges (e.g. *Carex acuta*, *C. acutiformis*, *C. rostrata* and *C. vesicaria*), *Scirpus sylvaticus*, *Phalaris arundinacea* and dicotyledonous species (*Caltha palustris* and *Lysimachia vulgaris*). The species richness of vascular plants mostly ranges between 15-20 species on the area of 16-25 m^2. The moss layer is mostly missing. This association occurs on soils affected by frequent flooding of the soil surface, above all in spring and after fruitful rains. However, the soil is aerated at least during some part of the year. The stands develop from persistently wet abandoned meadows following eutrophication, eventually as a successional stadium in hydroseries during ponds and water channels filling. When the regular cutting management is restored, above all in nutrient-limited ecosystems, these stands could develop back toward the vegetation of wet meadows. The cessation of the management leads to the next decrease in species and enhanced nutrient input leads moreover to the spread of ruderal species. Wet *Filipendula* grasslands have an insignificant agricultural importance and they are not utilized presently. The association is not endangered in the Czech Republic (Chytrý et al. 2007).

Fojt & Harding (1995) evaluated the changes at the plant community and individual species levels in three Suffolk fens following hydrological disturbance and the abandonment (and combination). Mires which were derelict, but where the natural hydrology persisted, showed the least change in associated flora. Partial dewatering led to *Filipendula ulmaria-Angelica sylvestris* tall herb fen with few non-fen ruderals. All mires showed a decline in conservation importance as measured by indices of botanical 'quality'. The mires which showed least decline were those which retained their hydrological integrity, regardless of management status. Management was subordinate to hydrology in this case (Fojt & Harding 1995).

Tall herbs suffer, almost without exception, from mowing, but late mowing may not be as detrimental as early mowing. On the studied meadow in northern Finland Huhta et al. (2001) observed sparsely occurence of several tall species (including *Angelica sylvestris,* *Anthriscus sylvestris*, *Filipendula ulmaria* and *Geum rivale*) even on mown plots (Huhta et al. 2001).

Blažková & Řehořek (1999) assume, that the recover of cutting of wet *Filipendula* heaths should lead to more species-rich stands. Hakrová & Wotavová (2004) observed the change in the ratio of present species in cutted plots: both the biomass and cover of dominant *Filipendula ulmaria* markedly decreased, and, on the contrary, both the biomass and cover of sedges (*Carex* sp.) distinctively increased, without the change of total biomass (Hakrová & Wotavová 2004).

van der Hoek et al (2004) found out, that the total biomass of rich-fen meadows in The Netherlands with dominant *Filipendula* did not change under different fertlization treatments, namely N, P and NP, whereas Wilson et al. (1995) observed the increase of above-ground yield of certain forbs, particularly *Filipendula*, *Scabiosa* and *Thymus* in artificial chalk grasslands after nitrogen application. Inputs of nitrogen stimulated the growth of the sward, but there was no further increase in growth when N inputs were increased from 20 kg N . ha^{-1} . year^{-1} to over 80 kg N . ha^{-1} . year^{-1}. There was no evidence that increases in atmospheric nitrogen deposition will result in grass dominance and a loss of species diversity.

The growth was probably limited by the availability of phosphorus. In these circumstances, high levels of N deposition may prove a greater threat to ground water quality than to species composition (Wilson et al. 1995).

Pauli et al. (2002) conducted two-year field experiment in calcareous fen meadows of low productivity in north-eastern Switzerland. These meadows with habitat generalist *F. ulmaria* belong to the most species-rich plant communities in central Europe. The community was fertilised (N) and (NPK). The addition of nitrogen increased aboveground community biomass by 32% and full fertilization with NPK increased it by 70%. The number of generalist species increased after the addition of NPK (Pauli et al. 2002).

2. The goals of the thesis

The object of the study are four grassland phytocoenoses in an advanced successional stage in the northern part of the catchment of the brook Horský potok in the Šumava Mts., which have been abandoned after the withdrawal of the German populations at the end of World War II. The cessation of the management led to the degradation of the stands and to the decrease in species diversity. The goals of the work could be summarized into following points:

1) To test statistically the differences between all meadow types within all seasons in these variables: cover, the amount of above-ground biomass (dry-matter) and Shannon-Wiener index of diversity.

2) To describe the impact of different management treatments (cutting, liming and fertilizing) on the cover, the biomass and the diversity of studied meadow types within individual years.

3) To describe preferences and the relation of grassland species depending on different management treatments and years.

3. Methods

3.1 The area of interest

The study area is the complex of non-forest plant communities on the right side of the Lipno dam in the catchment of the brook Horský potok in the Šumava Mts., the Czech Republic. (Fig. 1). This area belongs to the Sudeten range, known for the abandonment of agricultural management after World War II and for the subsequent degradation and even loss of grasslands.

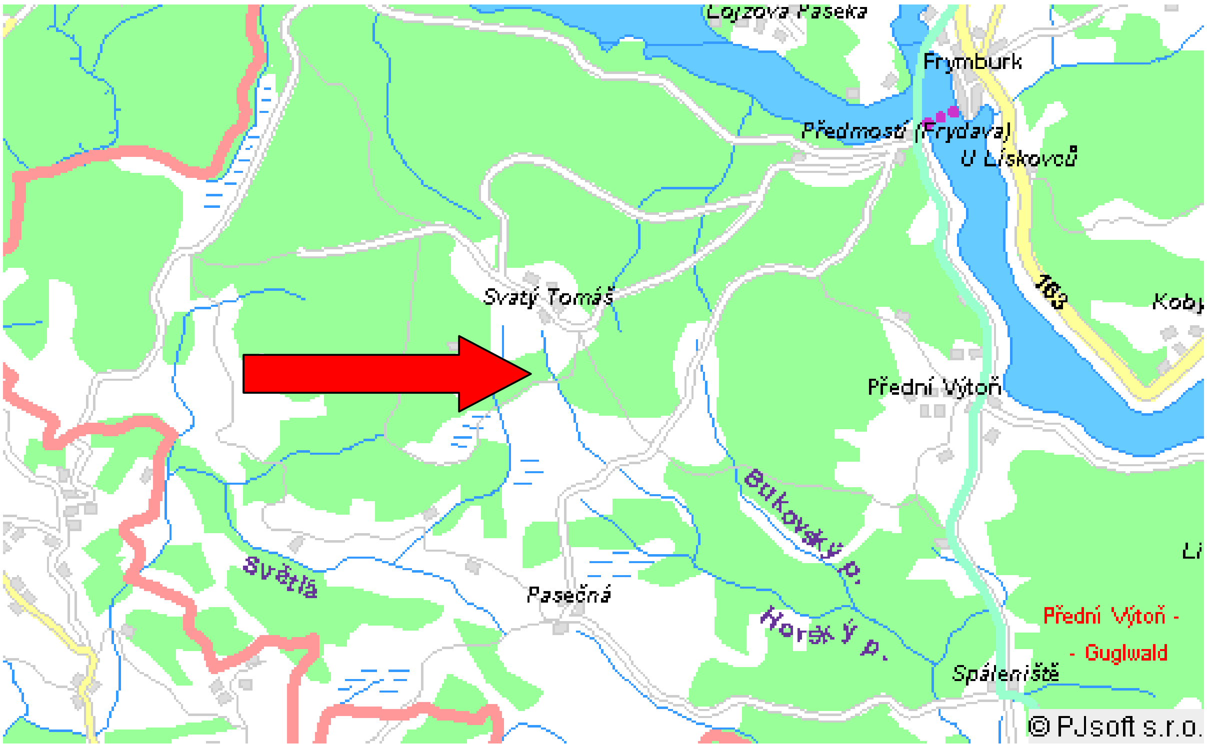


Fig. 1. The area of interest - the northern part of the catchment of the brook Horský potok on the right side of the Lipno dam.

Horský potok belongs to the drainage area of the Danube River. It flows through the the Czech Republic for the lenght of 8.6 km before entering the Austrian territory. The area of the Bohemian part of the catchment is 29.35 km^2 and it belongs to the Protected Landscape Šumava and the Biosphere reserve Šumava. This area represents predominately woodland (71%), unmanaged forest-free habitats (16%) and partly cut meadows (10%) (Procházka et al. 2001). The potential vegetation includes herb-rich beech forests, acidophilous beech forests in higher altitudes and spruce forests at the highest tops, in depressions waterlogged spruce forests and mires (Culek 1996). With reference to Neuhäuslové et al. (2001), the area studied occurs mainly in the zone of herb-rich beech forests with the occurrence of *Dentaria enneaphyllos* (*Dentario enneaphylli-Fagetum*) and only the top level belongs to the zone of spruce-beech forests (*Calamagrostio villosae-Fagetum*), which belongs to the group of communities of acidophilous beech forests and fir forests. The area is situated in oreophyticum - the extrazonal montane vegetation and flora, where thermophilous species are missing with few exceptions. The species are psychrophilous and mesophilous. The mean annual temperature is 4.3 ˚C and the mean annual precipitation is 1008 mm (České Budějovice Meteorological Station). The study sites lie at an altitude of 830 - 940 m a.s.l. and the distance between two furthest sites is 1 640 m in a straight line.

3.2 The description of studied meadows

The syntaxonomical position of the studied phytocoenoses is presented in the following overview. The phytosociological nomenclature follows Chytrý et al. (2007). The nomenclature of plant species follows Kubát et al. (2002). Studied communities are emphasized by bold type.

Class *Molinio-Arrhenatheretea* Tüxen 1937
  Alliance *Arrhenatherion elatioris* Luquet 1926
  Alliance *Polygono bistortae-Trisetion flavescentis* Br.-Bl. et Tüxen ex Marschall 1947
  Alliance *Cynosurion cristati* Tüxen 1947
  Alliance *Molinion caeruleae* Tüxen 1926
    Association *Molinietum caeruleae* Koch 1926
    Association *Junco effusi-Molinietum caeruleae* Tüxen 1954
  Alliance *Deschampsion cespitosae* Horvatić 1930
  Alliance *Calthion palustris* Tüxen 1937

Association *Angelico sylvestris-Cirsietum oleracei* Tüxen 1937

Association *Cirsietum rivularis* Nowiński 1927

Association *Angelico sylvestris-Cirsietum palustris* Darimont ex Balátová-Tuláčková 1973

Association *Crepido paludosae-Juncetum acutiflori* Oberdorfer 1957

Association *Polygono bistortae-Cirsietum heterophylli* Balátová-Tuláčková 1975

Association *Chaerophyllo hirsuti-Calthetum palustris* Balátová-Tuláčková 1985

Association *Scirpo sylvatici-Cirsietum cani* Balátová-Tuláčková 1973

Association *Scirpetum sylvatici* Ralski 1931

Association *Caricetum cespitosae* Steffen 1931

Association *Scirpo sylvatici-Caricetum brizoidis* Kučera et al. 1994

Association *Junco inflexi-Menthetum longofoliae* Lohmeyer ex Oberdorfer 1957

Association *Filipendulo ulmariae-Geranietum palustris* Koch 1926

Association *Lysimachio vulgaris-Filipenduletum ulmariae* Balátová-Tuláčková 1978

Association *Chaerophyllo hirsuti-Filipenduletum ulmariae* Niemann et al. 1973

Class *Mulgedio-Aconitetea* Hadač et Klika in Klika et Hadač 1944

Alliance *Calamagrostion villosae* Pawłowski et al. 1928

Association *Sphagno compacti-Molinietum caeruleae* Wagnerová in Berciková 1976

Association *Crepido conyzifoliae-Calamagrostietum villosae* (Zlatník 1925) Jeník 1961

Association *Violo sudeticae-Deschampsietum cespitosae* (Jeník et al. 1980) Kočí 2001

Alliance *Calamagrostion arundinaceae* (Luquet 1926) Jeník 1961

Alliance *Salicion silesiacae* Rejmánek et al. 1971

Alliance *Adenostylion alliariae* Br.-Bl. 1926

Alliance *Dryopterido filicis-maris-Athyrion distentifolii* (Holub ex Sýkora et Štursa 1973) Jeník et al. 1980

3.2.1 Wet *Molinia* meadow

The first meadow type is homogeneous stand with dominant *Molinia caerulea,* classified as the association *Junco effusi-Molinietum caeruleae* Tüxen 1954 - acidophilous *Molinia* meadows, belonging to the alliance *Molinion caeruleae*. The study site lies at an altitude of 940 m a.s.l. and the stand is species-poor, unmanaged in the long term and slightly ruderalized. This little meadow has an irregular shape, the area is less than half a hectare and it is more or less bordered with interspersed tree species (*Salix*, *Betula*). The exposition is southwest and the slope gradient is less than 10 degrees. North-easterly, the stand gradually loses its homogeneity and changes in the clearing vetation, which developed in the forest path under the electric line. However, *Molinia* is abundant here, but the species *Holcus mollis* strongly predominates in some places. The permanent monitoring plot was established in southwest tip of the little meadow (GPS: 48°38'14.459"N, 14°6'7.526"E) a few metres of the brook Horský potok. The pictures of the community are placed in the appendix (Fig. 31, 32).

3.2.2 The species-poor grassland dominated by *Carex brizoides*

Species-poor grassland dominated by *Carex brizoides* is the representative stand of the association *Scirpo sylvatici-Caricetum brizoidis* Kučera et al. 1994 - wet heath meadows dominated by *Carex brizoides*, belonging to the alliance *Calthion*. *C. brizoides* is the monodominant species, typically forming very dense stands preventing successful development of other species, thus the community is naturally species-poor (see table 3). The permanent monitoring plot was established in homogenous stand. In broader surrounding, the habitat forms the mosaic with other vegetation units, e.g. association *Scirpetum sylvatici* Ralski 1931 (wet meadows with *Scirpus sylvaticus*), other associations of *Calthion* and the clearing vegetation under the electric line with abundant *Holcus mollis*. These communities (and eventually others) are not spatially closely differentiated from each other, but they concur with variously smooth transitions. Thus, some parts of the grassland in the surrounding of the monitoring plot are not phytocoenologically distinctive, but they form transient communities (more vegetation units in one stand). Dispersed tree species are also present, mostly willows and birches. The stand with the monitoring plot is the representative homogenous community, spatially closely differentiated from neighbouring vegetation types. The study site lies at an altitude of 930 m a.s.l., the exposition is southwest and the slope gradient is less than 10 degrees. GPS localization is 48°38'9.049"N, 14°6'9.357"E. The brook

Horský potok is c. 10 m far from the monitoring plot. The pictures of the community are placed in the appendix (Fig. 33, 34).

3.2.3 The species-poor grassland dominated by *Calamagrostis villosa*

As described above (chapter 1.4.3 Species-poor grasslands dominated by *Calamagrostis villosa*), the studied species-poor community is probably closest to the association *Sileno vulgaris-Calamagrostietum villosae* Jeník et al. 1980, which is not considered as a separated vegetation type anymore. Nowadys (Chytrý et al. 2007), it became part of the species-richer association *Crepido conyzifoliae-Calamagrostietum villosae*. Although the sudied stand is species-poor, it belongs probably to this association, which is included in the alliance *Calamagrostion villosae* - subalpine tall-stalk grasslands. The stand dominated by *C. villosa* is homogenous and it forms only little enclave c. 300 m^2 along the northern edge of the spruce forest. Nevertheless, *C. villosa* does not occur in that forest. The snow lies for a longer time on the locality compared with other places in the surrounding. Northward, the stand verges into the mosaic of communities with the majority of the alliance *Calthion*. Orchids and disseminated willows are also present. GPS localization of the permanent monitoring plot is 48°38'7.018"N, 14°6'6.461"E and the altitude is 935 m a.s.l. The exposition is northeast and the slope gradient is 10-20 degrees. The pictures of the community are placed in the appendix (Fig. 35, 36).

3.2.4 Wet *Filipendula* meadow

The last of the studied meadow types is the representative stand of the association *Lysimachio vulgaris-Filipenduletum ulmariae* Balátová-Tuláčková 1978 (wet *Filipendula* grasslands with *Lysimachia vulgaris*), which is included in the alliance *Calthion*. Although *Filipendula ulmaria* and *Lysimachia vulgaris* are main dominants with considerable proportion of the biomass, the stand is species-rich (see table 4). The permanent monitoring plot was established in the stand with the easter exposition and the slope gradient less than 10 degrees. The site is lowermost and most distant from the others at the same time. Whereas three remaining sites lie at an altitude of 930-940 m a.s.l. and the distans between them is less than 220 m, the plot in wet *Filipendula* meadow is located 1 440 m far from them in a straight line southeast along a stream of the brook Horský potok at an altitude of 830 m a.s.l. The community *Lysimachio-Filipenduletum* forms several homogenous stands here, relatively closely differentiated from other habitats. This area (c. 2 hectares) is waterlogged, especially in spring. Species such as *Carex* spp. and *Scirpus sylvaticus* are abundant, as well as dispersed stands of willows and birches. In addition, there are several endangered species of category C3 (Vulnerable) abundantly near the monitoring plot, such as *Drosera rotundifolia*, *Menyanthes trifoliata* and *Oxycoccus palustris*. GPS localization of the permanent monitoring plot is 48°37'26.326"N, 14°6'42.14"E. The brook is c. 10 m far from the site. The pictures of the community are placed in the appendix (Fig. 37, 38).

These meadow types are hereafter referred to as „*Molinion*“, „*Carex* community“, „*Calamagrostis* community“ and „*Filipendula* community“. The abbreviations of the species names showed in CCA ordination diagrams represent first four letters of the genus and species, e.g. *EquiFluv* represents the species *Equisetum fluviatile* (see the list in the table 3).

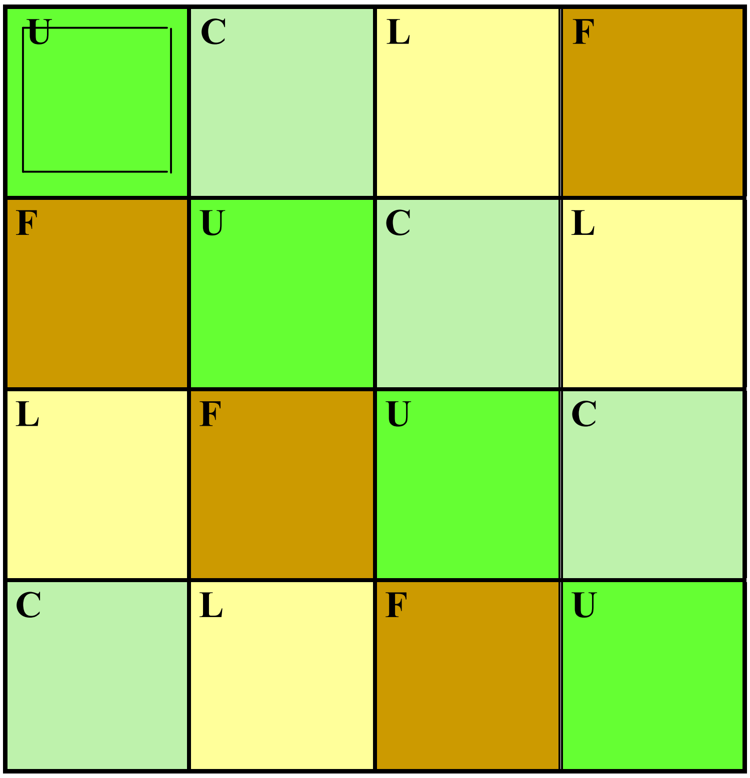
3.3 Experimental design and treatments

Four permanent monitoring plots (PMP) were established in May 2006 in latin square design. Each was situated in one of the studied phytocoenoses described above. The altitude and GPS localizations are presented in the table 1.

Table 1. The summary of established PMPs - the altitude and GPS localization.

No.	Community	Altitude (m a.s.l.)	GPS
1	*Molinion*	940	Loc: 48°38'14.459"N, 14°6'7.526"E
2	*Carex* com.	930	Loc: 48°38'9.049"N, 14°6'9.357"E
3	*Calamagrostis* com.	935	Loc: 48°38'7.018"N, 14°6'6.461"E
4	*Filipendula* com.	830	Loc: 48°37'26.326"N, 14°6'42.14"E

Each latin square was established in homogenous stand of a given community. The homogenity of the stand is an essential requirement for testing the impacts of different management treatments on vegetation characteristics. The dimensions of each PMP were 520 × 520 cm. Edges were fixed by nailed wooden nogs. The squares were divided into 16 plots 130 × 130 cm. To avoid the edge effect, the real area monitored was always 1 m^2 (see the figure 2 top left). The treatments consisted of (1) cutting once a year, (2) liming (250 g of ground limestone . m^{-2}, (3) fertilizing (1 kg of homogenized cattle slurry of local origin with 1 kg of water . m^{-2}) and (4) untreated control. The „management“ hereafter could be referred to as „treatment“. The position of the treatments within the latin square is shown in the following picture (Fig. 2).



U	**C**	**L**	**F**
F	**U**	**C**	**L**
L	**F**	**U**	**C**
C	**L**	**F**	**U**

Fig. 2. The areal organization of the treatments in the latin square. U - untreated control, C - cutting, L - liming, F - fertilizing. The real area monitored (1 m^2) is shown in the upper left plot.

In the middle of each 1 m^2 plot one small square 30 × 30 cm was established (Fig. 3). These 30 × 30 cm and 1 × 1 m squares are hereafter referred to as „small squares“ and „large squares“. In the large squares the changes of both total cover of plants and mosses and individual species cover were monitored. These variables were also investigated in the small squares, and in addition the total dry matter and the dry matter of individual species were monitored here. The dry matter is hereafter referred to as „biomass“.

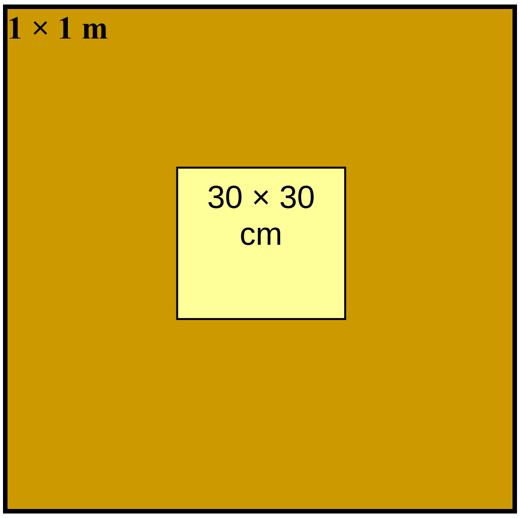


Fig. 3. The establishment of the small squares 30 × 30 cm in the middle of the large squares 1 × 1 m for the monitoring of changes in total above-ground biomass and the above-ground biomass of individual species (dry matter), as well as the total plant cover and individual species cover.

3.4 Field and laboratory works

The stands appropriate for the establishment of PMPs were chosen at the beginning of the vegetation season in 2006. All PMPs were established in May and 1 m^2 squares were limed and fertilized at the same time, e.g. 4 large squares were limed and 4 large squares were fertlized within each latin square. The ground limestone in the amount of 250 g was scattered carefully and equally from on high c. 30 cm on given 1 m^2 plot so that the neighbouring plots would not be hit by an application, see the illustrative photo in the appendix (Fig. 40). The fertlizer was applied using the watering can in the amount of 1 kg of homogenized cattle slurry with 1 kg of water, see the illustrative photo in the appendix (Fig. 39). This process was repeated annually at the beginning of the vegetation season, three times in total - 2006, 2007 and 2008.

The fourth latin square (*Filipendula* community) must have been abandonned immediately in the first vegetation season due to an inappropriate location - near the water spring in the slope, which was not evident during the plot establishment. The site was later inundated, so these conditions exclude the next continuation of the experiment. PMP was reestablished in more suitable place the same year in summer 2006. Thus, data from the first season from this meadow type are missing.

Data were recorded in the period of maximal production; i.e. in the middle of July (10. 7. 2006, 11. 7. 2007 a 6. 7. 2008). In all large squares 1 m^2 (64 in total in one latin square), the following variables were recorded: the total cover of E0 (mosses) and E1 (plants) in %, the complete list of vascular plants present in a given plot and the estimation of individual species cover in %. The cover was recorded in % for Microsoft Excel application use and for the subsequent statistical calculations, thus, no classical Braun-Blanquet degrees (1 to 5) were used. Only for the cover under 1% the Braun-Blanquet's combination scale of abundance and dominance was used: the value 0,02% for 1 or 2 individuals with minute cover (the degree „r“) was used and the value 0,1% for the cover less than 1% (the degree „+“) was used. Except of these two values, the others were rounded off the whole %.

The same procedure of phytocoenological survey was performed in the case of the small squares, 64 in total in one latin square. After the inventory, the estimation of the total cover E0 and E1 and cover of all individual species, the vegetation samples in the small squares were cut to ground level and the biomass was transported in plastic bags to the laboratory. In the case of the untreated control which was not cut, relevés of the small squares were not recorded in the middle of the large squares as in the case of other treatments, but outside of the latin square in homogenous stand, where the frame of 30 × 30 cm was randomly hit in. After the recording of four relevés 30 × 30 cm of untreated control beside the latin square, the biomass was removed in the same way as in the case of other treatments.

After all operations described above, mowing was carried out using a brushcutter in all plots except the control squares. Thus, in the case of fertilizing and liming, these treatments represent the mixture of two treatments together: in the followed text including the results, „fertilizing“ means always „fertilizing + cutting“ and „liming“ means always „liming + cutting“. Through these methods, both liming and fertilizing in conjunction with cutting, I aimed to simulate as far as possible real farming practices.

In the laboratory, the above-ground biomass samples of all small squares were sorted into species, oven-dried (85˚C, 17 hours) and weighed with an accuracy of one decimal of gramme, in the case of species lighter than 1 g with an accuracy of three decimals.

3.5 Soil sampling

The soil samples for the assesment of the content of extractable soil components were sampled only once, in May 2006 (the first season) using the sampling spear with the diameter of 3 cm. The samples were removed from the depth of 10 cm, sixteen samples within the latin square, i.e. one sample from each large square. The samples were subsequently mixed within treatmetns, hence four mixed samples arose from each PMP. These samples were sent to specialized company for an assesment of N-tot, P, K, C-ox, Mg, Ca, dry matter (%) a pH ($CaCl_2$).

3.6 Statistical analyses

STATISTICA 8.0 (StatSoft, Inc. 2007) and CANOCO for Windows (ter Braak & Šmilauer 1998) were used for statistical analyses. CanoDraw program (ter Braak & Šmilauer 2002) was used to visualize the results of the CANOCO analyses.

All statistical analyses could not be done as a whole (from all data set) because the data of *Filipendula* meadow from the first season are missing. Therefore two independent partial analyses must have been performed (see table 2):

1. The first set of results follows the data of only three meadow types (*Molinion*, *Carex* community, *Calamagrostis* community) and of all three seasons, i.e. all *Filipendula* community data were not considered.
2. The second set of results follows the data of all four meadow types and only two years, i.e. all data from the first season were not considered.

Table 2. Two different sets of results: (1) grey area represents set of data for statistical analyses where *Filipendula* community data were not included (i.e. only three meadow types within all three years are considered) and (2) black-boxed area represents set of data for statistical analyses where the data of the first season were not included (i.e. all meadow types within only second and the third year are considered). Both STATISTICA and CANOCO were used for both sets of data computing.

year 1	*Molinion*	*Carex* com.	*Calamagrostis* c.	NO DATA
year 2	*Molinion*	*Carex* com.	*Calamagrostis* c.	*Filipendula* com.
year 3	*Molinion*	*Carex* com.	*Calamagrostis* c.	*Filipendula* com.

STATISTICA

ANOVA (Analysis of Variance) for latin squares was used for testing the differences between meadow types, treatments and years in these 6 variables: (1) the amount of above-ground biomass (dry matter) in small squares, (2) the cover in the small squares, (3) the cover in the large squares (both covers separately), (4-6) the Shannon-Wiener diversity index derived from the data of (4) the biomass in the small squares, (5) the cover in the small squares and (6) the cover in the large squares. Data were ln-transformed.

Dependent variables were biomass amount, cover in the small and large squares, Shannon diversity index derived from biomass and cover in both squares. Independent variables were: treatments (cutting, liming, fertilizing, untreated control), meadow type, year, column and row of latin square. The row and the column were used as a random factor. Interactions were calculated for meadow type*year, meadow type*treatment, year*treatment. The untreated control in the first year was used for the baseline data.

Shannon diversity index H' was derived from the botanical composition data (biomass, plant cover in the large squares and plant cover in the small squares) in each square using CANOCO to quantify the distribution of individuals among species.

$$H' = -\sum_{i=1}^{S} p_i \ln p_i$$

n_i The number of individuals in species i; the abundance of species i.

S The species richness (number of species).

N The total number of all individuals.

p_i The relative abundance of each species, calculated as the proportion of individuals of a given species to the total number of individuals in the community:

$$\frac{n_i}{N}$$

p_i is the proportion of species i (abundance, cover, biomass or other meaningful value could be used) (Moravec et al. 1994). For any given number of species, the maximum possible H' ($H_{\max} = \ln S$) occurs when all species are present in equal numbers.

CANOCO

Total community response (i.e. relation of plant species) to treatments and period was analysed both by unconstrained and constrained ordinations in CANOCO package (ter Braak & Šmilauer 1998). Data were standardized and centralized. The length of the gradient in DCA analysis had higher value than 4. Therefore both the unimodal indirect (unconstrained) CA analysis and unimodal direct (constrained) CCA analysis were used. CA (Correspondence Analysis) for visualization the relations of plant species regardless of the treatments was used. CCA (Canonical Correspondence Analysis) for visualization the interactions of treatments, years and species was used. In this case, the impact of predictors on responses was presented using several gradients (canonical axes). Data were log-transformed (transformed into a logarithmic form) for both CA and CCA analyses. Explanatory variables were both categorial (cutting, liming, fertilizing, control) and quantitative (year).

The significance of axes and variables was calculated using the Monte Carlo permutation test under 499 permutations. It means that „p“ could have been lower than 0,002. The significance of the first canonical axis was considered as a test criterion for the Monte Carlo permutation test. The stepwise selection was used for the calculation of the significance of variables. Forward selection (fs) - the constriction of the number of explanatory variables was also used. Explanatory variables were not very correlated with others remained in the model. Furthermore, the restriction of 8% of the species was used (species conforming to the model at 8% and more remained in the ordination diagram).

4. Results

Table 3 shows all vascular plant species found within meadow types and the tracking period independently of the management, abundance and time of occurrence. Table 4 shows the number of species of the four meadow types within all years and the comparison with the usual range of species richness presented in literature (Chytrý et al. 2007). Table 6 shows values of the content of extractable soil components, pH ($CaCl_2$) and % of the dry matter of *Molinion*, *Carex* community and *Calamagrostis* community within management treatments.

Table 3. The list of all species found within the four sampling areas: *Molinion*, *Carex* community, *Calamagrostis* community, *Filipendula* community. The presence of taxa found (symbolized by „x“) within the meadow type is independent both on management treatments and years. The presence of given taxa in the table does not advert to its frequency and/or abundance in latin square.

Species	**Abbreviation**	***Molinion***	***Carex***	***Calamagrostis***	***Filipendula***
Agrostis capillaris	*AgroCapi*	x	x		
Agrostis stolonifera	*AgroCapi*	x			x
Achillea millefolium agg.	*AchiMill*	x			
Achillea ptarmica	*AchiPtar*				x
Anemone nemorosa	*AnemNemo*	x	x		x
Angelica sylvestris	*AngeSylv*	x			x
Bistorta major	*BistMajo*	x	x		x
Calamagrostis villosa	*CalaVill*			x	

Species	**Abbreviation**	***Molinion***	***Carex***	***Calamagrostis***	***Filipendula***
Caltha palustris	*CaltPalu*	x	x	x	x
Carex brizoides	*CareBriz*	x	x	x	
Carex nigra	*CareNigr*	x			x
Carex pallescens	*CarePall*	x			
Carex panicea	*CarePani*	x			x
Carex rostrata	*CareRost*				x
Cirsium heterophyllum	*CirsHete*	x			
Cirsium palustre	*CirsPalu*	x	x	x	x
Deschampsia cespitosa	*DeschCes*	x	x		
Dryopteris dilatata	*DryoDila*			x	
Epilobium palustre	*EpilPalu*	x	x		x
Equisetum fluviatile	*EquiFluv*				x
Equisetum sylvaticum	*EquiSylv*	x	x		
Eriophorum angustifolium	*ErioAngu*				x
Festuca rubra	*FestRubr*	x			
Filipendula ulmaria	*FiliUlma*				x
Galeopsis pubescens	*GalePube*	x	x	x	
Galium album	*GaliAlbu*	x			
Galium palustre	*GaliPalu*	x	x		x
Galium uliginosum	*GaliUlig*	x			x
Heracleum sphondylium	*HeraSpho*	x	x		
Holcus lanatus	*HolcLana*	x			
Holcus mollis	*HolcMoll*	x	x		x
Juncus effusus	*JuncEffu*	x			x
Juncus filiformis	*JuncFili*				x
Lysimachia vulgaris	*LysiVulg*				x
Maianthemum bifolium	*MaiaBifo*	x			
Molinia caerulea	*MoliCaer*	x	x		x
Myosotis nemorosa	*MyosNemo*	x	x	x	x
Picea abies juv.	*PiceAbie*			x	
Potentilla erecta	*PoteErec*	x			x
Potentilla palustris	*PotePalu*				x
Ranunculus acris	*RanuAcri*		x		x
Ranunculus auricomus	*RanuAuri*	x			x
Sanguisorba officinalis	*SangOffi*	x			x
Scirpus sylvaticus	*ScirSylv*		x		x
Senecio ovatus	*SeneOvat*	x	x	x	
Sorbus aucuparia juv.	*SorbAucu*			x	
Trientalis europaea	*TrieEuro*	x			
Urtica dioica	*UrtiDioi*		x		
∑ 48		33	19	10	28

Table 4. The number of species found within years and meadow types. The comparison of total number of species and usual range of the species number within meadow type following Chytrý et al. (2007).

Meadow type	**Number of species within 3 years**			**∑**	**Usual range**
	year 1	**year 2**	**year 3**		
Molinion	23	28	29	33	35 - 50
Carex com.	13	14	11	19	20 - 35
Calamagrostis com.	4	7	9	10	10 - 25
Filipendula com.	no data	26	25	28	15 - 20

The number of species found within meadow types and treatmetns changed during the tracking period, as the table 5 shows. In *Molinion*, the species richness ranged between 14 (limed plots in the first season) and 26 (the control in the second year). Although untreated plots had the highest species richness all the time, the number of plant species increased fast in the case of other treatments. The biggest increase during three years occurred in cut squares (7 species). The percentage share of grass species from the total species richness (not the cover or biomass) was more or less constant only in the control during the trackig period. Proportional share of grasses slightly decreased in cut and limed plots, whereas it increased in fertilized plots.

Over time the species *Holcus mollis* spread considerably in the *Molinion* at the expense of the species *Molinia caerulea*. Parallel rapid retreat of *Molinia* was observed in all treatments. For example, in the first season, *Holcus* occurred only in one untreated plot (of four plots) with 5% cover, whereas in the second year it was observed in all four squares with an average cover of 12.5%, and reached as high as 16% in the last season. Simultaneously the cover of *Molinia* decreased from initial 66% to 49%. This trend was the most evident in cut squares, the average cover of *Holcus* increasing from 16% to 53% during tracking period, whereas *Molinia* decreased from 65% to 19%. Neither limed nor fertilized plots showed such extreme results. They differed only in having a relatively higher cover of *Molinia* and lower cover of *Holcus* in fertilized plots compared to limed squares.

In species-poor *Carex brizoides* community, the species number ranged between 4 (cut and limed squares in the last season) and 12 (the control in the second season). The number of species decreased in all treatments except of fertilized plots, where it slightly increased.

Calamagrostis villosa stand is the most species-poor community of all meadow types. The number of species ranged between 2 and 7. None the less, the species richness increased in all treatments, mostly in limed squares.

The data of the first year of *Filipendula* community are missing. Other two seasons were well-balanced in the number of species in all treatments (22 species maximum, 17 species minimum) as well as the percentage share of grass species. The species richness decreased in untreated and limed squares, slightly increased in cut squares and it remained unchaged in the case of fertilized plots.

Table 5. The number of species found within years, meadow types and management treatments. Each number is the sum of all taxa found in four large squares of one treatment in one meadow type in one year. **M** - *Molinion*, **Cb** - *Carex* community, **Cv** - *Calamagrostis* comminty, **F** - *Filipendula* community, **no** - no management, **cut** - cutting, **lim** - liming, **fert** - fertilizing, ∑ **spec** - the sum of all taxa. Species are divided into grasses and forbs. Grasses also include genera *Carex*, *Eriophorum*, *Juncus*, *Scirpus*. Forbs also include genera *Dryopteris*, *Equisetum* and seedlings of trees *Picea* and *Sorbus*.

	year 1			**year 2**			**year 3**		
Meadow	**grasses**	**forbs**	**∑ spec**	**grasses**	**forbs**	**∑ spec**	**grasses**	**forbs**	**∑ spec**
M-no	5	15	20	8	18	26	6	17	23
M-cut	4	11	15	6	15	21	7	15	22
M-lim	3	11	14	5	14	19	4	15	19
M-fert	3	12	15	4	15	19	6	14	20
Cb-no	3	8	11	3	9	12	3	3	6
Cb-cut	3	4	7	2	4	6	2	2	4
Cb-lim	2	4	6	2	5	7	2	2	4
Cb-fert	3	3	6	3	4	7	2	6	8
Cv-no	2	0	2	2	0	2	2	1	3
Cv-cut	2	0	2	2	0	2	2	2	4
Cv-lim	2	0	2	2	5	7	2	5	7
Cv-fert	2	2	4	2	5	7	2	4	6
F-no	-	-	-	8	14	22	5	12	17
F-cut	-	-	-	6	13	19	7	15	22
F-lim	-	-	-	7	13	20	6	12	18
F-fert	-	-	-	6	14	20	7	13	20

Table 6. The content of extractable soils components and other variables. The results of analyses are in 100% dry matter. **Molin** - *Molinion*, **Carex** - *Carex* community, **Calam** - *Calamagrostis* community, **no mng** - untreated control, **cut** - cutting, **lim** - liming, **fert** - fertlizing.

Sample	L.dry matter [%]	pH($CaCl_2$) []	P [mg/kg]	K [mg/kg]	Mg [mg/kg]	Ca [mg/kg]	Dry matter [%]	C-ox [%]	N-tot [%]
Molin - no mng	91.33	4.07	< 10	262	200	1622	22.89	51.1	1.49
Molin - cut	90.37	4.00	< 10	239	253	1821	25.29	58.4	1.52
Molin - lim	91.63	4.37	< 10	215	474	2208	25.73	48.5	1.32
Molin - fert	90.82	4.08	< 10	207	291	2131	27.72	55.3	1.34
Carex - no mng	89.55	3.78	< 10	236	229	1494	32.27	69.1	1.76
Carex - cut	89.43	3.79	< 10	247	221	1454	31.40	66.3	1.69
Carex - lim	89.40	3.98	< 10	181	358	1555	29.46	66.6	1.63
Carex - fert	90.48	3.81	< 10	176	189	1326	30.49	59.4	1.56
Calam - no mng	94.65	3.51	< 10	102	31	450	59.86	23.4	0.68
Calam - cut	94.39	3.40	< 10	129	36	< 150	49.91	24.7	0.72
Calam - lim	95.23	3.54	< 10	97	102	181	56.23	22.2	0.74
Calam - fert	95.42	3.57	< 10	107	31	< 150	63.99	21.2	0.65

4.1 The statistical evaluation of three meadow types in three seasons

4.1.1 The comparison of meadows

All three plant communities studied (*Molinion*, *Carex* community, *Calamagrostis* community) differ significantly in all measured dependent variables: the amount of biomass (F(2)=44.409, p=0.006), the cover in the small squares (F(2)=53.847, p<0.001), the cover in the large squares (F(2)=78.771, p<0.001) and in three diversity values: Shannon index *H'* derived from biomass (F(2)=23.596, p=0.001), cover in the small squares (F(2)=23.593, p=0.001) and cover in the large squares (F(2)=123.924, p<0.001).

Figure 4 shows the differences between three meadow types in the cover in the large squares independently of the year and treatment. Graphs displaying the differences in the biomass and cover in the small squares are not shown, because they are almost identical with figure 4. The biomass and the cover of *Carex* community was higher considerably than in the case of *Molinion* and *Calamagrostis* community in all three graphs. The cover in the small squares differed from the cover in the large squares in slightly lower values in all meadows and also in wider range of values in *Molinion*. In graph with the biomass, the amout of the biomass in *Molinion* is higher only slightly compared to *Calamagrostis* community, whereas it is higher considerably in the case of both covers.

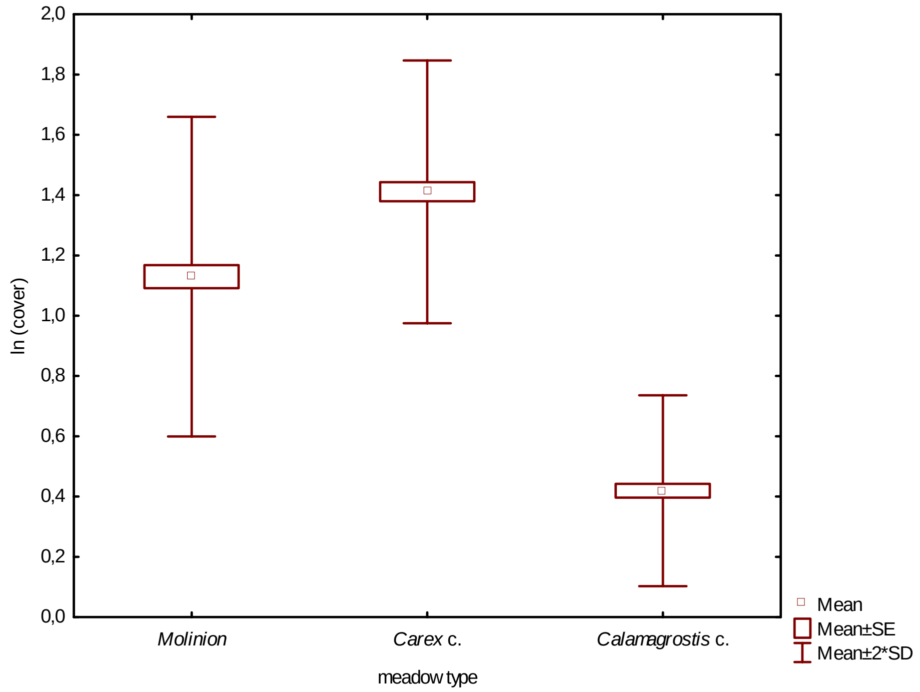


Fig. 4. The comparison of the cover in the large squares (ln transformed) of three meadow types independently of the year and treatment. The readings for the cover differ significantly (p<0.001): *Molinion* 87%, *Carex* community 96%, *Camagrostis* community 40%.

The comparison of the cover and biomass amount during the tracking period independently of the treatment and meadow type, i.e. „the impact of the year“ both on cover and biomass, provided the following results. The cover in the small squares during the monitored period differed significantly: F(2)=21.905, p=0.006 as well as the cover in the large squares: F(2)=11.767, p=0.025. There was virtually no difference between both graphs, thus only on is shown: the comparison of the cover in the small squares (Fig 5). In the first season, the cover in the small squares was higher (75%) than in both following years (65%). The biomass amount among three years differed only on an indicative level of significance (F(2)=58.096, p=0.057). The trend was opposite in this case: the first two seasons were identical both in the mean and in the range, whereas the biomass increased in the last year and the range of values increased also. Graph is not shown.

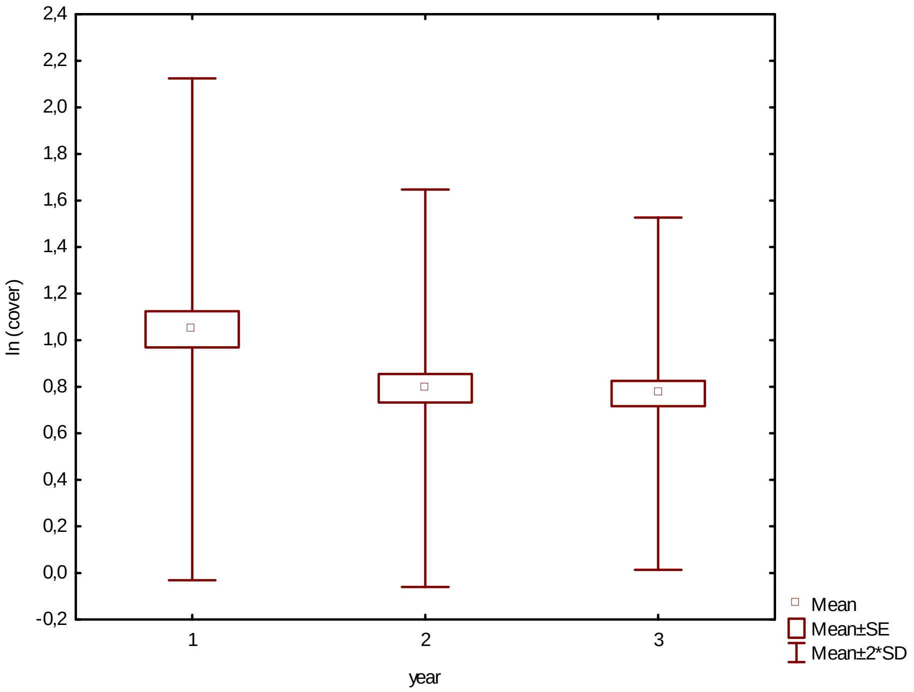


Fig. 5. The comparison of the cover in the small squares (ln transformed) during the tracking period independently of the treatment and meadow type. The readings for the cover differ significantly ($p<0.006$): 75% (year 1), 65% (year 2 and 3).

The interaction meadow type*year independently of the treatments had significant impact both on biomass: $F(4)=8.805$, $p<0.001$, the cover in the small squares: $F(4)=6.172$, $p<0.001$ and the cover in the large squares: $F(4)=3.865$, $p=0.006$ as well as Shannon index which will be described later. These results mean, that all meadow types differ significantly with each other within all years in the biomass and both covers. Figure 6 shows the significant changes in biomass of the meadows during the monitored period. Whereas *Calamagrostis* community with the lowest amount of biomass had quite stable values, in *Molinion* the increase in biomass in the third season was recorded. This trend was most evident in *Carex* community, which increased the biomass considerably in the last season following the slight decrease in the second year. The cover in the small squares of *Carex* community and *Molinion* had slightly downward trend (graph not shown): the cover of *Carex* community decreased from initial 100% to 91% (the last year) and the cover in *Molinion* decreased from 85% to 68%. The cover of *Calamagrostis* community was quite stable: from 40% to 36%. The covers in the large squares (Fig. 7) slightly differ from those in the small squares. The cover of *Molinion* was slightly higher and also had downward trend during the three years (from 93% to 83%), which was soft. *Carex* community decreased the cover in the second year to 94% from initial 100% and then increased it slightly to 96% at the end of the monitored period.

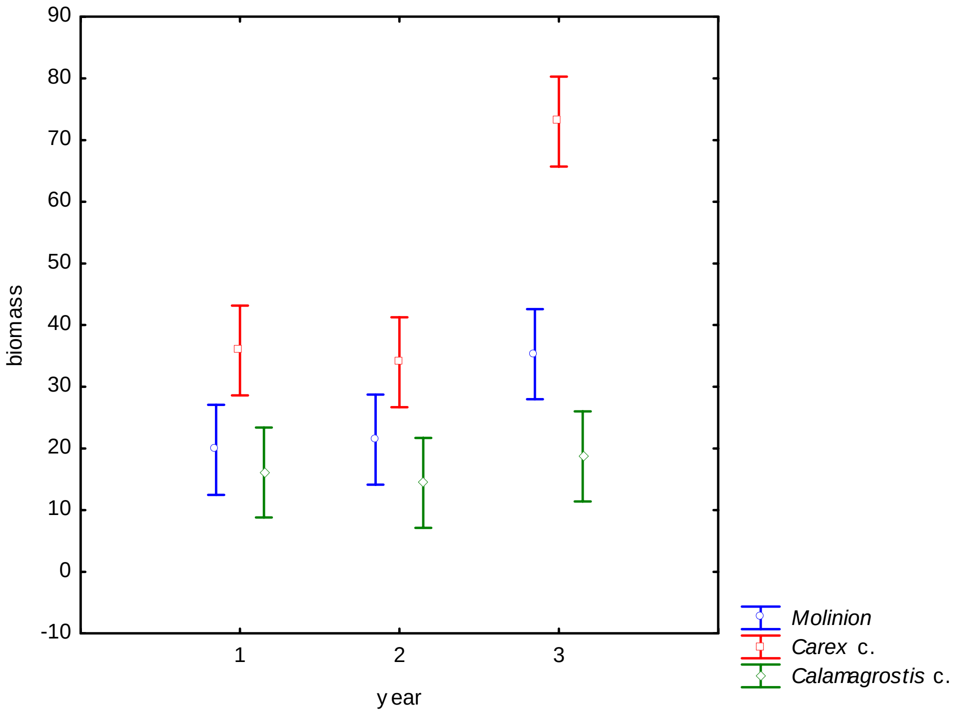


Fig. 6. The influence of interaction meadow type*year on the biomass in the small squares independently of management. The readings for the biomass of plant communities significantly differ ($p<0.001$) during the tracking period. The average weight of the dry matter: *Molinion*: 20g - 21g - 35g. *Carex* community: 36g - 34g - 73g. *Calamagrostis* community: 16g - 14g - 19g.

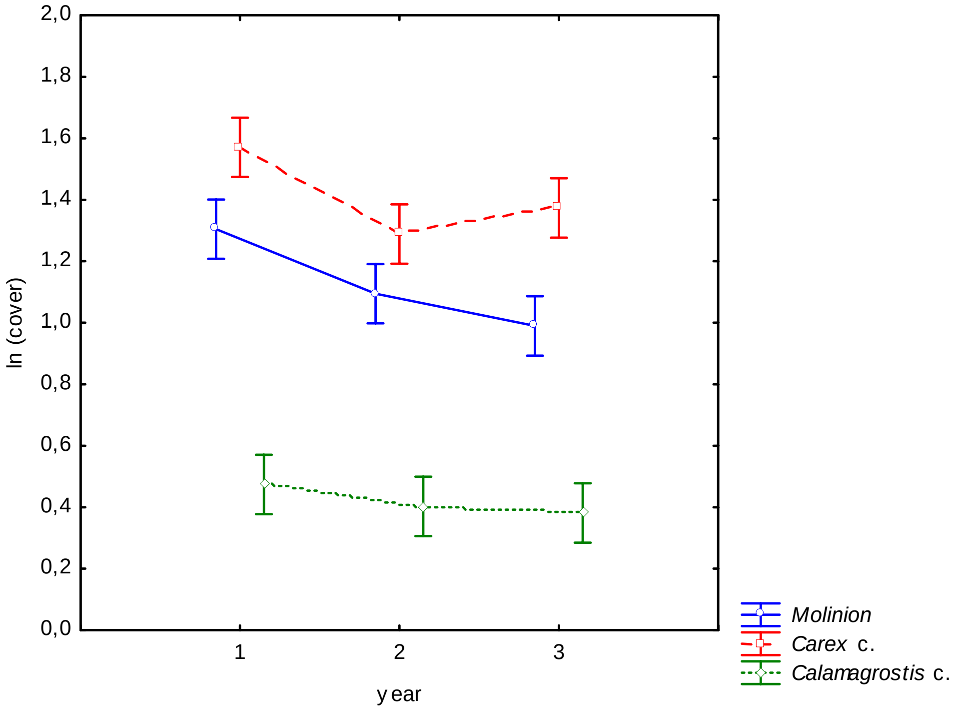


Fig. 7. The influence of interaction meadow type*year on the cover (ln transformed) in the large squares independently of management: The readings for the cover of plant communities significantly differ ($p=0.006$) during the tracking period. The average cover in the large squares during three seasons: *Molinion*: 93% - 87% - 83%. *Carex* community: 100% - 94% - 96%. *Calamagrostis* community: 45% - 39% - 37%.

All the three meadow types studied (*Molinion*, *Carex* community, *Calamagrostis* community) significantly differed in Shannon-Wiener diversity index independently of both the year and management, derived from both biomass: F(2)=23.596, p<0.001, cover in the small squares: F(2)=23.593, p=0.001 and cover in the large squares: F(2)=123.924, p<0.001. There was virtually no difference between all three graphs, thus only on is shown: graph displaying the differences between meadows in *H´* index values, which were derived from biomass data (Fig. 8). The highest values of Shannon diversity were in *Molinion* (*H´*=1.29), lower values were in *Calamagrostis* community (*H´*=1.01) and the lowest *H*‘ was in *Carex* community (*H´*=0.89).

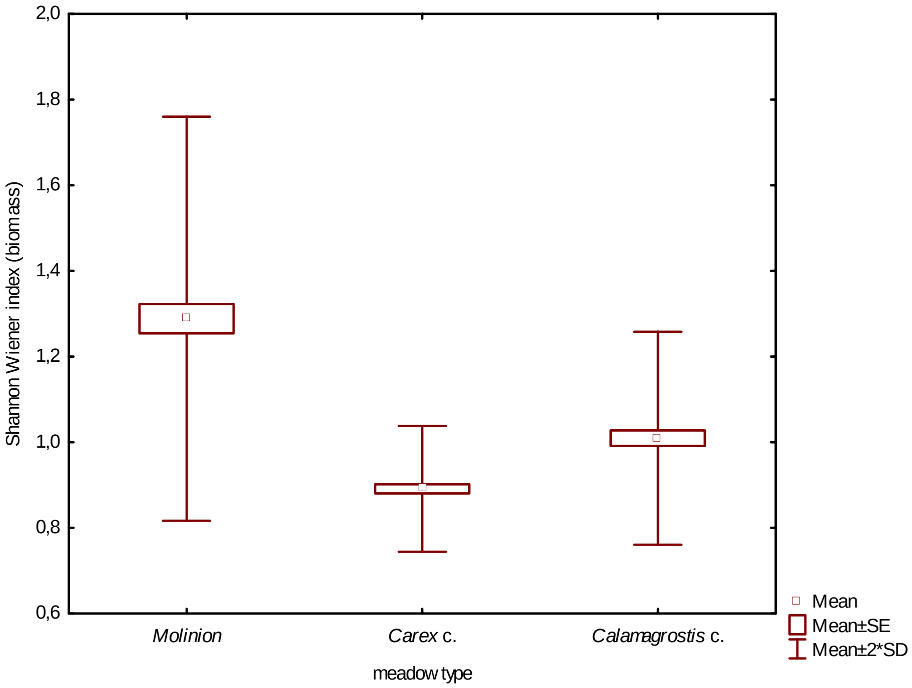


Fig. 8. The comparison of the Shannon-Wiener index (derived from the biomass) of all three meadow types independently of the year and treatment. Shannon index (*H´*) values differ significantly (p<0.001): *Molinion* 1.29, *Carex* community 0.89, *Calamagrostis* community 1.01.

The comparison of *H´* values in all years independently of the treatment and meadow type showed, that all years tested differed with one another significantly only in *H´* index derived from the cover in the small squares: F(2)=15.667, p=0.013 (Fig. 9). The Shannon index showed an upward trend during the three years (from 0.87 to 1.03) as well as the increasing range of values (maximum and minimum). The values of *H´* derived from the data of both biomass and cover in the large squares did not differ significantly during the tracking period.

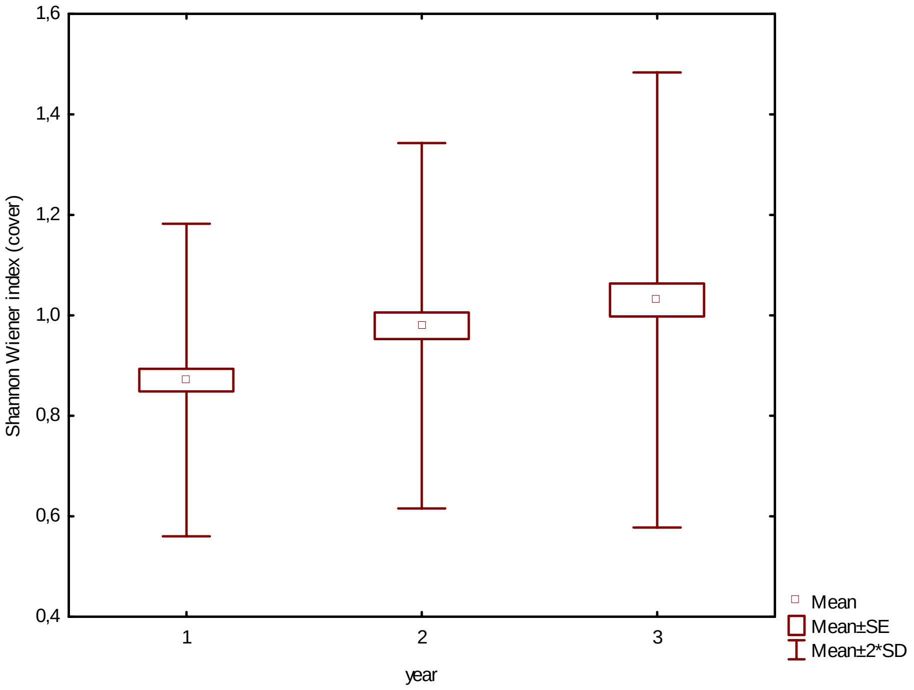


Fig. 9. The comparison of *H´* index (derived from the cover in the small squares) during the tracking period independently of the treatment and meadow type. Shannon index (*H´*) values differ significantly (p=0.013): 0.87 (year 1) - 0.98 (year 2) - 1.03 (year 3).

Also, the interaction meadow type*year independent of treatments was significant for *H´* index derived from both cover in the small squares: F(4)=3.195, p=0.017 (Fig. 10), and cover in large squares: F(4)=3.651, p=0.009 (Fig. 11).

Thus, plant communities differ from each other significantly during the tracking period in the *H´* index derived from both covers. The increase of *H‘* index occurred in both cases in all meadows. The lowest *H‘* index had the *Carex* community all the time and it increased only slightly: from initial 0.75 to 0.83 at the end of the experiment. The *Calamagrostis* community had higher values and a quicker increase compared to the *Carex* community: from 0.88 to 1.00. The quickest increase and the highest values were recorded in the *Molinion*: from 0.99 to 1.26. Shannon index values derived from the cover in the large squares (Fig. 11) had only slightly higher values in all meadows compared to those decribed above (Fig. 10).

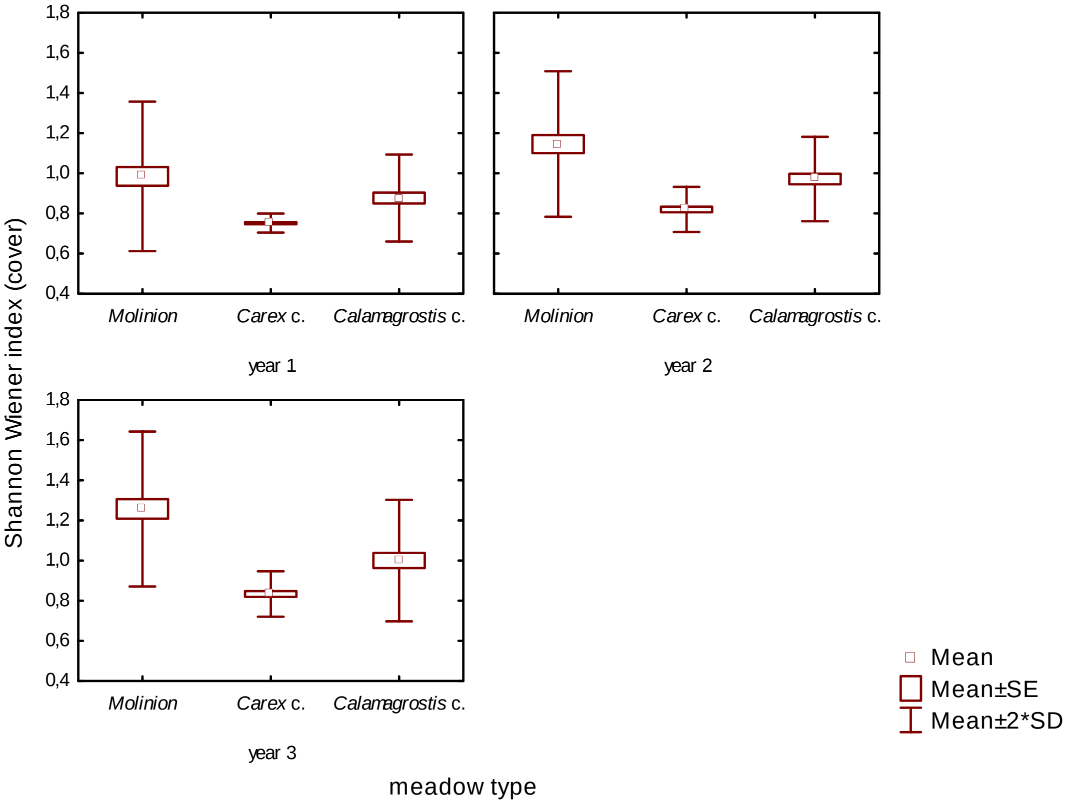


Fig. 10. The influence of interaction meadow type*year on the *H´* index (cover in the small squares) of plant communities independently of treatments. *H‘* index values of the three meadows differ significantly (p=0.017) during the three seasons. *Molinion*: 0.99 - 1.15 - 1.26. *Carex* community: 0.75 - 0.82 - 0.83. *Calamagrostis* community: 0.88 - 0.97 - 1.00.

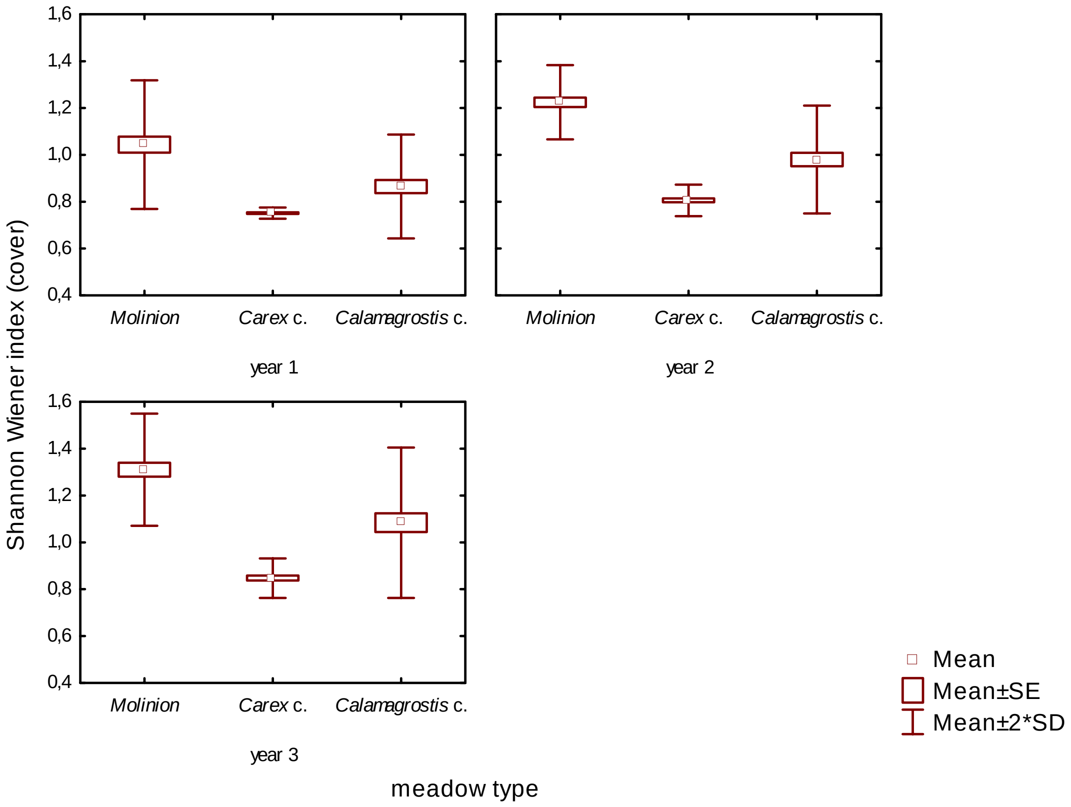


Fig. 11. The influence of interaction meadow type*year on the *H´* index (cover in the large squares) of plant communities independently of treatments: *H‘* index values of the three meadows differ significantly (p=0.009) during the three seasons. *Molinion*: 1.04 - 1.22 - 1.31. *Carex* community: 0.75 - 0.81 - 0.85. *Calamagrostis* community: 0.87 - 0.98 - 1.08.

The biomass and both covers were tested by Correspondence Analysis (CA). All three ordination diagrams were almost identical, therefore only one of them is used (the cover in the large squares, Fig. 12). This graph displays the relation of species of the three meadows independently of the management. The first canonical axis explained 90.3% of variability, second axis explained 73.4% of variability. Plant species are spatially very distinctive and they form three separated aggregates within the ordination. *Carex brizoides* prefers entirely different conditions compared to species present in *Molinion*, e.g. *Molinia caerulea*, *Holcus mollis*, *Carex nigra*, *Sanguisorba officinalis* and others. These species are clustered in the ordination and were also present in lesser extent in other meadows at the same time, mainly in the *C. brizoides* community. *Calamagrostis villosa* and seedlings of the spruce *Picea abies* stand outside the two remaining groups of species. *Bistorta major* does not share the stand with *Calamagrostis villosa*.

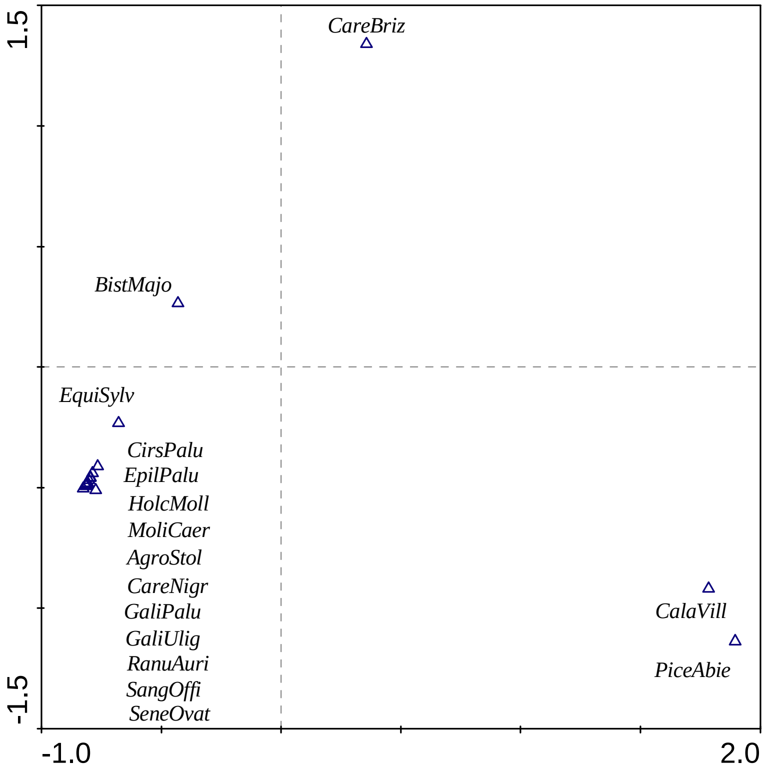


Fig. 12. CA ordination diagram displaying the relation of species (no restriction was used) of the three meadow types (cover in the large squares) independently of the treatments. 1st axis explains 90.3% of variability, 2nd axis explains 73.4% of variability. Abbreviations of species names represent first four letters of the genus and species (see tab. 3).

4.1.2 The impact of management treatments on meadow characteristics

The impact of the treatment independently both of year and meadow type was significant only on an indicative level of significance (F(3)=3.962, p=0.071) in the case of the cover in the small squares. Treatments differed with each other in their impact on the cover independently of the meadow type and year. Different responses in plant cover to cutting, liming and fertilizing are shown in graph (Fig. 13). The average cover within years and meadow types was 64%. Both liming and fertilising increased the cover (liming 74%, fertilising 71%) and cutting slightly decreased it (63%). The range of values was comparable in all treatments. No other significant changes in the cover, biomass and Shannon diversity following management treatments were not proved, except of the influence of interactions treatment*year and treatment*meadow type, which will by described later.

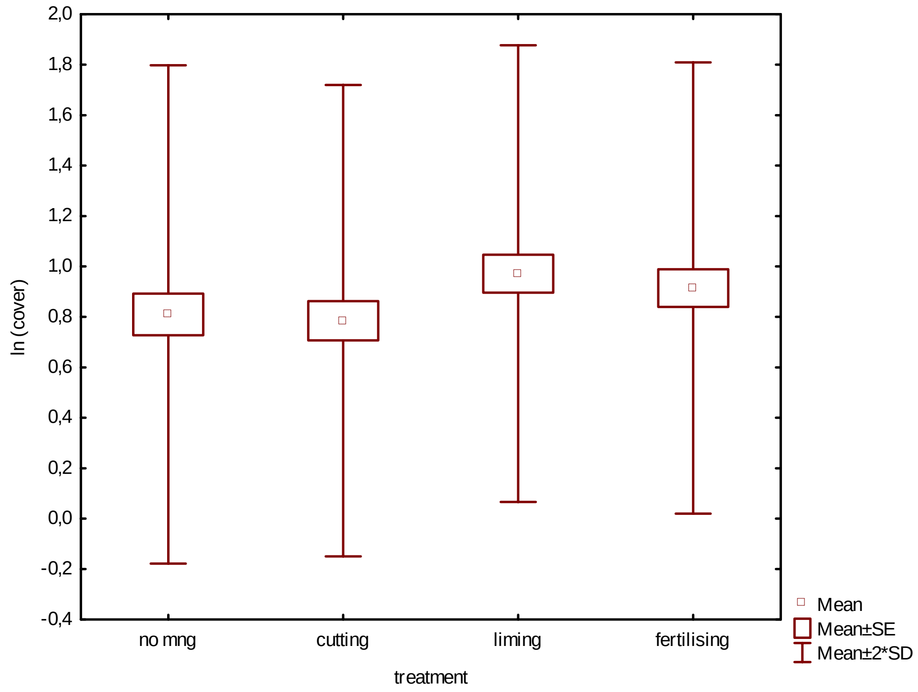


Fig. 13. The impact of the treatments (cutting, liming and fertilizing) on plant cover (ln transformed) in the small squares was almost significant (p=0,071). The changes in plant cover under three treatments differ independently both of meadow type and year. The readings for the cover: no mng 64%, cutting 63%, liming 74%, fertilizing 71%.

The interaction meadow type*treatment independently of year had significant influence on the cover in both the small squares: F(6)=3.071, p=0.009 (Fig. 14) and the large squares: F(6)=5.296, p<0.001 (Fig. 15). Thus, treatments differ significantly in their impact on the cover of meadow types. Slightly higher cover was recorded in the large squares compared to the small squares. The lowest cover in all treatments were in the *Calamagrostis* community, there were higher values in the *Molinion* and the highest cover was observed in the *Carex* community.

The average cover during the three seasons in the mown small squares (Fig. 14) of the *Calamagrostis* community was the same as in the control (30%), whereas the cover in the limed and fertilized small squares was higher (45% in limed plots, 42% in fertilized plots). The cover in the small squares of the *Molinion* was higher in the fertilized plots (78%) and particularly in the limed plots (83%) compared to the control (67%). The lowest cover was found in the cut squares (65%). The response of the *Carex* community cover to the treatments was quite different. The average cover during the three years was very similar in all treatments; 94% in the fertilized and cut plots, 95% in limed and untreated plots. However, the decrease of cover from 100% to 90-93% in all treatments occurred during the tracking period.

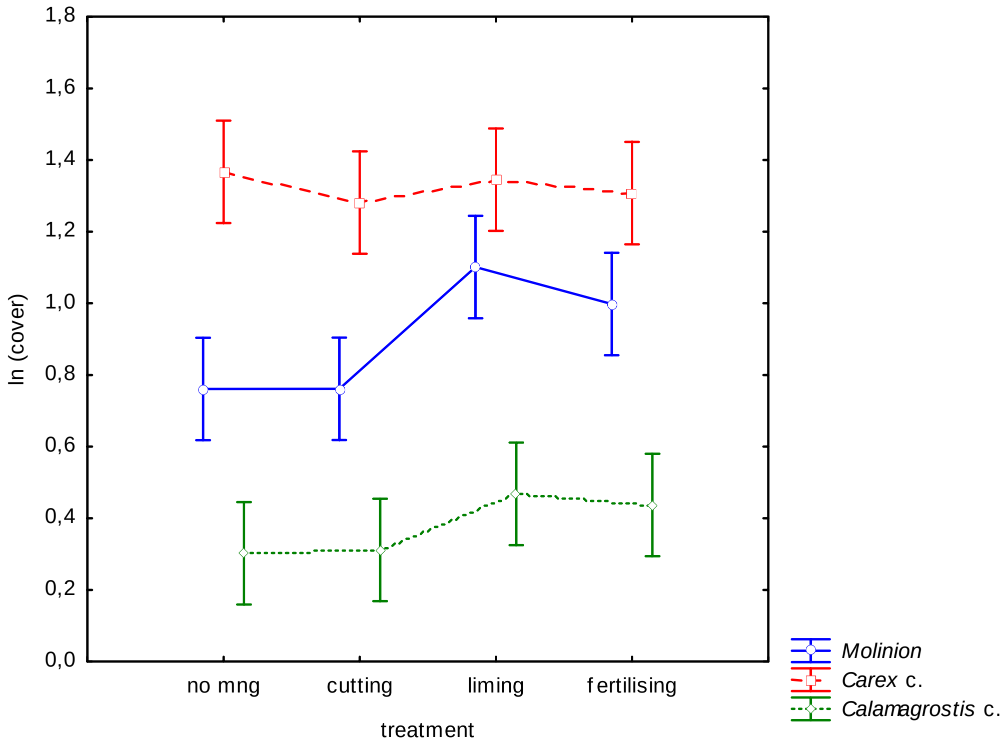


Fig. 14. The influence of interaction meadow type*treatment on the cover (ln transformed) in the small squares of plant communities independently of the year. Treatments differ significantly (p=0.009) in their impact on the cover in all meadows. The average cover in the small squares within the management treatments during the three seasons: *Molinion*: 67% - 65% - 83% - 78%. *Carex* community: 95% - 94% - 95% - 94%. *Calamagrostis* community: 30% - 30% - 45% - 42%.

The cover in the large squares responded to the management treatments slightly differently (Fig. 15). The *Calamagrostis* community had lower cover in the large cut squares (28%) compared to the control (38%). Limed (50%) and fertilized (46%) plots had higher cover of plants than the control, as was observed in the case of the small squares. The mown *Molinion* showed only a slightly decrease in cover (85%) as seen in the small squares. The average cover during the three seasons in the limed *Molinion* was the same as in the control (89%), but the cover decreased from 98% to 85% during this period, whereas the cover in untreated plots remained unchanged. The cover of the *Carex* community in the large squares differed from the cover in the small squares. Values were less consistent, with the highest cover being observed in the cut plots (98%), whereas the lowest cover was found in the mown small squares (94%).

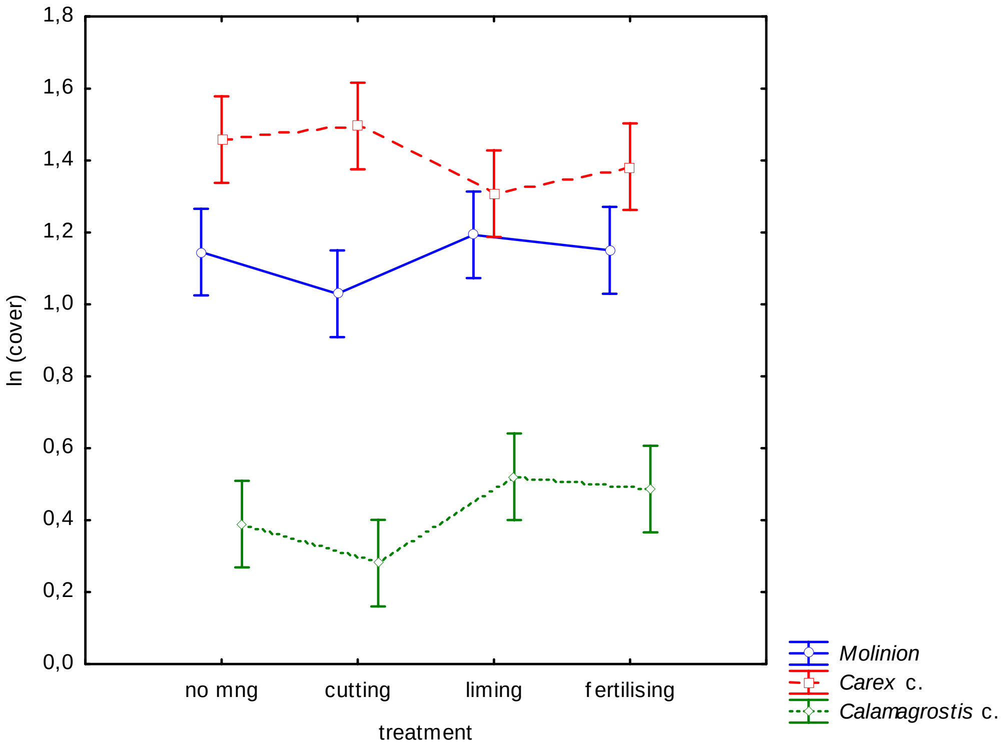


Fig. 15. The influence of interaction meadow type*treatment on the cover (ln transformed) in the large squares of plant communities independently of the year. Treatments differ significantly (p<0.001) in their impact on the cover in all meadows. The average cover in the large squares within the management treatments during the three seasons: *Molinion*: 89% - 85% - 89% - 87%. *Carex* community: 97% - 98% - 94% - 96%. *Calamagrostis* community: 38% - 28% - 50% - 46%.

The influence of the interaction meadow type*treatment on the biomass independly of the year was not significant (F(6)=1.305, p=0.264). Nevertheless, biomass increased in all treatments during the three years in *Carex* community. The decrease occurred only in cut and untreated plots in the second season. In *Calamagrostis* community, the biomass slightly increased in the control and cut plots, and slightly decreased in limed and fertilised plots during the monitored period. In *Molinion* the enhanced production of the biomass in all treatments was recorded except of cut plots, where the slight decrease in the second season occurred.

The interaction year*treatment independently of the meadow type had significant influence both on the biomass: F(6)=2.94, p=0.012 (Fig. 16) and the cover in the large squares: F(6)=3.666, p=0.003 (Fig. 17), i.e. treatments had different impact both on biomass and cover in the large squares independently of the meadow type during the three years.

The values of the biomass (Fig. 16) within all treatments were quite stable during the first two seasons and lower compared to the last year. In the last season, the biomass increased in all the treatments, the most in the control. Limed and fertilized plots had more or less linear increase of the biomass. The values of the cover in the large squares (Fig. 17) under all treatments were variable in the first season. It was due to higher cover in limed and fertilzed plots (84%) compared to cut (73%) and untreated squares (75%). In the second and the last year, the cover was more stable in all treatments.

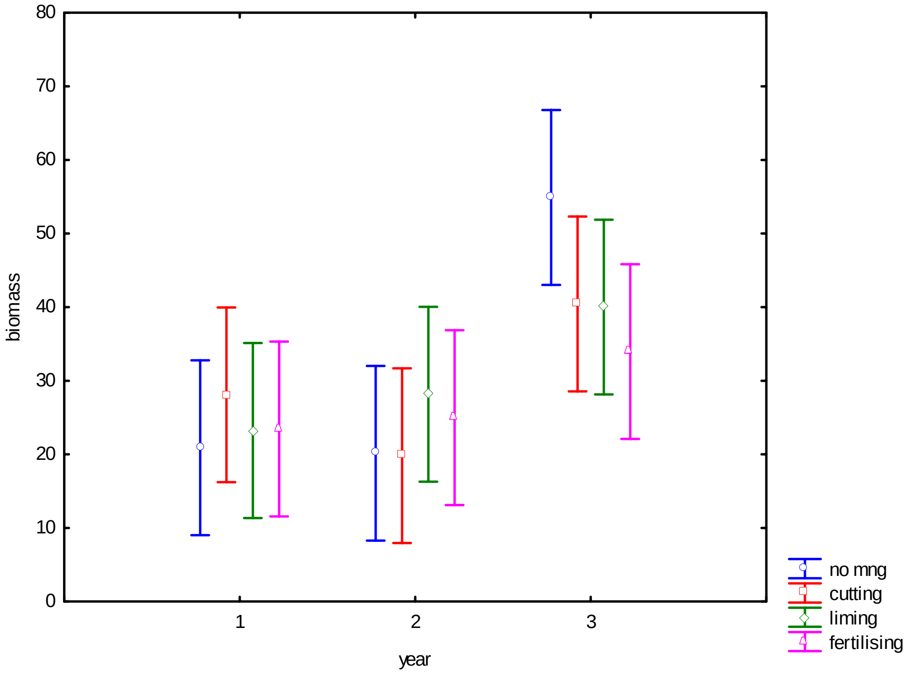


Fig. 16. The influence of interaction year*treatment on the biomass of plant communities independently of the meadow type. Treatments differ significantly (p=0.012) in their impact on the biomass during the three seasons. The average weight of the dry matter within the meadow types during the tracking period: no mng: 21g - 20g - 55g. cutting: 27g - 20g - 40g. liming: 23g - 28g - 40g. fertilizing: 23g - 25g - 34g.

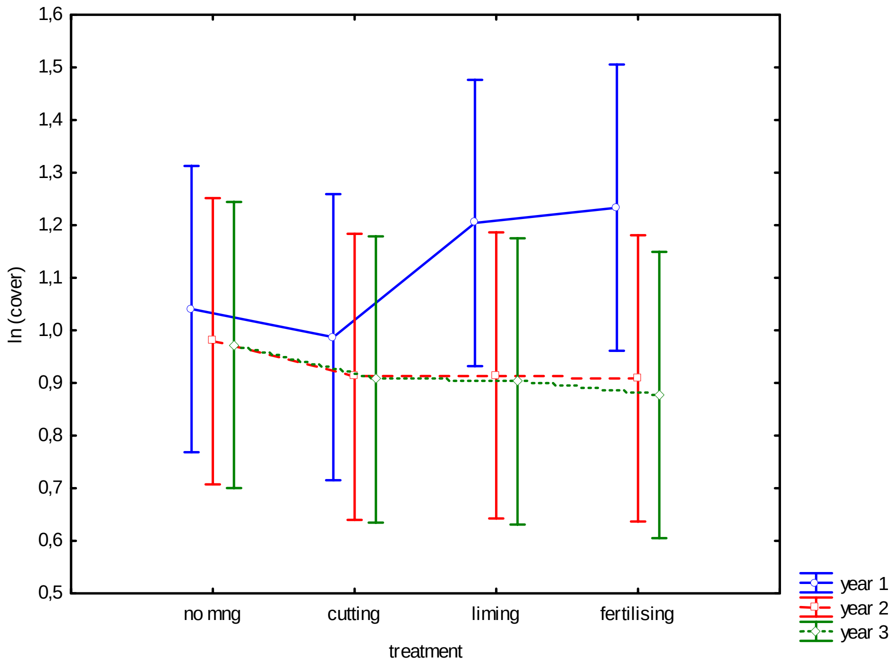


Fig. 17. The influence of interaction year*treatment on the cover (ln transformed) in the large squares independently of the meadow type. Treatments differ significantly (p=0.003) in their impact on the cover during the three seasons. The average cover of the meadow types during the tracking period: no mng: 75% - 74% - 74%. cutting: 73% - 69% - 68%. liming: 84% - 75% - 74%. fertilizing: 84% - 74% - 70%.

The influence of the interaction meadow type*treatment on Shannon-Wiener index was not significant (*H*‘ derived from biomass: F(6)=0.366, p=0.899, cover in the small squares: F(6)=1.248, p=0.291, cover in the large squares: F(6)=1.68, p=0.136).

Nevertheless, the diversity increased almost in all meadows, treatments and *H*‘ indices during the tracking period (*H*‘ derived from both biomass and cover in the small and large squares). Slight decrease of Shannon index was observed only in untreated *Molinion* and cut and limed *Carex* community in the case of *H*‘ derived from the biomass data.

In untreated *Molinion* slightly decreased *H´* (biomass) from 1.39 in the first year to 1.28 in last season, whereas in all treatments the *H´* index increased, the most in fertlized plots: from 1.09 to 1.43. Diversity index *H*‘ (cover in the small squares) increased in all treaments of the *Molinion* including the control. The biggest increase also occurred in fertlized plots: from 0.83 to 1.3. The situation was very similar in the large squares. The average value of *H´*(cover in the large squares) increased in all treatments and the biggest increase in fertlized and cut squares was recorded.

Carex brizoides community had higher species richness compared to *Calamagrostis villosa* community, but it had lower diversity values. *H´* index values (biomass) in *Carex* community did not change in all treatmets during the three years. However, *H*‘ indices derived from both covers increased in all treatments. These values ranged between 0.75 a 0.96.

All diversity indices slightly increased during studied period in *Calamagrostis villosa* community in all treatments. The most rapid increase occurred in limed large squares: from 0.8 to 1.07.

Relations between individual species (independent of meadow type), management treatments, years and cover of plants and mosses were tested in CANOCO by Canonical Correspondence Analysis (CCA). The Monte Carlo permutation test confirmed the significance of the whole model (F = 51.047, $p < 0.002$) (Fig. 18). The first canonical axis explained 78.6% of variability, second axis explained 8.3% of variability. Cutting, fertilizing and year 2 had only small influence, but were also significant. The CCA ordination diagram implies that species such as *Ranunculus auricomus*, *Sangusiorba officinalis* and *Caltha palustris* were positively correlated with fertilizing and species *Myosotis nemorosa*, *Galium uliginosum*, *Cirsium palustre* and *Juncus effusus* with liming. Although the effect of cutting was minimal, the occurrence of *Anemone nemorosa*, *Ranunculus auricomus* and *Senecio ovatus* were positively correlated with it. The impact of the years was not correlated with plant cover. Plant cover correlated best with the first axis. *Carex brizoides* and *Scirpus sylvaticus* occurred in stands with high plant cover. The cover of mosses was negatively correlated with plant cover. The characteristic species growing in places with a higher cover by mosses is *Calamagrostis villosa*, which forms open stands with low cover. *Calamagrostis* and seedlings of *Sorbus aucuparia* and spruce *Picea abies* were negatively correlated with cover E1. The occurrence of most of the species was affected by the year and it was not correlated with plant cover. Over time the species *Holcus mollis* spread considerably in the *Molinion* at the expense of the species *Molinia caerulea*. Parallel rapid retreat of *Molinia* was observed in all treatments. For example, in the first season, *Holcus* occurred only in one untreated plot (of four plots) with 5% cover, whereas in the second year it was observed in all four squares with an average cover of 12.5%, and reached as high as 16% in the last season. Simultaneously the cover of *Molinia* decreased from initial 66% to 49%. This trend was the most evident in cut squares, the average cover of *Holcus* increasing from 16% to 53% during tracking period, whereas *Molinia* decreased from 65% to 19%. Neither limed nor fertilized plots showed such extreme results. They differed only in having a relatively higher cover of *Molinia* and lower cover of *Holcus* in fertilized plots compared to limed squares.

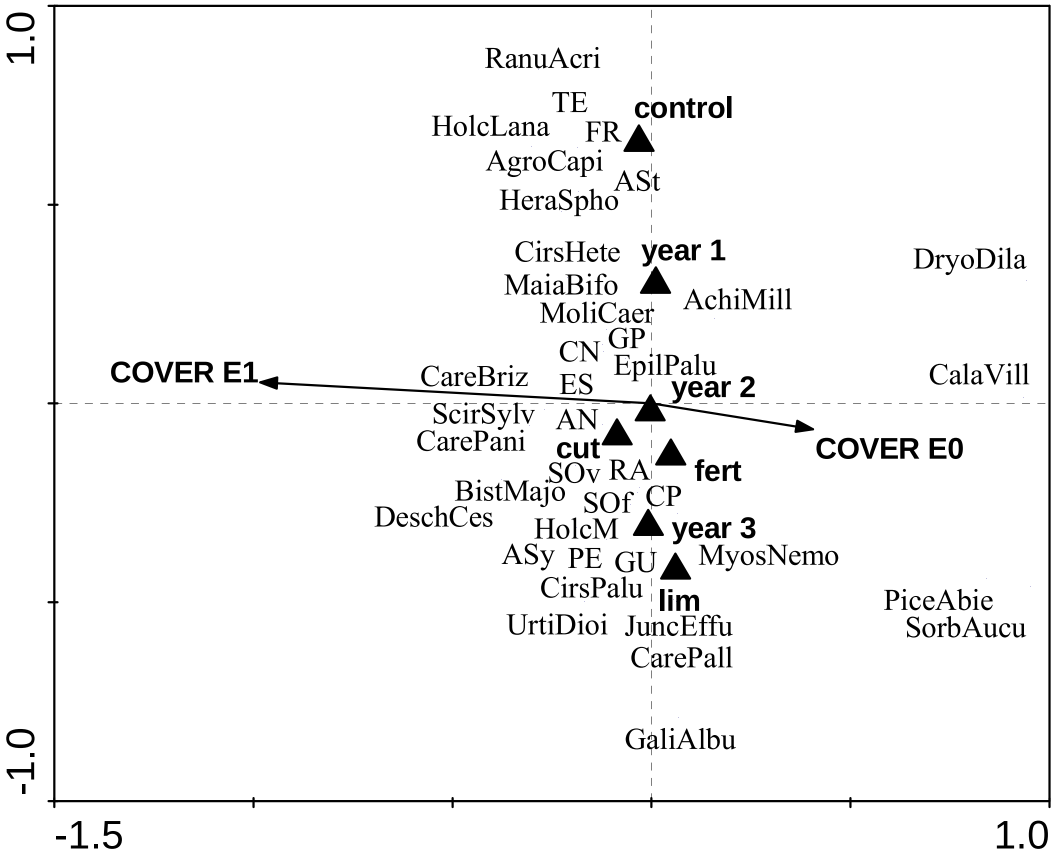


Fig. 18. CCA ordination diagram displaying the relation of species of three meadow types, explanatory variables (treatments and years) and response (cover E1 and E0). Management treatments were : cut - cutting, fert - fertilizing, lim - liming. Responses were: COVER E1 - cover of vascular plants, COVER E0 - cover of mosses. 1st axis explains 78.6% of variability, 2nd axis explains 8.3% of variability. Summary of Monte Carlo permutation test: Eigenvalue = 0.786, F = 51.047, p < 0,002. Abbreviations of species names represent first four letters of the genus and species. Exceptions are: **AN** - *Anemone nemorosa*, **ASt** - *Agrostis stolonifera*, **ASy** - *Angelica sylvestris*, **CN** - *Carex nigra*, **CP** - *Caltha palustris*, **ES** - *Equisetum sylvaticum*, **FR** - *Festuca rubra*, **GP** - *Galeopsis pubescens*, **GU** - *Galium uliginosum*, **HolcM** - *Holcus mollis*, **PE** - *Potentilla erecta*, **RA** - *Ranunculus auricomus*, **SOf** - *Sanguisorba officinalis*, **SOv** - *Senecio ovatus*, **TE** - *Trientalis europaea*.

Relations between individual species (independent of meadow type), management treatments, years and biomass of plants were also tested in CANOCO by Canonical Correspondence Analysis (CCA). The Monte Carlo permutation test confirmed the significance of the whole model (F = 17.622, p < 0.002) (Fig. 19). The first canonical axis explained 40.3% of variability, second axis explained 7.9% of variability.

The CCA ordination diagram implies that the biomass correlated best with the first axis. The most of species are not correlated with the biomass. These species, such as *Deschampsia cespitosa*, *Trientalis europaea*, *Cirsium palustre*, *Agrostis stolonifera*, *Holcus mollis*, *Bistorta major* and others, are capable to grow both in dense and thinner stands. Many species studied are negatively correlated with the biomass. This group of species, favouring thinner stands, is represented by e.g. *Calamagrostis villosa*, *Galeopsis pubescens*, *Galium uliginosum* and *Galium palustre*. On the other hand, *Carex brizoides* and *Scirpus sylvaticus* occurred in stands with high plant biomass. The occurrence of most of the species is affected by the year. The impact of the years was not correlated with plant biomass. Cutting, fertilizing and year 2 had only small influence, but were also significant. Although the effect of cutting and fertilizing was minimal, the occurrence of *Bistorta major* was positively correlated with it. *Juncus effusus* was positively correlated with liming.

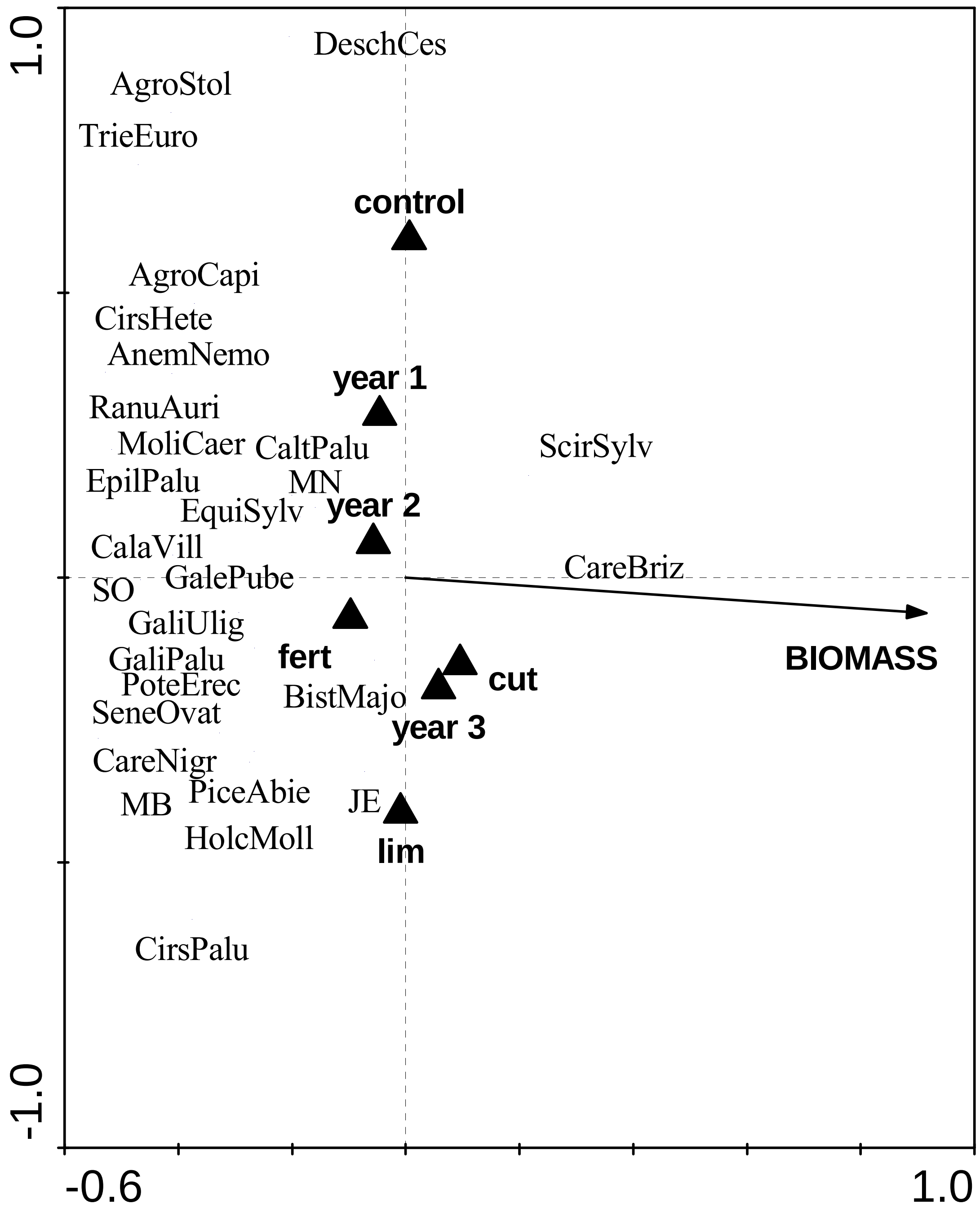


Fig. 19. CCA ordination diagram displaying the relation of species of three meadow types, explanatory variables (treatments and years) and response (biomass). Management treatments were : cut - cutting, fert - fertilizing, lim - liming. 1st axis explains 40.3% of variability, 2nd axis explains 7.9% of variability. Summary of Monte Carlo permutation test: Eigenvalue = 0.403, F = 17.622, p < 0,002. Abbreviations of species names represent first four letters of the genus and species. Exceptions are: **JE** - *Juncus effusus*, **MB** - *Maianthemum bifolium*, **MN** - *Myosotis nemorosa*, **SO** - *Sanguisorba officinalis*.

4.2 The statistical evaluation of four meadow types in two seasons

4.2.1 The comparison of meadows

All four plant communities studied (*Molinion*, *Carex* community, *Calamagrostis* community, *Filipendula* community) differ significantly in all measured dependent variables: the amount of biomass (on an indicative level of significance, F(3)=96.426, p=0.06), the cover in the small squares (F(3)=39.482, p<0.001), the cover in the large squares (F(3)=55.524, p<0.001) and in three diversity values: Shannon index *H'* derived from biomass (F(3)=42.533, p<0.001), cover in the small squares (F(3)=65.395, p<0.001) and cover in the large squares (F(3)=240.791, p<0.001).

Figure 20 shows the differences between four meadow types in the cover in the small squares independently of the year and treatment. Graphs displaying the differences in the biomass and cover in the large squares are not shown, because they are almost identical with figure 20.

The biomass and the cover of *Carex* community was higher considerably compared to other meadows. The cover and the biomass decrease in the order *Carex* community - *Molinion* - *Filipendula* community - *Calamagrostis* community. Only in the case of the biomass, the values in *Molinion* and *Filipendula* community were stable and this amout was higher only slightly compared to *Calamagrostis* community, whereas it was higher considerably in the case of both covers. The cover in the large squares differed from the cover in the small squares in slightly higher values in all meadows and also in lesser range of values in *Molinion*.

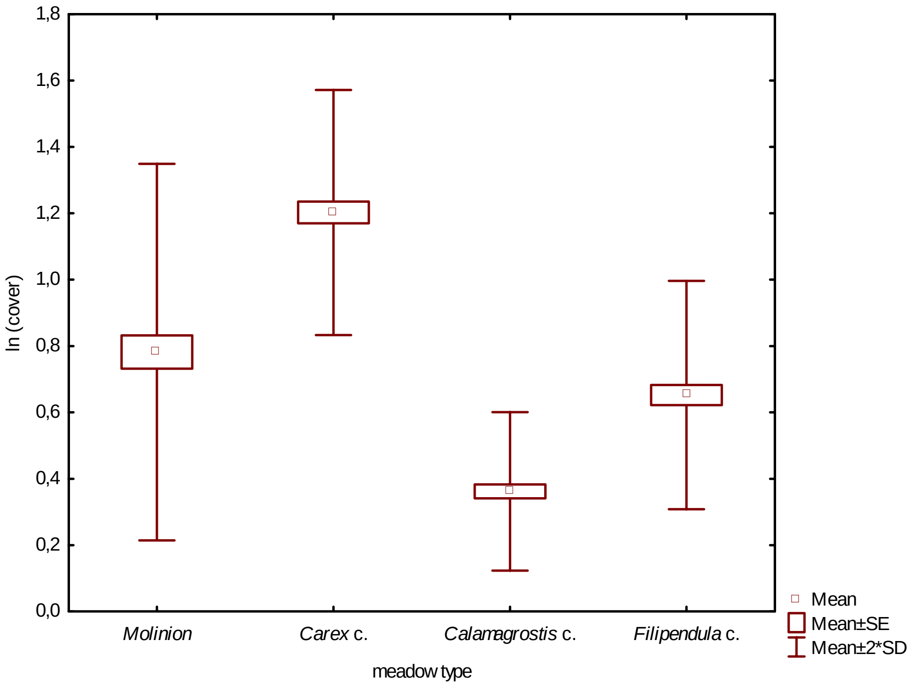


Fig. 20. The comparison of the cover in the small squares (ln transformed) of all four meadow types independently of the year and treatment. The readings for the cover differ significantly ($p<0.001$): *Molinion* 68%, *Carex* community 92%, *Camagrostis* community 35%, *Filipendula* community: 60%.

The comparison of the cover and biomass amount during the tracking period independently of the treatment and meadow type, i.e. „the impact of the year“ both on cover and biomass, provided the following results. No significant changes in the cover were recorded ($p>0.05$). The comparison of the biomass during the two years is shown in the following graph (Fig. 21). Independently both on meadow type and treatments, the significant increase ($F(1)=33.647$, $p=0.039$) in the biomass occurred during the monitored period: biomass was lower in the second season (23g) compared to the last year (40g) as well as the range of values.

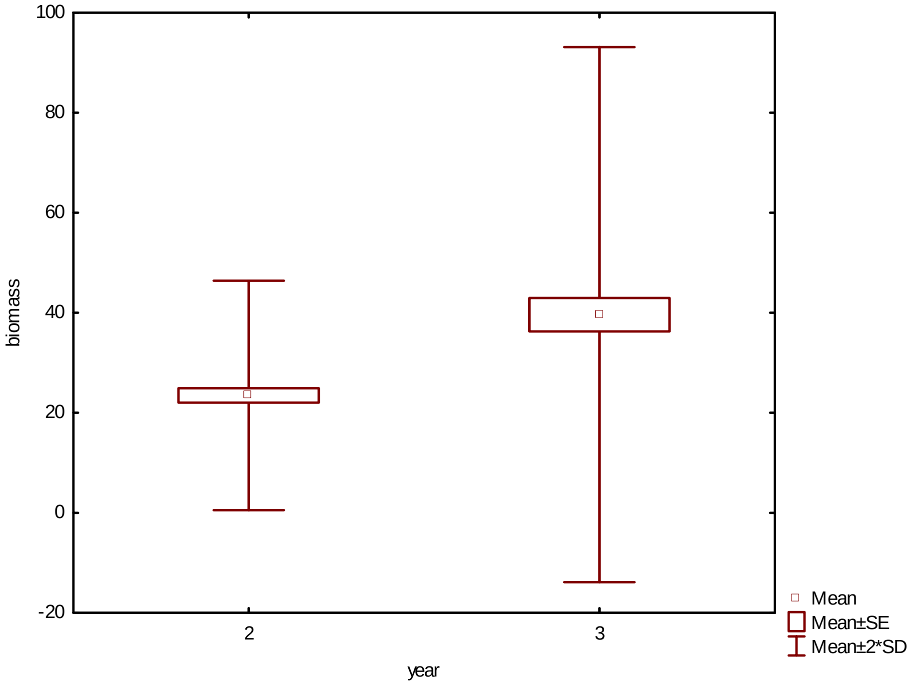


Fig. 21. The comparison of the biomass during the tracking period independently of the treatment and meadow type. The readings for the biomass differ significantly (p=0.039): 23g (year 2), 40g (year 3).

The interaction meadow type*year independently of the treatments had significant impact both on biomass: F(3)=9.679, p<0.001 the cover in the large squares: F(3)=4.059, p=0.01 and the Shannon index derived from biomass data: F(3)=4.329, p=0.007. The impact was not significant in the case of the cover in the small squares. These results mean, that all meadow types differ significantly with each other within two years in the biomass, the cover in the large squares as well as the diversity. Graph (Fig. 22) shows the significant changes in biomass of the meadows during the monitored period. The biomass increased in all meadows, but with different intensity. The lowest increase in *Calamagrostis* community was recorded (14g and 19g), which was the most sparse stand of all meadows studied. The more rapid increase in both *Filipendula* community (24g and 31g) and *Molinion* (21g and 35g) was recorded, whereas the quickest increase had *Carex* community (34g and 73g).

The significant changes in the cover in the large squares shows the graph (Fig. 23). The cover increases in the order *Calamagrostis* community - *Filipendula* community - *Molinion* - *Carex* community. The differences between the second and the last season were not too large. Whereas *Carex* and *Filipendula* communities increased slightly the cover, in the *Molinion* the opposite trend was recorded. The slight decrese in *Calamagrostis* community also occurred.

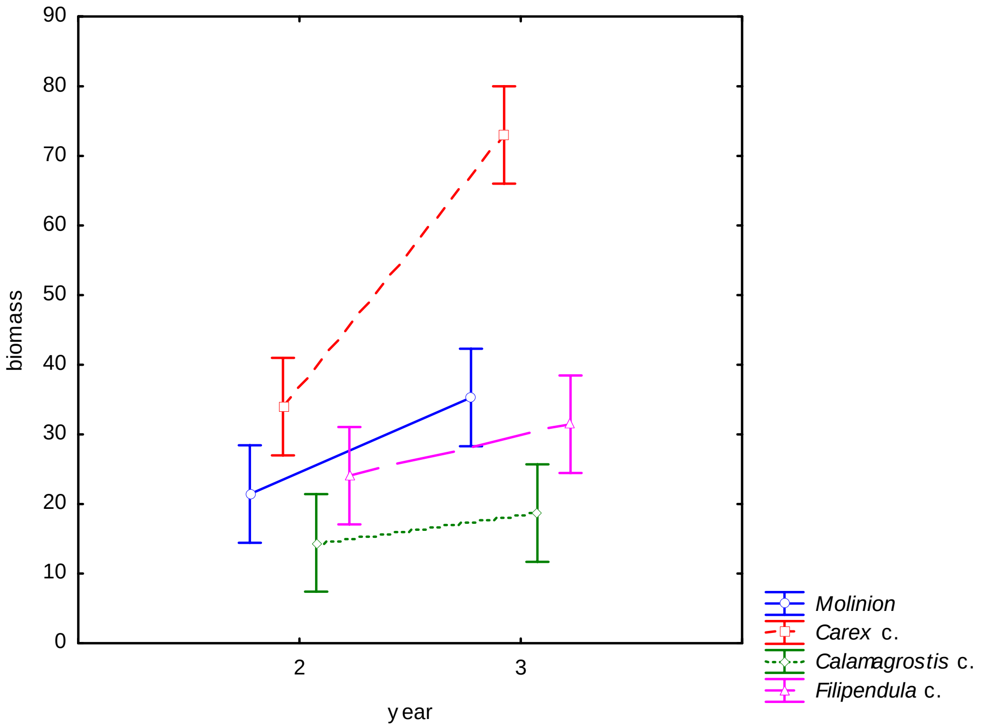


Fig. 22. The influence of interaction meadow type*year on the biomass independently of management. The readings for the biomass of four plant communities significantly differ ($p<0.001$) during the second and the third year. The average weight of the dry matter: *Molinion*: 21g - 35g. *Carex* community: 34g - 73g. *Calamagrostis* community: 14g - 19g. *Filipendula* community: 24g - 31g.

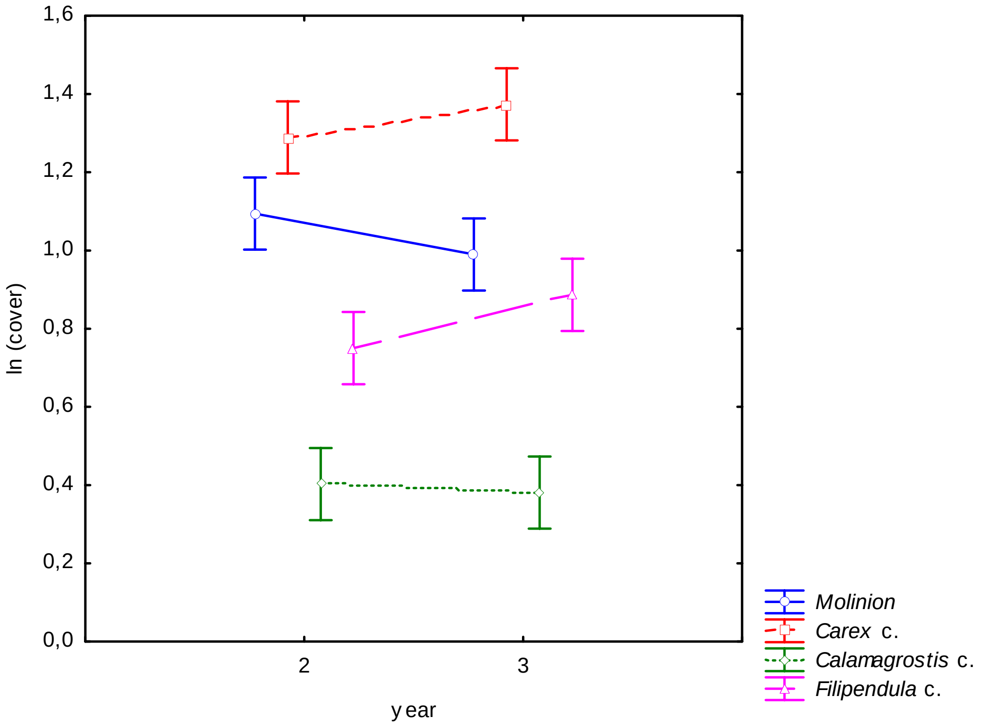


Fig. 23. The influence of interaction meadow type*year on the cover in the large squares (ln transformed) independently of management. The readings for the cover of four plant communities significantly differ ($p=0.01$) during the second and the third year. The average cover: *Molinion*: 87% - 83%. *Carex* community: 94% - 96%. *Calamagrostis* community: 39% - 37%. *Filipendula* community: 67% - 77%.

All the four meadow types studied (*Molinion*, *Carex* community, *Calamagrostis* community and *Filipendula* community) differed significantly in Shannon-Wiener diversity index independently of both the year and management, derived from both biomass: F(3)=42.533, p<0.001, cover in the small squares: F(3)=65.395, p<0.001 and cover in the large squares: F(3)=240.791, p<0.001. There was virtually no difference between all four graphs, thus only on is shown: graph displaying the differences between meadows in *H´* index values derived from the data of the cover in the large squares (Fig. 24). The highest values of Shannon diversity were in *Filipendula* community (*H´*=1.55), lower index was recorded in the *Molinion* (*H´*=1.27) and *Calamagrostis* community (*H´*=1.03) and the lowest values in *Carex* community were recorded (*H´*=0.83). Graph with diversity values derived from the cover in the small squares differs only in having slightly lower values in all meadows and wider range of values of the *Molinion*.

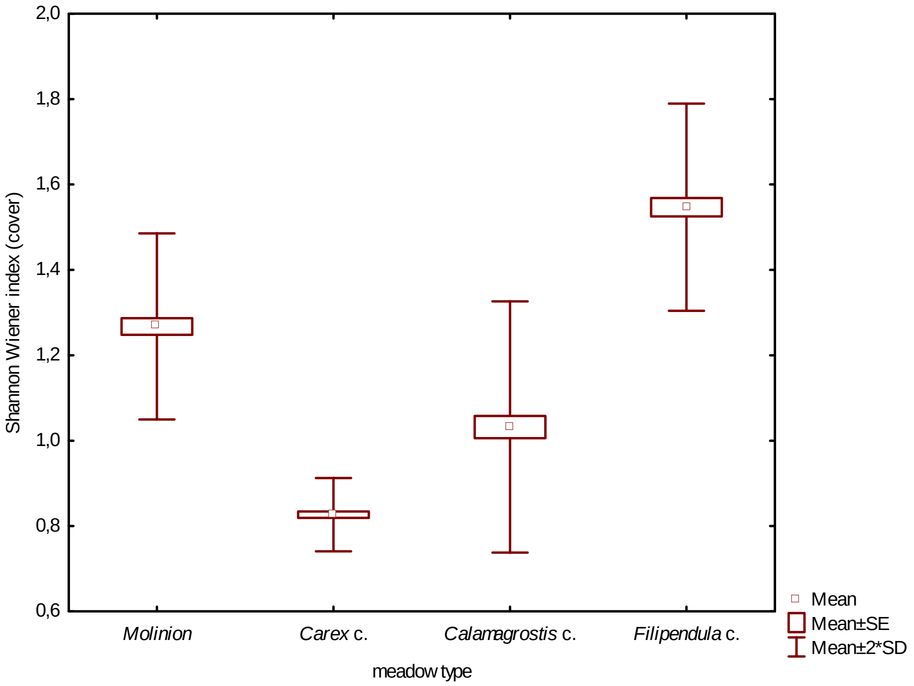


Fig. 24. The comparison of the Shannon-Wiener index (derived from the cover in the large squares) of all four meadow types independently of the year and treatment. Shannon index (*H´*) values differ significantly (p<0.001): *Molinion* 1.27, *Carex* community 0.83, *Calamagrostis* community 1.03, *Filipendula* community 1.55.

The interaction meadow type*year independent of treatments was significant for *H´* index derived from the biomass: F(3)=4.329, p=0.007 (Fig. 25). Thus, plant communities differ from each other significantly during the tracking period in the *H´* index derived from biomass data. The interaction was not significant in covers. The lowest *H‘* index had the

Carex community all the time. The *Calamagrostis* community had higher values. In both of them, slight decrease in the diversity between the second and the third season was observed. Both *Molinion* and especially *Filipendula* community had higher values of Shannon diversity compared to previous meadows, and the values increased during the tracking period.

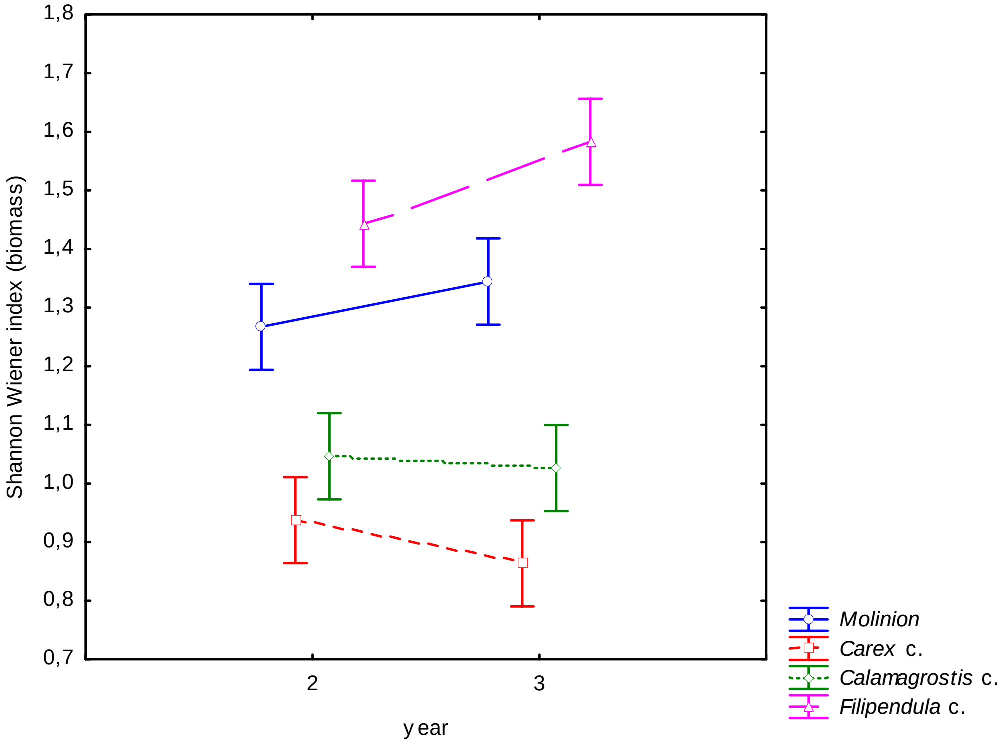


Fig. 25. The influence of interaction meadow type*year on the *H´* index (biomass) of plant communities independently of treatments. *H'* index values of the four meadows differ significantly (p=0.007) during the second and the third year. *Molinion*: 1.27 - 1.34. *Carex* community: 0.94 - 0.86. *Calamagrostis* community: 1.05 - 1.03. *Filipendula* community: 1.44 - 1.58.

The biomass and both covers were tested by Correspondence Analysis (CA). All three ordination diagrams were almost identical, therefore only one of them is used (biomass, Fig. 26). This ordination displays the relation of species (8% restriction was used) of the four meadows independently of the management. The first canonical axis explained 94.2% of variability, second axis explained 77.5% of variability. The position of species in the ordination diagram is very similar to the species distribution showed in Fig. 12 - where only three meadows and all three years were considered. Plant species are spatially very distinctive and they also form three separated aggregates within the ordination. One of them is the species *Carex brizoides*, favouring entirely different conditions compared to *Calamagrostis villosa*. The third group represents species all present in *Filipendula* community (e.g.

Equisetum fluviatile, *Eriophorum angustifolium*), but some of them were present also in other meadows, mainly in the *Molinion*, e.g. *Agrostis stolonifera* and *Carex nigra*. These species do not grow with *Carex brizoides*, e.g. they prefer stands without the occurrence of *Carex brizoides*. The position of *Molinia caerulea* and *Bistorta major* in the ordination suggests the spatial difference from *Calamagrostis villosa*.

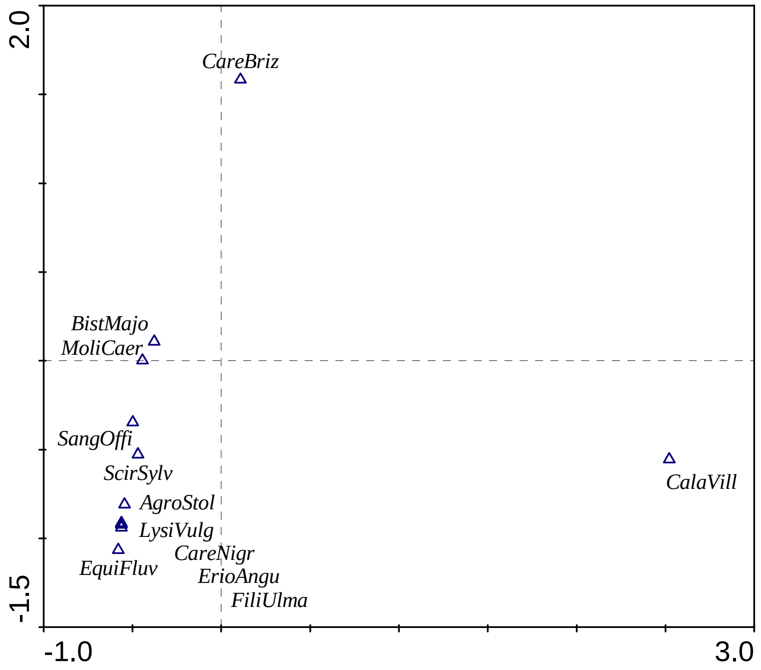


Fig. 26. CA ordination diagram displaying the relation of species (8% species restriction) of the four meadow types (biomass) independently of the treatments. 1st axis explains 94.2% of variability, 2nd axis explains 77.5% of variability. Abbreviations of species names represent first four letters of the genus and species (see tab. 3).

4.2.2 The impact of management treatments on meadow characteristics

The interaction meadow type*treatment independently of year had significant influence only on the cover in the large squares: F(9)=3.802, p<0.001 (Fig. 27). Thus, treatments differed significantly in their impact on the cover of meadow types. The cover in meadows decreases in the order *Carex* community - *Filipendula* community - *Molinion* - *Calamagrostis* community. The most variable values of the cover within the treatments had *Calamagrostis* community, the most stable values had *Filipendula* community. The response of the cover to the management treatments varied between the meadow types. Cutting decreased the cover in all meadows except of *Carex* community, where the increase was observed. Liming decreased the cover in all meadows except of *Calamagrostis* community. Fertilising increased the cover only in *Calamagrostis* community, whereas in both *Carex* community and *Molinion* the decrease was observed. *Filipendula* community had the same cover in fertilized and untreated plots.

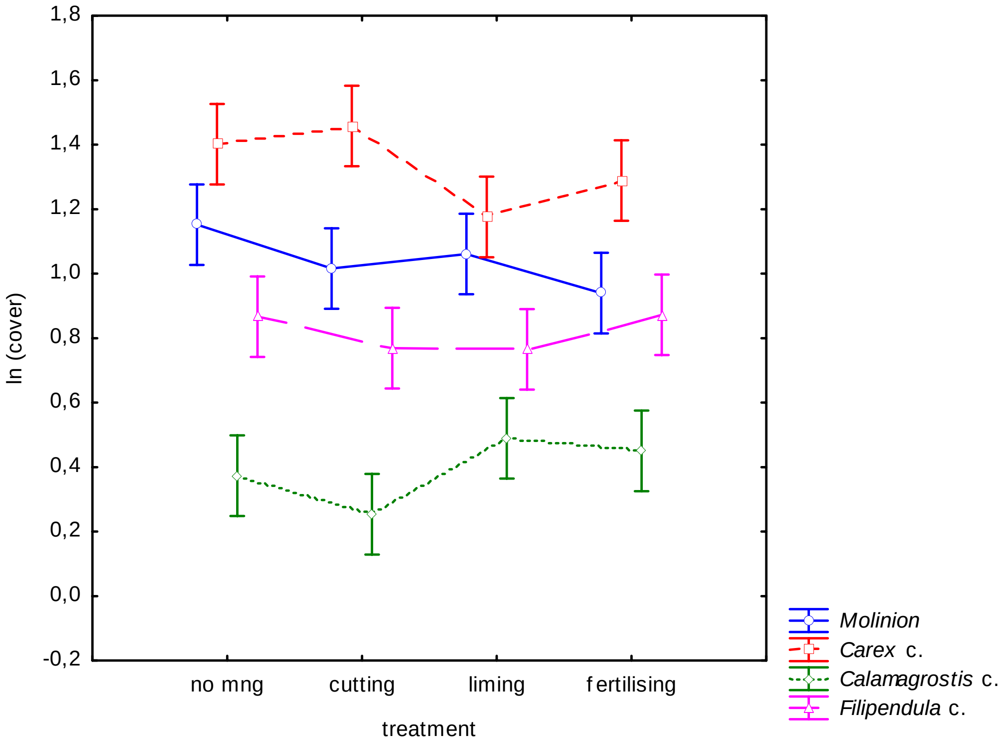


Fig. 27. The influence of interaction meadow type*treatment on the cover (ln transformed) in the large squares of four plant communities independently of the year. Treatments differ significantly ($p<0.001$) in their impact on the cover in all meadows. The average cover in the large squares within the management treatments: *Molinion*: 90% - 84% - 85% - 80%. *Carex* community: 96% - 98% - 91% - 94%. *Calamagrostis* community: 36% - 25% - 47% - 43%. *Filipendula* community: 75% - 68% - 69% - 76%.

The interaction year*treatment independently of meadow types had significant influence only on biomass: F(3)=2.821, p=0.045, e.g. treatments had different impact on biomass independently of the meadow type during the two years. These biomass changes under different treatments within the two years are shown in Fig. 28. Generally, the biomass was higer in the third season compared to the second year. The biomass amount was lower in all treatments compared to the control in both seasons. The biomass values were the same in cut and limed plots in the second season (21g) as well as in the last year (38g).

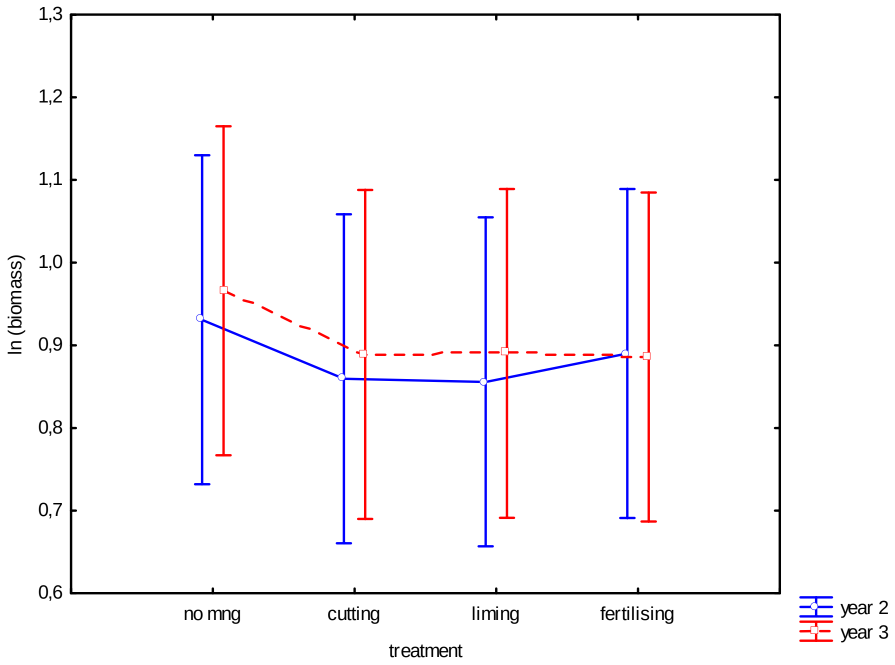


Fig. 28. The influence of interaction year*treatment on the biomass (ln transformed) of plant communities independently of the meadow type. Treatments differ significantly (p=0.045) in their impact on the biomass during the two seasons. The average weight of the dry matter within the meadow types during the tracking period: no mng: 27g - 49g. cutting: 21g - 38g. liming: 21g - 38g. fertilizing: 25g - 34g.

The influence of management on Shannon-Wiener index was not significant when four meadow types and two years are considered. Nevertheless, obvious changes in *H´* values occurred. The *Molinion*, *Carex* community and *Calamagrostis* community diversity changes were mentioned on page 54. In *Filipendula* community, the Shannon index was monitored only in the second and the third season. In this period, the index (both biomass and covers) increased in all treatments, except of the cover in the small cut squares, where the diversity remained unchanged. The *H´* values ranged between 1.35 and 1.65 within the whole latin square. The most rapid increase occurred in biomass in small fertilized squares: from 1.4 to 1.61.

Relations between individual species (independent of meadow type), management treatments, years, biomass and cover of plants and mosses were tested in CANOCO by Canonical Correspondence Analysis. The Monte Carlo permutation test confirmed the significance of the whole model (F = 17.289, p < 0,002) (Fig. 29). The first canonical axis explained 54.6% of variability, second axis explained 14.5% of variability. Cutting, liming and fertilizing are not displayed, becuase they did not pass through the forward selection. The

CCA ordination diagram implies that the impact of the years was not correlated with plant cover. The cover of *Agrostis stolonifera*, *Agrostis capillaris* and *Eriophorum angustifolium* increased with time. In contranst, *Galeopsis pubescens* was negatively correlated with time. Plant cover correlated best with the first axis and the total biomass was positively correlated with it. The occurrence of *Juncus effusus* was positively corelated with the cover and *Achillea ptarmica* with the biomass. *Carex brizoides* was positively correlated with both variables. Species such as *Cirsium heterophyllum*, *Cirsium palustre*, *Ranunculus auricomus*, *Myosotis nemorosa* were not correlated with the cover. The lower value of the biomass was preferred by the species of the *Molinion* and wet *Filipendula* grasslands: *Carex nigra*, *Filipendula ulmaria*, *Lysimachia vulgaris*, *Angelica sylvestris*, *Carex panicea* and others. Species favouring stands with low cover were *Carex rostrata* and *Equisetum fluviatile* (species of wet *Filipendula* grasslands), *Calamagrostis villosa* and seedlings of spruce *Picea abies*. The cover of mosses was not correlated with plant cover. The occurrence of most of the species was affected by the year; these species prefer places without the occurrence of mosses.

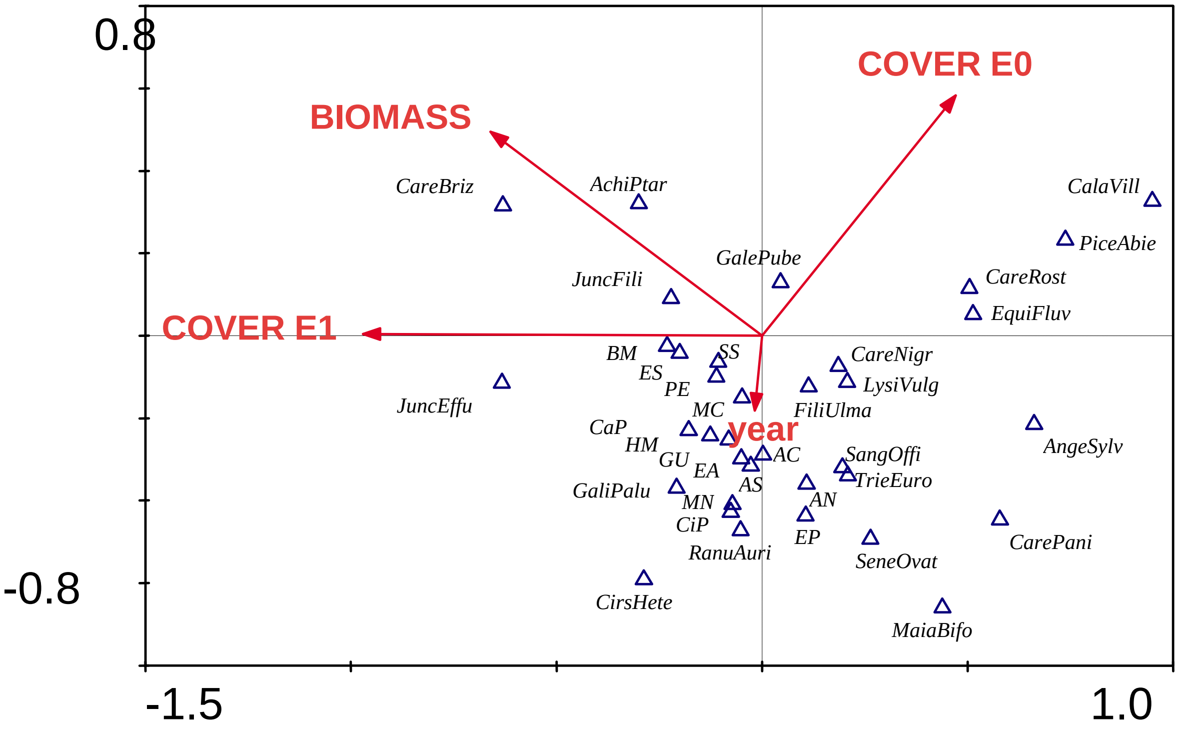


Fig. 29. CCA ordination diagram displaying relation of species of four meadow types, explanatory variables (treatments and years) and responses (biomass, cover E1 - vascular plants and cover E0 - mosses). Forward selection was used: cutting, liming and fertilizing did not pass through. 1st axis explains 54.6% of variability, 2nd axis explains 14.5% of variability. Summary of Monte Carlo permutation test: Eigenvalue = 0.55, F = 17.3, p < 0,002. Abbreviations of species names represent

first four letters of the genus and species. Exceptions are: **AC** - *Agrostis capillaris*, **AN** - *Anemone nemorosa*, **AS** - *Agrostis stolonifera*, **BM** - *Bistorta major*, **CaP** - *Caltha palustris*, **CiP** - *Cirsium palustre*, **EA** - *Eriophorum angustifolium*, **EP** - *Epilobium palustre*, **ES** - *Equisetum sylvaticum*, **GU** - *Galium uliginosum*, **HM** - *Holcus mollis*, **MC** - *Molinia caerulea*, **MN** - *Myosotis nemorosa*, **PE** - *Potentilla erecta*, **SS** - *Scirpus sylvaticus*.

Relations between individual species (independent of meadow type), management treatments, years and cover of plants and mosses in the small squares were tested by CCA. The Monte Carlo permutation test confirmed the significance of the whole model ($F = 15.981$, $p < 0.002$) (Fig. 30). The first canonical axis explained 52.1% of variability, second axis explained 9.5% of variability. CCA ordination diagram implies that cutting and fertilizing had only small influence, but were also significant. Although the effect of cutting was minimal, the occurrence of *Caltha palustris*, *Equisetum sylvaticum*, *Scirpus sylvaticus* and *Bistorta major* were positively correlated with it. All treatments were positively correlated with the year. The species positively correlated with the time were *Galeopsis pubescens*, *Cirsium palustre* and *Eriophorum angustifolium*, which preferred limed places at the same time. On the other hand, *Agrostis capillaris* and *Trientalis europaea* were negatively correlated with all management treatments. Treatmets and years were not correlated with the plant cover, which correlated best with the first axis. *Carex brizoides* grows on places with high cover, and, on the contrary, *Calamagrostis villosa*, *Equisetum fluviatile*, *Angelica sylvestris*, *Carex panicea*, *Maianthemum bifolium* and seedlings of spruce *Picea abies* were negatively correlated with cover E1. The lower value of the cover E1 was preferred also by the species of wet *Filipendula* grasslands: *Carex nigra*, *Filipendula ulmaria* and *Lysimachia vulgaris*. Several species were not correlated with the cover of plants and mosses: *Ranunculus auricomus*, *Cirsium heterophyllum* and particularly *Achillea ptarmica*. The cover of mosses was not correlated with plant cover. The species growing in places with a higher cover by mosses is *Carex rostrata*. The occurrence of most of the species was affected by the year and it was negatively correlated with cover of mosses.

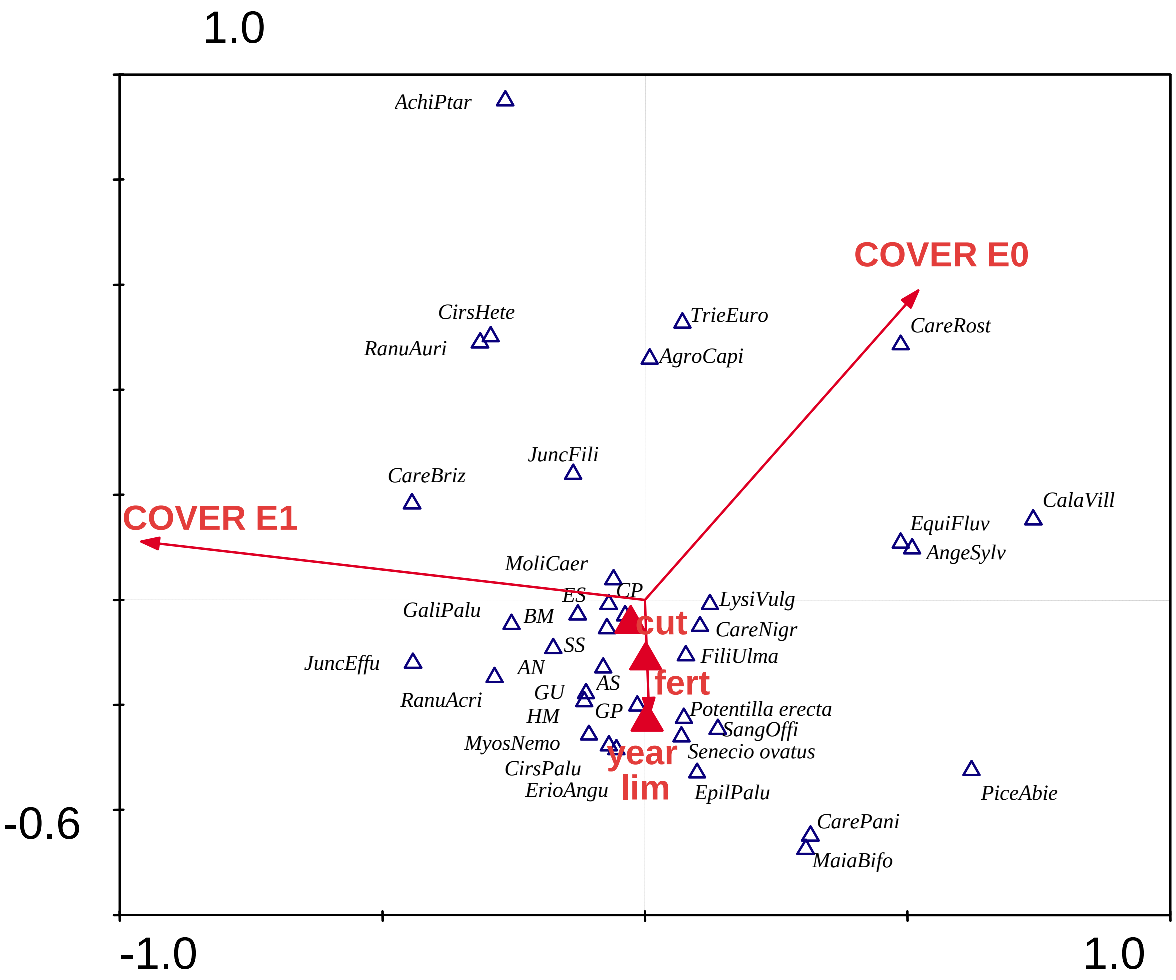


Fig. 30. CCA ordination diagram displaying relation of species of four meadow types, explanatory variables (treatments and years) and response (cover E1 and E0). Management treatments were: cut - cutting, lim - liming, fert - fertilizing. No restriction was used. 1st axis explains 52.1% of variability, 2nd axis explains 9.5% of variability. Summary of Monte Carlo permutation test: Eigenvalue = 0.52, F = 15.981, p < 0,002. Abbreviations of species names represent first four letters from either name of species. Exceptions are: **AN** - *Anemone nemorosa*, **AS** - *Agrostis stolonifera*, **BM** - *Bistorta major*, **CP** - *Caltha palustris*, **ES** - *Equisetum sylvaticum*, **GP** - *Galeopsis pubescens*, **GU** - *Galium uliginosum*, **HM** - *Holcus mollis*, **SS** - *Scirpus sylvaticus*.

5. Discussion

Molinion

Among all meadow types studied, *Molinion* had high values of both the cover and biomass independently of the treatment; only *C. brizoides* community had higher values (Fig. 20). The average biomass gradually increased independently of the treatment within the tracking period, but, in contrast, the average cover slightly decreased. In term of the diversity, *Molinion* represents the stand with the second highest values of the Shannon diversity (following *Filipendula* community) and this index increased during the tracking period (Fig. 11).

It is known that *Molinia caerulea* is a species which is relatively very sensitive to cutting and grazing (Grant et al. 1996) and it particularly recedes from grasslands following early cutting (approximately mid-July) (Blažková & Řehořek 1999). However, these generalities may not always be valid. For example, Hansson & Fogelfors (2000) observed an increase of *Briza media*, *Carex panicea* and *Molinia caerulea* densities in southern Sweden, although flowering and seed set by *Molinia* were reduced. Nevertheless, most papers confirm the decrease of *Molinia* cover after cutting and my results also indicate this. The average cover over the three seasons in cut squares (both large and small) was lower (85% in the large squares) compared to the control (89%). At the same time, this was the lowest value of all treatments.

Rychnovská et al. (1985) claimed that the species capable of quick regeneration after cutting, e.g. grasses, are always favoured. In contrast the competitiveness of slow-growing species which flower and ripen in late summer gradually decreases (namely *Molinia caerulea*), resulting in the decrease of their vitality and in their slow withdrawal from the stand (Rychnovská et al. 1985). Cutting experiments in southern Scotland (Grant et al. 1996) showed that frequency and severity of defoliation were more important than timing in their effects on *Molinia*. Weights of clippings declined in successive years only in response to treatments that involved repeated within-season cutting. Three years of repeated light defoliation (33% lamina length removed each June, July and August), compared with uncut controls, reduced leaf production in a fourth uninterrupted growing season by 40%, while repeated heavy defoliation (66% lamina removal) reduced it by 78%. Single annual cuts only reduced leaf production at 66% lamina removal when they took place late in the season (Grant et al. 1996).

The reduction of *Molinia* cover was also observed by Milligan et al. (2004). Cutting three times had the greatest effect, maintaining a reduced *Molinia* cover over four years. The most effective treatment was the combination of grazing and cutting thrice, which maintained a low *Molinia* cover for the longest period and had less variation in moorland species in the fourth year. Cutting once had very little impact compared to the uncut treatment (Milligan et al. 2004).

Jakrlová (1997) decribed the collapse of the *Molinia* community with 20 plant species after cutting three times per year. *Molinia* was replaced by *Agrostis canina* and the total number of species decreased. A similar phenomenon showing this species substitution also occurred in my *Molinion*, where *Molinia caerulea* markedly receded to the benefit of *Holcus mollis* (see page 39). This change occurred in all treatments including the control and it was most evident in the cut plots. In contrast to Jakrlová's experiment (Jakrlová 1997), my stand was cut only once a year and the number of species increased from 15 to 22.

The decrease in *Molinia* biomass is also particularly notable after cutting. Hakrová & Wotavová (2004) observed considerable decrease of biomass in the *Molinion* to a half or even one third of initial amount. It was due to the decrease of biomass of the dominant species. Jakrlová (1997) observed similar changes after three years in experimental plots mown once a year: the biomass decreased to approximately one third of the initial level in her experiment. In my study, the influence of cutting on the biomass of the *Molinion* was not proven. The observed figures for the biomass in cut and untreated plots were variable during the three seasons, but on average they showed no change. Milligan et al. (2004) observed, that the only treatment that had consistent effect on *Molinia* decrease and *Calluna* enhancement was cutting, where there was increased bare ground, reduced vegetation height and increased species diversity.

The response of botanical composition and species richness to the mowing of *Molinia* communities was varied. Both the changes in species composition and the decrease of species richness of the *Molinion* following cutting were significant in the experiment of Hakrová & Wotavová (2004). In contrast, Jongepierová et al. (1994) observed the increase of species richness in stands dominated by *Molinia* after 2 years. My results are in accordance with the observations of Jongepierová et al. (1994): species richness showed an increase as a result of all treatments by the second year of the study (table 5). There was further enhancement in the number of species in all squares by the third season. The maximum increase in the number of species was recorded in the mown squares. The increase in species richness in the *Molinia* community under a mowing regime was an effect also mentioned by Lepš (1999), who explored the response of a species-rich *Molinion* to fertilization, mowing and removal of dominant species over four years. Both species richness and seedling recruitment were positively influenced by mowing and to a lesser extent by removal of the dominant species (*Molinia*).

The influence of fertilising on the cover of *Molinia caerulea* has been sparsely discussed. Roem et al. (2002) described a significant increase of *Molinia* cover after NP treatment and only a slight increase after the separate N and P treatments. In my experiment, the average cover over three years was higher in the fertilized small squares compared to the control. In the first season, the cover of fertilized plots was higher than the control by as much as 19%, but a reduction in the overall cover over the three years was observed in both cases (slight in the control and more notable in the fertilized plots). Thus, the growth in the fertilized squares seem to be encouraged mainly in the first season. In the large squares, the average cover over three years was almost the same in fertilized plots and in the control, but whereas the cover remained almost unchanged in the control, the cover in fertilized plots decreased rapidly within three vegetation seasons from 100% to 75%. Here the cover after the first slurry application was also considerably higher than in untreated plots. These results show the most sensitive response of the cover follows the first fertilizer application.

Papers concerning the influence of fertilising on biomass production are more numerous. Lepš (1999) observed an immediate increase of biomass of species-rich *Molinion*. Nitrogen application in the experiment of Tomassen et al. (2003) also significantly increased the above-ground biomass of a *Molinia* community. Individual above-ground biomass of *Molinia* increased by more than threefold. Other works confirm the same with NP fertilization (Roem et al. 2002) and with NPK (Aerts & Caluwe 1989). In my study, the significant impact of fertilization on biomass was not proven, although the total biomass from the small squares increased and the average biomass of fertilized plots was higher over the three years compared to the control. The increasing trend in biomass production is evident. In the experiment of van der Hoek et al. (2004), N addition boosted biomass production of *Molinia caerulea* and *Carex panicea*, but only during the first year. The species composition was not changed. P fertilization increased the biomass production in second season and changed the species composition from a vegetation dominated by *Carex panicea* to a grassland community with abundant *Holcus lanatus*.

The impact of fertilization on Shannon-Wiener index of the *Molinion* was not significant in my study, but the increase of *H´* was observed in all treatments. The biggest increase occurred in fertilised plots. The results of Roem et al. (2002) showed that acidification was the most important factor in reducing species diversity in *Molinia* meadows. Species richness was clearly influenced by the soil pH, but not by nutrient availability treatments (N, P, NP). Species richness clearly increased during the first years following only the P treatment but this effect later diminished. In contradiction of these observations, Lepš (2004) recorded a decrease of both species number and diversity following fertilization. Shift in community composition was also evident.

Roem et al. (2002) found out that liming increased species richness in *Molinia caerulea*, *Calluna vulgaris* and *Erica tetralix* dominated heathland. The increase was quick during the first years of the experiment, because tree seedlings appeared. Later the number of species decreased slightly and a few years later, the number of species was not significantly different from the control. My results were very similar. The number of species in limed *Molinion* plots increased from 14 in the first year to 19 in both second and the third year. Roem et al. (2002) claimed that neither the above-ground biomass nor the cover of three dominant species (*Molinia*, *Calluna* and *Erica*) was significantly influenced by liming and acidification. My limed small squares showed higher values of cover than the control (the average of three years): 83% and 67%. The biomass changes following liming were not significant.

Roem et al. (2002) also mentioned the increase of the number of bryophyte species by liming. Individual bryophyte species were not observed in this work. Results showed the increase of bryophyte cover in limed plots during three years. In the first vegetation season, the moss occurred only in one limed plot with negligible cover, whereas in next two years it occurred in all plots with the average cover 1% and 7%, respectively.

Carex brizoides community

Carex brizoides community had significantly the highest values of both the biomass and cover in both the small and large squares independently of the treatments compared to other meadows studied. The total biomass in small squares (independently of treatments) slightly decreased in the second season but in contrast it increased markedly in the last year. The total biomass in small squares (independently of treatments) slightly decreased in the second season but in contrast it increased markedly in the last year. The cover of the *Carex* community in the large squares hardly changed during the monitored period, but a slight decrease was observed in the small squares. It does not correspond so much to the considerable increase of biomass in the last season. Nevertheless, *Carex brizoides* is the species with a considerable length of decumbent leaves, typically forming curly stands. As a result of these features, it frequently forms monodominant stands with high % cover. Hence the % cover is not much affected by an increase of biomass. In my case, the bare ground occurred sporadically in the second and the third season, in spite of the increase in the biomass. Apparent incompatibility between cover and biomass trends were also described by Štursová (1985), who monitored biomass and cover changes of *Nardus stricta* following liming. The above-ground biomass was markedly increased by a very high number of Mat Grass panicles, although the actual cover did not change very much. Thus, the decrease of cover does not necessarily mean the decrease of biomass (Štursová 1985).

The results of Hakrová & Wotavová (2004) are in accordance with results of Vacková (1997), i.e. both the total biomass and the biomass of *Carex brizoides* alone decreased after cutting. However, the biomass of *Carex* vigorously increased in one of four experimental plots of Hakrová & Wotavová (2004). In my study, the biomass of the *Carex* community was not significantly affected by cutting. Nevertheless, the biomass decreased following cutting in the second season and than increased. This effect was more obvious in the control. The biomass of both limed and fertilized *Carex* increased equally during the tracking period, but this increase was more rapid in the case of limed plots.

The *C. brizoides* community had the lowest diversity compared with other meadows and the increase of Shannon index increased only slightly during the monitored period. Shannon index was not significantly influenced by management. The number of species decreased in all treatments except of fertilization, where the the slight increase was observed.

A shift in species composition took place in the *Carex* community over the study period. 19 species occurred in total, but 7 of them were found only in one square in one season. The only species found in all squares within all years apart from *Carex brizoides* was *Bistorta major*. Hadincová et al. (1997) claims, that *Bistorta* becomes an expansive species in higher altitudes of Krkonoše Mts. thanks to the abandonment of traditional management, and gradually it completely breaks up the original connected sward. Their results showed the quick decrease of *Bistorta* following annual cutting, more rapid in fertilized plots compared with mown plots. *Bistorta major* did not respond to management changes in species-rich grasslands, where it was only an additional species. However, this was not proved in my experiment, because *Bistorta* slightly increased in cover in all treatments during whole period. In the first season, the average cover was very low (0.1% in all plots). It was about 1% in the second year and more than 2% at the end of the experiment. The cover increased only slightly in the control, whereas it was almost 3% in remaining treatments.

The CCA ordinations imply that *Carex brizoides*, *Deschampsia cespitosa* and *Scirpus sylvatiscus* were positively correlated with both the cover E1 and biomass at the same time. *Carex* forms these dense stands with high cover. Although *Scirpus* and *Deschampsia* had lower cover and biomass values, they were also confined to dense stands with high cover. *Deschampsia* occurred only in the *Carex brizoides* community, *Scirpus* occurred infrequently also in the *Molinion*.

Most of the authors have observed a decrease in biomass of *Carex brizoides* following cutting, but in my 3-year experiment, the total biomass increased in all treatments during this period. Only in the second season did the biomass decrease in the cut squares. This acknowledges how problematic a species *Carex brizoides* is and how difficult it is to eliminate on former farmlands. Cutting was found to bring about a smaller increase in the biomass in the last season, whereas other treatments increased it considerably.

Calamagrostis villosa community

The *Calamagrostis villosa* community had the lowest above-ground biomass and cover of all meadows. Independently of treatments, the total biomass slightly decreased in the second year and increased in the last season, whereas the cover slightly decreased in both the large and the small squares during the monitored period. The *C. villosa* stand had very low values of the Shannon index (but not so extremely as *Carex brizoides* community) and the diversity slightly increased during the tracking period. The diversity was not significantly influenced by the management treatment. The number of species ranged between 2 and 7 and it increased in all treatments.

Pyšek (1990) found out higher species richness in the moist meadow invaded by *Calamagrostis villosa* than in the deforested site completely dominated by this species. He also proved (Pyšek 1994), that both tiller height and thickness of the litter layer decreased with soil moisture. My meadow community with dominant *C. villosa* is characterized by dry place with thick litter layer, which is a consistent feature. In the experiment of Pyšek (1994) the species diversity was not related to the moisture level but it was shown to be correlated positively with available light and negatively with soil acidity.

In my experiment, treatments did not significantly affect the biomass. Nevertheless, the average biomass of *Calamagrostis* community over the three seasons was lower in all treatments (especially the cut squares) compared to the control. The biomass was minutely lower on average in fertilized plots compared to the control during three years. My results correspond considerably to those obtained by Hejtman et al. (2009). Biomass production of *C. villosa* was relatively stable in control and fertilized plots, but yielded a highly negative response to regular cutting management.

Morávková-Lipnická (1991) described the reduction of both total above-ground plant mass and root mass following mowing, but only in the year of treatment. This result, however, strongly depends on the timing of perturbation during the growing period (Morávková-Lipnická 1991). Pyšek (1993) revealed that plant mass production is higher in deforested sites than in forest habitats, i.e. it increases with increasing available light.

The average cover of the *C. villosa* community was the same in the cut small squares as in the control during three seasons, whereas the cover in the cut large squares decreased by

10%. The reduction of cover following cutting was also observed by Hejcman et al. (2009). Cutting reduced the cover, biomass, sward height and tiller density of *C. villosa* during 6 years of the experiment.

Hejcman et al. (2009) observed also high year-to-year variability in plant species composition in *C. villosa* dominated communities. Plant species composition was significantly influenced by cutting but not by fertilization. Pyšek (1994) also observed that nitrogen levels had no significant effect on the community composition.

Pyšek (1990) observed that the number and total biomass of the other species in the *C. villosa* community decreased with the biomass of *C. villosa*. Both the number of the other species and their total biomass are more closely related to the amount of *C. villosa* litter than to its living biomass. In my experiment 10 plant species occurred within all years and all treatments, but only *Calamagrostis villosa* and *Carex brizoides* were consistently present, whereas the occurence of other species was variable.

The average value of the biomass in my limed plots was minutely lower than in the control. In contrast, Morávková-Lipnická (1991) noticed that liming encouraged *C. villosa* with an increase in above-ground living plant mass in the two years following its application.

Lysimachio vulgaris-Filipenduletum ulmariae

The *Filipendula ulmaria* community had lower average cover and biomass independently of treatments than both *Molinion* and *Carex* community and higher than *Calamagrostis* community. The total biomass and the cover in the large squares independently of treatments increased significantly during the tracking period (two years). The stand had the highest values of the Shannon index of all meadow types and an increase was recorded. However, *H‘* was not significantly influenced by management. The number of species ranged between 17 and 22 and it increased in cut plots and decreased in both limed and untreated plots. Fertilization did not change the number of species.

Rychnovská et al. (1985) claimed, that the competitiveness of slow-growing species which flower and ripen in late summer gradually decreases (e.g. *Filipendula ulmaria*) following cutting, resulting in the decrease of their vitality and in their slow withdrawal from the stand. Blažková & Řehořek (1999) assume, that the restoration of cutting of wet *Filipendula* meadows should lead to the development od species-rich communities. Otherwise, Hakrová & Wotavová (2004) did not record the considerable increase in species richness in cut squares, but the proportion of present species changed - both the biomass and the cover of initially predominant *Filipendula ulmaria* decreased considerably.

Simultaneously both the cover and biomass of sedges (*Carex* sp.) increased, without the change of the total biomass.

There was also an increase in the species richness in cut squares observed in my experiment (from 19 to 22). The total cover in cut squares increased only slightly during the monitored period, whereas an increase of the total biomass was little faster. Nevertheless, the decrease of both cover and biomass of single *Filipendula* following cutting was not observed. On the contrary, *Filipendula* increased the average cover from 15% to 19% in the large squares and it increased the biomass at the same time. *Carex nigra* also increased similarly both the cover and biomass. In contrast, *Lysimachia vulgaris* decreased both variables under a mowing regime. These results do not support the theory that the cover of *Filipendula* decreases after cutting. However, my results followed only 2-year experiment and this fact must be taken to consideration.

The impact of fertilization on the biomass of *Filipendula* is various. van der Hoek et al. (2004) studied the biomass production and species composition changes in a species-rich fen meadow in central Netherlands. At little drained margin site of Nature reserve (species such as *Holcus lanatus*, *Plantago lanceolata* and *Filipendula ulmaria* are abundant), neither the N- nor the P-supply affected total biomass production in the first and subsequent years. In contrast, Wilson et al. (1995) observed the increase of biomass in artificial chalk grasslands following nitrogen application. The increase in dry weight of the sward was a result of a greater above-ground yield of certain forbs, particularly *Filipendula*, *Scabiosa* and *Thymus*. Pauli et al. (2002) also proved the increase of biomass following fertilization. Their two-year field experiment was conducted in calcareous fen meadows of low productivity in north-eastern Switzerland, where the fertilizer (N and NPK) was applied. They recorded the above-ground biomass of habitat specialist *Succisa pratensis* and a generalist *Filipendula ulmaria*. The addition of both nitrogen and full fertilization with NPK increased aboveground community biomass. On the other hand, *Filipendula* increased the ratio between shoot and root biomass, grew taller and could thus keep pace with neighbouring plants (Pauli et al. 2002).

The biomass of my *Filipendula* community was not significantly affected by the treatment, but the significant impact of the interaction meadow type*treatment on the cover was proved. Both the cover of single *Filipendula* and the total cover in the small fertilized squares increased. In the large fertilized squares, the cover of the whole community increased moderatly (from 73% to 79%), whereas the cover of single *Filipendula* slightly decreased (from 35% to 31%).

In the experiment of Pauli et al. (2002) neither total species richness nor the number of specialist species was significantly affected after two years of nutrient application. However, the number of generalist species increased after the addition of NPK. Changes in the abundance of the four taxonomic-functional groups and of single species suggested that species composition and richness would change over longer periods of eutrophication (Pauli et al. 2002). The species number in fertilised plots of my *Filipendula* community also remained unchanged (20 species both in the second and the third year), thus it also should be monitored for a longer period for eventual changes.

As the CA analysis shows (Fig. 26), the species of *Filipendula* community prefer very similar conditions as species of the *Molinion*. CA ordination diagram displaying the relation of species independently of the treatments showed an obvious associativity of species found only in *Filipendula* community (*Lysimachia vulgaris*, *Equisetum fluviatile*, *Eriophorum angustifolium* and *Filipendula ulmaria*) and species found in *Filipendula* community and in *Molinion* at the same time (*Carex nigra*, *Agrostis stolonifera* and *Sanguisorba officinalis*). This group is clearly differentiated from the *Carex brizoides* stand as well as *Calamagrostis villosa* stand with its environmental requirements. This is supported by next results (Fig. 29, 30). The species of *Filipendula* community, negatively correlated both with the cover and biomass, prefer stands with lower cover and biomass in contrast to *C. brizoides*.

6. Conclusion

1) All the studied meadow types differed significantly in all the tested variables: the dry-matter of an above-ground mass, the plant cover in both the small and large squares, Shannon diversity index derived from both the biomass and the cover data. The biomass and the cover decrease in the following order: *Carex* community, *Molinion*, *Filipendula* community, *Calamagrostis* community. Whereas the biomass increased significantly during the monitored period independently of management (with very different intensity among phytocenoses), the cover either slightly decreased significantly (*Molinion* and *Calamagrostis* community) or increased (*Carex* community in the third year and *Filipendula* community). Shannon index decreases in the following order: *Filipendula* community, *Molinion*, *Calamagrostis* community, *Carex* community. The diversity mostly increased during the three seasons. The most rapid increase in *Molinion* and *Filipendula* community was recorded.

2) The impact of the treatments on the cover in the small squares was proved on an indicative level of significance independently of meadow type and year. Liming and fertilizing slightly increased the average cover during the studied period and cutting slightly decreased it. The biomass and cover changes following all treatments are very heterogeneous in all meadows. Although the impact of the interaction meadow type*treatment on the biomass was not significant, the biomass increased during the three years in all meadows and treatments except of the limed and fertilized plots of *Calamagrostis* community (slight decresae was observed in this case).

The changes in the cover showed significant correlation with the types of treatment used. The response of the cover to the management treatments varied between the meadow types. Limed and fertilized plots of the *Molinion* and *Calamagrostis* communities had higher values of the cover compared to the control, but the cover of the *C. brizoides* community was not affected. In contrast, cut plots of the *Molinion* and *Filipendula* community had slightly reduced cover. The cover of *Filipendula* community increased slightly after fertilization and descreased following liming.

The changes in Shannon diversity were not significant after imposing different treatments, but the index increased relatively in all meadows. The species richness increased in *Molinion* and in *Calamagrostis* community, decreased in *Carex* community and remained more or less unchanged in *Filipendula* community.

3) The CA ordination diagrams showed the strict distribution of the species (independently of the treatments) into three groups: *Carex brizoides*, *Calamagrostis villosa* (eventually with tree seedlings) and the group of several species belonging to the *Molinion* and *Filipendula* community, which showed an obvious associativity. These species, grouped together in ordination, indicate the same conditions and preferences for growth, different from environmental requirements of *C. brizoides* and *C. villosa* communities. In addition, CCA ordination diagrams showed that some species of *Filipendula* community as well as species of the *Molinion* were negatively correlated both with the cover and biomass, which was preferred by *C. brizoides*. The total biomass was positively correlated with the plant cover. The occurrence of *Calamagrostis villosa*, prefering more likely thinner stands, was negatively correlated with both the cover and the biomass. All managements and the most of species were positively correlated with the year and were not correlated with both the biomass and plant cover. The cover of mosses was either negatively correlated or were not correlated with the cover of plants.

All four meadows are more or less species-poor communities at an advanced successional stage and have been affected by abandonment. Although the impact of the treatments on the Shannon index was not significant, new species occurred and the diversity increased in all meadows and treatments, especially in the *Molinion*. The cover of *Calamagrostis villosa* community showed a decreasing trend after cutting. This result could be considered as the next confirmation that cutting is an appropriate management tool for *C. villosa* control. Despite most of the other observations, significant changes in *Carex brizoides* community biomass were not proved. The decrease of biomass occurred only in cut plots in the second season.

Longer term monitoring of the communities is needed to study their responses to different management treatments. These would provide more consistent results and thereby more reliable guidance as to the best methods for the control of *Carex brizoides* and *Calamagrostis villosa*.

7. Summary in Czech

Příhraniční oblast Šumavy na pravém břehu Lipna v severní části povodí Horského potoka (povodí Dunaje, 940-830 m.n.m.) je od druhé světové války málo zemědělsky využívána. Opuštění luk způsobilo degradaci širokého spektra rostlinných společenstev. Tříletý experiment proběhl na čtyřech typech lučních porostů v pokročilé sukcesi charakteristických pro danou oblast: svaz *Molinion caeruleae*, degradovaný *Calthion* s dominující *Carex brizoides*, louky s dominující expanzní *Calamagrostis villosa* a asociace *Lysimachio-Filipenduletum*. Cílem práce bylo popsat vliv kosení, vápnění a hnojení na pokryvnost rostlin, nadzemní biomasu a diverzitu luk a popsat distribuce a preference lučních druhů v závislosti na těchto zásazích. Práce byla založena na fytocenologickém snímkování trvalých ploch v designu latinských čtverců a na statistickém zpracování v programech STATISTICA a CANOCO. Analýzy prokázaly průkazné rozdíly mezi všemi loukami ve všech parametrech. Vápnění a hnojení nezávisle na typu louky a roku slabě zvýšilo celkovou pokryvnost luk, zatímco kosení ji snížilo. Změny v biomase a pokryvnostech vlivem zásahů se u luk velmi liší. Vliv zásahů na biomasu nebyl statisticky průkazný, na rozdíl od pokryvnosti rostlin. Obecně se množství biomasy u luk zvýšilo, zatímco pokryvnosti byly více variabilní. Vyšší hodnoty pokryvnosti v porovnání s kontrolou byly zjištěny u vápněných a hnojených čtverců louky svazu *Molinion* a porostu s dominantní *C. villosa*, zatímco společenstvo s *Carex brizoides* nebylo ovlivněno. Naopak snížení pokryvnosti bylo pozorováno u kosených čtverců louky svazu *Molinion* a společenstva s *Filipendula ulmaria*. Pokryvnost této louky byla zvýšena v hnojených a snížena ve vápněných čtvercích. Změny v diverzitě (Shannon-Wienerův index) vlivem zásahů nebyly průkazné, ale index v průběhu let narůstal u všech luk. CA a CCA analýzy ukázaly těsnou asociativitu a podobné nároky na prostředí druhů luk s *Molinia caerulea* a *Filipendula ulmaria*. Některé z těchto druhů, stejně jako *C. villosa*, byly negativně korelovány s pokryvností a množstvím biomasy na rozdíl od druhu *Carex brizoides*. Kosení a hnojení měly jen malý vliv, ale i ten byl průkazný. Všechny managementy a většina druhů nebyly korelovány s biomasou ani pokryvností a zároveň byly pozitivně korelovány s rokem.

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9. Appendix

Fig. 31. The establishment of the permanent monitoring plot in association *Junco effusi-Molinietum caeruleae* Tüxen 1954.

Fig. 32. The stand of the *Molinion* in summer.

Fig. 33. The establishment of the permanent monitoring plot in the association *Scirpo sylvatici-Caricetum brizoidis* Kučera et al. 1994.

Fig. 34. The stand of *Carex brizoides* in summer.

Fig. 35. The establishment of the permanent monitoring plot in the association *Crepido conyzifoliae-Calamagrostietum villosae* (Zlatník 1925) Jeník 1961.

Fig. 36. The stand of *Calamagrostis villosa* in summer.

Fig. 37. The establishment of the permanent monitoring plot in the association *Lysimachio vulgaris-Filipenduletum ulmariae* Balátová-Tuláčková 1978.

Fig. 38. The stand of *Filipendula ulmaria* in summer.

Fig. 39. The slurry application in the *Molinion*.

Fig. 40. The limestone application in the *Molinion*.

Fig. 41. The delicacy.

Fig. 42. Fertilized plot in *Calamagrostis villosa* community in spring.

Fig. 43. The detail of the sparse stand of *Calamagrostis villosa* in summer.

Fig. 44. The juvenile stand of *Filipendula ulmaria* in the spring.

Fig. 45. The *Molinion* after winter.

Fig. 46. The typical landscape of the alluvium of the brook Horský potok.