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SEMINÁŘ STUDIUM LESNÍCH EKOSYSTÉMŮ A JEJICH POŠKOZENÍ

4. ročník – Metody zpracování dat užívané při studiu lesů, v lesnictví a příbuzných oborech

Seminář se uskutečnil 4. března 1998 v prostorách Lesnické fakulty České zemědělské univerzity v Praze-Suchbale, LF byla rovněž spolupořadatelem tohoto semináře. Následující články mají jednoho společného jmenovatele, kterým je jejich prezentace v souvislosti s tímto seminářem.

V průběhu jednání se opět potvrdilo několik skutečností – zvláště politováníhodný fakt, že zpracování dat je zanedbávanou částí práce mezi značným podílem lesnických výzkumníků, ekologů a biologů i přes v současnosti probíhající nezanedbatelný pokrok v této oblasti. Zpracování často končí na úrovni výpočtu průměrů, v lepším případě u použití analýzy variance. Zpracování dat bývá také zaměňováno s vizualizací dat, a to i v případě, že jsou uplatněny takové progresivní nástroje jako geografické informační systémy (GIS). Stále častěji se objevují nástroje spojující funkce sběru dat a jejich bezprostřední vizualizace. Jako příklad je možné uvést systém Field-Map firmy IFER Jílové u Prahy pro měření v terénu. Stále více potřebná je nutnost formulovat jasné otázky, na které se v průběhu studia budeme snažit hledat odpověď. To musí umožnit sběr vhodných dat, která budou použitelná pro účely matematicko-statistického testování. Tyto otázky musí být přímým podkladem pro plánování experimentů.

Následující články ukazují rozdílnou možnost přístupu ke zpracování dat. V prvním se autoři (V a c e k , M a t ě j k a) na základě originální aplikace a případné úpravy matematicko-statistických postupů snaží vytěžit maximální objem informace z poměrně obsáhlého datového souboru. U dalších článků je hlavním objektem zájmu lesní půda a půdní procesy na lokalitách pod vli-

vem imisí. Jejich zvlášť významným tématem je dynamika dusíku jako prvku, o jehož důležitosti se v současnosti stále častěji mluví v souvislosti s narušováním výživy lesních porostů pod vlivem imisí. Zatímco se depozice sloučenin síry postupně snižují, emise oxidů dusíku se stále nedaří omezit. Poukazuje se i na funkci náhradních travních porostů na imisních holinách, o jejichž významu nejen pro obnovu lesa nelze pochybovat. Poslední z těchto článků (P. H o l u b: *Hodnocení příjmu dusíku a jeho retranslokace travinami na odlesněných plochách*) bude z technických důvodů uveřejněn v následujícím čísle časopisu.

Pozornost si zaslouží několik prezentovaných příspěvků, které však nebyly zpracovány pro publikaci zde. Z nich je možné zmínit zejména následující: V. P u š (Lesnická fakulta ČZU): *Indexy pro popis prostorové struktury ekosystémů* (byl proveden rozbor používaných indexů, možnosti jejich použití i s příklady a uvedena problematika interpretace výsledků); T. V r š k a (Správa Národního parku Podyjí): *Sledování dynamiky vývoje pralesovitých rezervací v ČR – metodika a aplikace výsledků* (byly prezentovány výsledky sledování prostorové struktury stromového patra velkoplošných komplexů přirozených lesů za využití GIS); M. Č e r n ý (Ústav pro výzkum lesních ekosystémů): *Využití geostatistických metod při zpracování výsledků pozemního monitoringu zdravotního stavu lesů* (v referátu zazněla i řada obecnějších postřehů z použití GIS v lesnickém a lesnicko-ekologickém výzkumu a v monitoringu); J. S i m o n a kolektiv (Fakulta lesnická a dřevařská MZLU v Brně): *Analýza možností zjišťování stavu lesa pomocí počítačové analýzy obrazu*.

Ing. Karel Matějka, CSc.,
organizátor semináře

STUDY OF FOREST ECOSYSTEMS AND THEIR DAMAGE – A SEMINAR

4th Year – Methods of data processing used to study forests, in forest management and related disciplines

The seminar was held at the Faculty of Forestry of Czech University of Agriculture in Prague-Suchdol on 4th March 1998, the faculty was one of the seminar organizers. The following papers have one point in common: presentation on the occasion of this seminar.

Some facts were confirmed during sessions – unfortunately, data processing is a neglected field of work among many forest researchers, ecologists and biologists even though great advances have been made in this field. Data processing often terminates in calculations of means or in analysis of variance if feasible. Data processing can also be confused with data visualization, even in case such progressive tools as geographic information systems (GIS) are applied. Tools combining data-collection feasibility with immediate data visualization appear still frequently. An example is the system Field-Map developed by the firm IFER Jílové u Prahy destined for field measurements. It is more and more urgent to ask clear questions and seek answers to them through relevant studies. Collection of appropriate data that will be usable for mathematico-statistical testing will facilitate to cope with this task. The questions should become a framework for experiment planning.

The following papers document different approaches to data processing. The authors of the first paper (V a c e k, M a t ě j k a) try to gain maximum information volume from a relatively extensive data set on the basis of original application and potential adjustment of mathematico-statistical procedures. Forest soil and soil processes at localities with air pollution impacts were the main goals of research in other papers. Their very important subject was dynamics of nitrogen as an ele-

ment currently discussed more and more frequently with respect to disorders of forest stand nutrition that are caused by air pollution impacts. While sulphur depositions were decreasing, emissions of nitrogen dioxides failed to be limited. Functions of substitute grass covers on clearcut areas due to air pollution are also described; their importance for forest regeneration cannot be made doubtful. The last of this paper set (P. H o l u b: *The evaluation of nitrogen uptake and retranslocation by grasses of deforested areas*) will be published in the next issue regarding the technical feasibility.

Several papers presented to seminar sessions should be mentioned even though they are not published in this journal. Let's mention these papers: V. P u š (Faculty of Forestry of the Czech University of Agriculture): *Indexes for Description of Spatial Ecosystem Structure* (analysis of indexes in use, description of possibilities of their use and respective examples, outline of result interpretation); T. V r š k a (Administration of Podyjí National Park): *Study of Development of Virgin Forest Preserves in the CR – Methods and Application of Results* (results of studies of the spatial structure of tree layer in large-area complexes of natural forests are presented as obtained by help of GIS); M. Č e r n ý (Institute of Forest Ecosystem Research): *Use of Geostatistical Methods to Data Processing from Ground Monitoring of Forest Health* (the report contained some general evidence of GIS use for forest and forest-environmental research and for monitoring); J. S i m o n et al. (Faculty of Forestry and Wood Technology of Mendel University of Agriculture and Forestry at Brno): *Analysis of the Ways of Forest State Identification by Computer-based Image Analysis*.

Ing. Karel Matějka, CSc.,
Seminar organizer

THE STATE OF FOREST STANDS ON PERMANENT RESEARCH PLOTS IN THE KRKONOŠE MTS. IN YEARS 1976–1997

STAV LESNÍCH POROSTŮ NA VÝZKUMNÝCH PLOCHÁCH V KRKONOŠÍCH V LETECH 1976–1997

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ABSTRACT: Health state of maturing and mature Norway spruce (*Picea abies* L.), mixed spruce-beech and beech (*Fagus sylvatica* L.) stands was studied on 32 permanent plots (5th–8th altitudinal vegetation belt, pollution-damage zone A–C) in the Krkonoše Mts. during 1976–1997. Analysis of air pollution and ecological factors was based on reactions of all single trees in the plot stand. Damage (indicated by defoliation) of a tree was classified every year according to Tesař, Temmlová (1971) scale for spruce and to Vacek, Jurásek (1985) scale for beech respectively. Changes in the structure of ground (herb and moss) vegetation were studied using classification (TWINSPAN) and ordination (DCA) methods. Results were related to the defoliation process of trees. Three periods of tree damage were distinguished. The first period (1976–1980) was characteristic by slow defoliation. The second one (1981–1988) was the main period of tree damage (increase of defoliation as well as of share of dead trees in the stand, total destruction of several stands was observed). In the last period since 1989, stabilisation of the defoliation increase has been observed. Highest resistance was documented in beech stands, lower in mixed spruce-beech stands, and the lowest in spruce stands. Tree damage was positively correlated with the altitude. Air pollution stress was strengthened by insect pest occurrence.

air pollution; classification; defoliation; *Fagus sylvatica* L.; ground vegetation; health state; mixed stands; ordination; *Picea abies* (L.) Karst.; statistics; transition matrices

ABSTRAKT: Zdravotní stav a vegetační dynamika dospívajících a dospělých smrkových, smrkobukových a bukových porostů byly studovány na 32 trvalých výzkumných plochách (5.–8. LVS, pásmo ohrožení A–C) v Krkonoších v letech 1976–1997. Analýza imisně ekologických vlivů vycházela z dendroekologických reakcí jednotlivých stromů v rámci textury porostů. Poškození bylo klasifikováno každoročně podle olistění. Olistění smrku bylo hodnoceno podle Tesaře, Temmlové (1971) a buku podle Vacka, Jurásk (1985) se zvláštním zřetelem na cenotické postavení. Změny přízemní vegetace (bylinné a mechové patro) byly sledovány při užití klasifikačních (TWINSPAN) a ordinačních (DCA) metod. Výsledky byly hodnoceny v souvislosti s probíhajícím procesem defoliace. Podle dynamiky olistění byla v průběhu sledovaných let vylišena tři charakteristická období. V prvním období (léta 1976–1980) došlo k mírnému nárůstu defoliace, ve druhém období (léta 1981–1988) došlo k výraznému poškození stromů a porostů (k nárůstu defoliace stejně jako ke zvětšení podílu mrtvých stromů v porostu, byla pozorována úplná destrukce několika porostů). V posledním období od roku 1989 byla pozorována stabilizace zdravotního stavu. Největší toleranci jevíly bukové porosty, jen mírně nižší smrkobukové porosty a nejmenší smrkové porosty. Poškození porostů většinou narůstalo se stoupající nadmořskou výškou. Imisní stres byl provázen zvýšeným výskytem hmyzích škůdců.

imise; klasifikace; defoliace; *Fagus sylvatica* L.; přízemní vegetace; zdravotní stav; smíšené porosty; ordinační; *Picea abies* (L.) Karst.; statistika; přechodové matice

INTRODUCTION

Forests in the Krkonoše National Park have been influenced by air-pollution stresses for almost four decades as it can be supposed from the origin, position and size of pollutant sources. A pollutant stress in the western part of these mountains has probably existed since 1972 as a result of the operation of power stations in the environs of eastern German and south-western Polish frontiers. These power stations emitted ca. 900 kt per annum SO₂ in the late eighties (Balcar et al., 1994). An increased air-pollution stress is supposed to have existed in the eastern Krkonoše Mts. since 1959 when a power station at Poříčí near Trutnov (EPO II) was launched into operation. Other pollutant sources have impacts on the Krkonoše nature as it is evident from calculations of the share of industrial areas and the particular operations in sulphur deposition in the Krkonoše National Park area (Kurfürst et al., 1991): Turow 18.5%, Počerady 3.6%, Lužice 21.8%, Tušimice I-II 6.4%, Belchatow 3.6%, Prunéřov I-II 7.1%, Walbrzych 1.1%, Komořany 1.5%, Trutnov-Poříčí 1.0%, Litvínov CHEZA 1.8%, Opatovice 0.9%, Leipzig, Halle, Dresden 6.9%, Chvalětice 4.4%, Mělník I-III 4.6%, other sources 16.8%.

The first greater damage to Krkonoše spruce stands appeared after climatic shocks in March 1977 and at the beginning of 1979, and also in connection with a larch bud moth outbreak in 1977-1981 (Vacek, Vašina, 1991). Pollutant effects, accompanied by other negative factors (pathogenic organisms and extreme climatic episodes) resulted in extensive forest decline in the next years (Vacek et al., 1994, 1996). A total of ca. 7000 ha forest stands were felled in the Krkonoše Mts. by so called salvage cuttings (Vacek et al., 1994).

Even though SO₂ pollution has been decreasing and is expected to decrease in future, forest decline will continue in the next 10-15 years although the level of stand damage has stagnated since 1988-1989 (Vacek, 1995). Soil conditions were substantially altered (soil acidification) and ecological limits for the existence of the present forest stands were exceeded (Vacek, Podrázský, 1994, 1995; Podrázský, 1996). Therefore research on the dynamics of forest stand damage was undertaken. The knowledge of structural processes in forests induced by air-pollution stresses (cf. Vacek et al., 1987, 1988; Vacek, 1990) is necessary to define specific guidelines of their management (Vacek, 1984a).

The present paper is a follow-up of the preceding one (Matějka et al., 1998), in which trends of spruce stand damage in the territory concerned were described as determined by terrestrial observations and evaluated on the base of satellite imagery. Mathematico-statistical processing of data is also the core of this paper. Assessment of transition matrices describing the probability of a change in the evaluation of single tree defoliation from one to another defoliation class was the basic method. A similar procedure was used in preceding pa-

pers (Vacek, 1987a, 1989a; Vacek, Lepš, 1987, 1995, 1996).

Damage of the tree layer in forest stands should be parallelly understood as an element of the change in forest ecosystem influenced by air-pollution stress. Therefore the basic knowledge of the structure of primary producer cenoses which is indicated by vegetation assessment using plant coenological relevés was applied to evaluate defoliation.

MATERIAL AND METHODS

EXPERIMENTAL PLOTS

Thirty-two permanent research plots (PRP) established in 1976 and 1980 to study the structure and development of mountain forests were used to examine forest stand health. Research plots in maturing and mature spruce, spruce-beech and beech stands were selected in order to represent the growth conditions of the particular ecosystems (Vacek, 1986a). Tab. I shows basic data on research plots. PRPs are mostly 0.25 ha in size, only the area of PRP 7 is 1.0 ha and PRP 6 is 0.5 ha in size. All PRPs comply with the requirements for long-term research plots.

VEGETATION CONDITIONS

Plant coenological relevés were obtained by routine procedures in 1976 and 1980 while eleven-grade Domin-Hadač combined scale for abundance and dominance was used. Relevés were taken regularly in five-year intervals in the course of plot observation (in 1980, 1985, 1990, 1995, on PRP 11-15 also in 1976); the respective periods are designated by Nos. 1-4 and 5, respectively). The values of the scale were replaced by the average values of coverage for the separate degrees. Relevés were evaluated by these methods:

1. Evaluation of species diversity indexes in the period of observation: general species diversity (Shannon-Wiener index of species diversity 'H'), total number of species (S) and equitability (e).
2. General developmental trends on groups of plots (spruce, mixed and beech stands, spruce plots PRP 11-15 were evaluated separately due to their different nature) were evaluated from "average" relevés in which the presentation of each species was calculated as average coverage on all plots of the group.
3. Classification of all phytocenological relevés by TWINSPLAN procedure (Hill, 1979a).
4. Hierarchical agglomerative classification of relevés, average-linkage method; Euclidean distance was used as the dissimilarity measure.
5. Relevé ordination by DCA method (Hill, 1979b); relevés from all plots were used. The most different plots were excluded during the second run of the procedure to avoid inappropriate deformation of ordination space.

I. Summary of the basic properties of permanent research plots (according to working plans, state by 1981)

	Plot name	Forest establishment	Stand	Woody species (sm - Norway spruce, bk - beech)	Age (years)	Mean height (m)	Mean diameter (cm)	Yield class	Growing stock (m ³ ·ha ⁻¹)	Altitude (m)	Orientation (S - north, V - east, J - south, Z - west)	Inclination (°)	Forest type	Forest stand group	Pollution-damage zone	Rock	Soil type
1+	U tunelu	Harrachov	221 A8	sm bk	105 105	23 20	25 20	5 6	390 40	730	JZ	26	6K5	01	C	biotite granite	cambisol
2	Vilémov	Harrachov	415 B16	sm bk	145 145	25 22	30 28	6 6	210 140	600	JZ	22	5Y1	01	C	mica schist phyllite	ranker
3++	U Lubošské bystřiny	Harrachov	514 A1	sm	216	22	32	9	185	1150	JZ	22	8Z1	02	A	granite	podzol
4	Pod Voseckou boudou	Harrachov	511 A17	sm	196	18	30	9	250	1180	JZ	12	8T2	02	B	biotite granite	organosol, gleysol
5	Pod Lysou horou	Harrachov	508 B17	sm	216	22	32	9	185	1130	S	17	7T1	02	B	biotite granite	organosol, gleysol
6	Bažinky 2	Harrachov	311 A17	sm bk	205 205	28 20	48 30	6 8	260 280	1060	V	22	8K3	72	C	gneiss, phyllite	cambisol
7	Bažinky 1	Harrachov	311 A17	sm bk	205 205	33 26	58 38	5 6	110 420	940	V	24	6S2 (6H, V2)	54	C	gneiss	cambisol
8	Nad Benzínou 2	Harrachov	306 C14	sm bk	131 131	24 30	36 20	6 8	20 240	1190	JZ	24	6A6	02	B	biotite granite	cambisol
9	Nad Benzínou 1	Harrachov	306 C17	sm bk	141 141	24 20	36 30	6 8	20 240	1170	JZ	17	6A6	02	B	biotite granite	cambisol
10	Pod Vysokým kolem	Vrchlabí	103 E17	sm	185	14	30	9	120	1240	J	16	8K2 (8R1)	02	B	biotite granite	podzol, gleysol
11	Strmá stráň A	Vrchlabí	117 B17	sm	188	18	30	9	170	1220	SV	29	8Z4	02	A	biotite granite	podzol
12	Strmá stráň B	Vrchlabí	117 B17	sm	188	18	30	9	170	1170	SV	26	8Z4	02	A	biotite granite	podzol
13	Strmá stráň C	Vrchlabí	117 C17	sm	186	26	30	9	350	1120	SV	23	8N1	01	B	biotite granite	podzol
14	Strmá stráň D	Vrchlabí	117 C17	sm	186	26	30	9	350	1050	SV	24	8N1	01	B	biotite granite	podzol
15	Strmá stráň E	Vrchlabí	117 C17	sm	186	26	30	9	350	990	SV	22	8N1	01	B	biotite granite	podzol
16	Pod Martinovkou	Vrchlabí	105 D16	sm	145	18	27	9	250	1170	JV	16	8K2	02	B	biotite granite	podzol
17++	U Bílého Labe	Vrchlabí	219 A14	sm	122	21	27	9	320	1070	SV	29	7N1 (8N1)	01	A	biotite granite	podzol
18	U Certovy strouhy	Vrchlabí	213 A17	sm	169	19	30	9	260	1200	JZ	23	8N1	01	A	granite	podzol
19	U Klinové boudy	Vrchlabí	310 A16	sm	141	19	20	8	260	1170	JV	22	8K4 (8Z3)	02	B	mica schist, phyllite	podzol
20	Pod Liščí horou	Vrchlabí	237 A17	sm	144	17	34	9	150	1260	JZ	19	8Z4	02	A	mica schist, phyllite	podzol
21	Modrý důl	H. Maršov	233 A13	sm	107	13	28	9	160	1230	J	21	8Z4	02	B	mica schist, phyllite	podzol
22	Obří důl	H. Maršov	234 B16	sm	122	15	30	9	170	1160	V	32	8Z4 (9K)	02	B	mica schist, phyllite	podzol
23	Václavák	H. Maršov	101 B17	sm	160	13	29	9	100	1190	SV	4	8R1	02	B	orthogneiss	organosol
24	Střední hora	H. Maršov	330 D17	sm	164	21	28	8	360	1250	JV	20	8Z3	02	B	mica schist, phyllite	podzol
25	Pod Koulí	H. Maršov	331 A17	sm	165	21	28	7	365	1140	SV	28	8K9	01	B	mica schist, phyllite	podzol
26++	Lysečinský hřeben	H. Maršov	303 D1	sm	142	14	18	9	248	1170	Z	3	8Z3	02	A	orthogneiss	podzol
27	U bukového pralesa A	H. Maršov	525 C16	bk	147	10	21	9	190	1030	JZ	3	7K2 (7K3)	72	B	phyllite	cambisol
28	U bukového pralesa C	H. Maršov	536 A19	bk	121	21	33	7	280	940	JV	15	6K5	52	C	phyllite	cambisol
29	U bukového pralesa B	H. Maršov	536 A16	sm bk	142 142	25 21	54 41	5 8	59 202	950	JV	16	6S2	54	C	phyllite	cambisol
30	U hadí cesty D	H. Maršov	542 D16	bk	141	27	30	6	140	790	SV	24	6D4	55	C	metadiabase	cambisol
31	U hadí cesty F	H. Maršov	542 C14	sm bk	124 124	24 24	34 39	6 6	20 301	740	SV	23	6B1	54	C	metadiabase	cambisol
32	U hadí cesty E	H. Maršov	542 B13	sm bk	107 107	24 22	25 37	5 6	5 231	760	SV	35	6B9 (6A1)	54	C	metadiabase	cambisol

Note: +- partial cutting (plot 1 in 1991)

++ - clear-cutting and reforestation with spruce

6. The level of vegetation change during the observation period (beginning in 1980) was calculated as the sum of variances of the first DCA n -axes (one to four axes) (results of relevé ordination from all plots were used) – applying the expression

$$\sqrt{\sum_{i=1}^n \text{var}(DCA_{\cdot i})}$$

Random changes occurring on every plot during development and not corresponding to general trends of cenotic structure changes were eliminated in this way.

STAND HEALTH

The health of forest stands was evaluated with respect to foliage every year in the period (1976)–1981–1997. The method of Tesář, Temmllová (1971) was used for spruce evaluation while classification developed by Vacek, Jurásek (1985) was used for beech evaluation. Average foliage of stand is expressed as an arithmetical mean of foliage of all trees on a plot. Defoliation (complement of foliage to 100%) with special regard to the cenotic position and morphological type of tree-crown was estimated and recorded using six classes of defoliation that correspond to degrees of tree damage:

Defoliation	Defoliation interval (%)	Average defoliation (%)	Tree attribute
0	(0–10)	5	health
1	(10–30)	20	moderately damaged
2	(30–50)	40	medium damaged
3	(50–70)	60	greatly damaged
4	(70–100)	85	dying back
5	100	100	dead

Traits describing tree-crown health were also evaluated (damage caused by snow, frost, wood-decaying fungi and insects; cf. Vacek, 1992, 1993).

Degrees of damage were transformed to percent values of defoliation for the purposes of further calculations (average values for a defoliation class). Plot evaluation was based on trends of arithmetical means of defoliation (in per cent) of all living trees present on a plot at the same time (means for defoliation classes 0–4), standard deviations of defoliation and share of dead trees (totally defoliated trees). Each tree species was evaluated separately. General means of foliage of all trees including trees with total defoliation were calculated to evaluate the stand state in general.

Processes of defoliation changes and decline were studied on the basis of calculating transition matrices (cf. Matějka et al., 1998) for the particular defoliation classes, for two successive years in all cases. The matrices (6 x 6 in size) were classified by a hierarchical agglomerative average linkage method while Euclidean

distance of transition matrices was taken as the level of dissimilarity. With regard to the construction and basic characteristics of transition matrices, a maximum theoretically possible distance of two transition matrices is limited by the value

$$D_{\max} = p^{(0)} \sqrt{10}$$

where $p^{(0)}$ is theoretical maximum of any element of transition matrix (it equals 1; 100 when it is expressed in per cent).

It is to note that there is some probability (2α) a tree is classified to the nearest class not to its proper class (Vacek et al., 1996b). Let us consider the situation of two successive years when trees were classified to their “proper classes” in the first year, and with the above mentioned error in the next year while their health state did not change. Transition matrix P_{α} instead of the expected unit matrix I is produced in this case. Their Euclidean distance, $D(P_{\alpha}, I)$ is the value insignificant for the difference in two transition matrices evaluated:

$$D(P_{\alpha}, I) = \alpha \sqrt{22}$$

Classification of transition matrices enabled to determine their types corresponding to classes of the process of defoliation changes and tree dieback. It was important to ask about the sequence of these processes, whether it was possible to define some periods with typical processes (occurring on all plots under observation or on most of them), and about differences in their sequences on the separate plots. The first question is answered by simply describing the separate sequences. Frequencies of the separate types over the years were described to answer the second question. The following statistical analysis of identified sequences was made to answer the last question:

Let c_{py} be the type of transition matrix on plot p in year y (comparing years y with $y + 1$), $f_Y(c, y)$ is frequency of type c in year y (on all plots) and $f_p(c, p)$ is frequency of type c on plot p (in all years). Let's equate

$$F_Y(y) = \sum_c f_Y(c, y)$$

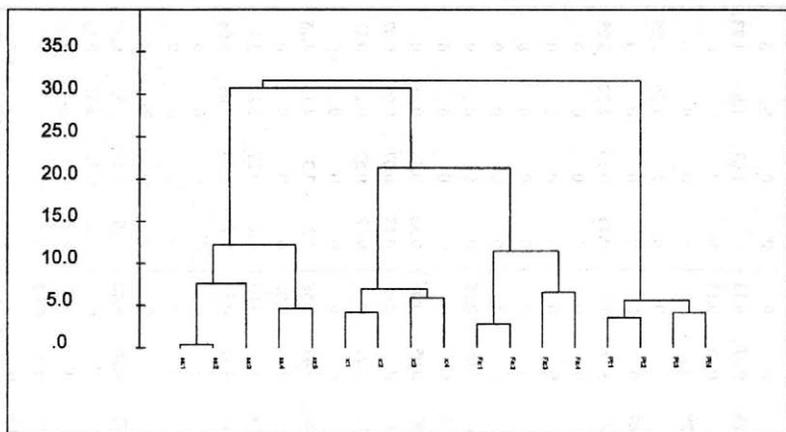
and

$$F_p(p) = \sum_c f_p(c, p)$$

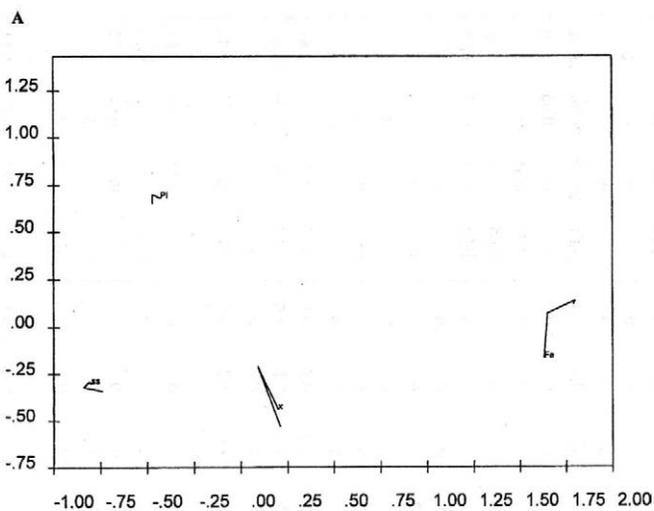
the sum is calculated through all types c . The probability $P(c_{py} = c)$ that the type c of transition matrix will occur on plot p in year y can be expressed by the equation

$$P(c_{py}) = 1 - \left(1 - \frac{f_Y(c_{py}, y)}{F_Y(y)} \right) \cdot \left(1 - \frac{f_p(c_{py}, p)}{F_p(p)} \right)$$

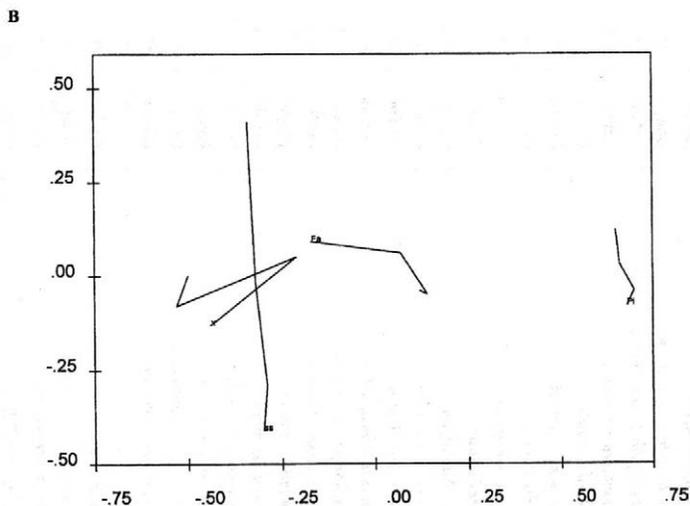
It is possible to test a null hypothesis that the sequence of types of transition matrices on plot p corresponds to the generally occurring sequence within the whole set of plots under observation, in contrast with a hypothesis that this sequence is different. The null



1. Classification of the mean relevés of the plot groups ss (permanent plots 11 to 15), Pi (the other spruce stands), x (mixed stands) and Fa (beech stands); mean relevés see Tab. II



2. DCA ordination of the mean relevés (see Tab. II) in groups of stands, axis 1 and 2 (A) or axis 2 and 3 (B). Trajectories of the mean relevés are drawn, respective group of plots is described by end of the trajectory



II. Mean relevés for the plot groups: ss (plots 11 to 15), Pi (the other plots with spruce), x (mixed stands) and Fa (plots with beech). Samplings were done in four or five periods

	Abbr- viation	ss 1	ss 2	ss 3	ss 4	ss 5	Pi 1	Pi 2	Pi 3	Pi 4	x 1	x 2	x 3	x 4	Fa 1	Fa 2	Fa 3	Fa 4
<i>Abies alba</i>	Abialb	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.03
<i>Acer pseudoplatanus</i>	Acapse	0.1	0.1	0.1	0.4	0.4	0	0	0	0	0.27	0.18	0	0	0.28	0.15	0.37	3.12
<i>Actea spicata</i>	Actspi	0	0	0	0	0	0	0	0	0	0	0	0	0	0.33	0.33	0.05	0.05
<i>Adenostyles alliariae</i>	Adeall	5.9	5.9	3.7	1.8	1.2	0	0	0	0	0	0	0	0	0	0	0	0
<i>Anthoxanthum odoratum</i>	Antodo	0	0	0	0	0	0.06	0.03	0	0	0	0	0	0	0	0	0	0
<i>Asperula odorata</i>	Aspodo	0	0	0	0	0	0	0	0	0	0.05	0.02	0	0	0.37	0.37	0.33	0.33
<i>Asplenium trichomanes</i>	Asptri	0	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0.02	0.02	0
<i>Athyrium distentifolium</i>	Athdis	18.9	19.2	25.7	29	31.3	1.35	1.49	1.77	1.79	0	0	0	0	0	0	0	0
<i>Athyrium filix-femina</i>	Athfil	0	0	0	0	0	0	0	0	0	0.73	0.28	0.07	0.13	1	1.07	1.53	1.93
<i>Betula pendula</i>	Betpen	0	0	0	0	0	0	0.15	0.15	0.14	0	0	0.05	0.13	0	0	0	0
<i>Blechnum spicant</i>	Biespi	0	0	0	0	0	0.13	0.05	0	0	0.37	0.28	0.1	0.03	0	0	0	0
<i>Brachypodium sylvaticum</i>	Brasyl	0	0	0	0	0	0	0	0	0	0	0	0	0	0.6	1	1.75	3.25
<i>Calamagrostis arundinacea</i>	Calaru	0	0	0	0	0	0	0	0	0	0.27	0.07	0	0	0	0	0	0
<i>Calamagrostis villosa</i>	Calvil	13.6	13.6	14.8	13.2	15.4	28.61	29.26	30.24	29.31	16.15	14.27	14.18	13.1	0.13	0.35	1.22	3.25
<i>Caluna vulgaris</i>	Calvul	0	0	0	0	0	0	0	0.06	0.14	0	0	0	0	0	0	0	0
<i>Campanula latifolia</i>	Camlat	0	0	0	0	0	0.04	0.01	0	0	0	0	0	0	0	0	0	0
<i>Carex limosa</i>	Carlum	0	0	0	0	0	0.05	0.05	0.05	0.05	0	0	0	0	0	0	0	0
<i>Carex pauciflora</i>	Carpau	0	0	0	0	0	0.02	0.02	0.13	0.13	0	0	0	0	0	0	0	0
<i>Carex pilosa</i>	Carpil	0	0	0	0	0	0	0	0	0	0	0.02	0.05	0.05	0	0	0	0
<i>Cicerbita alpina</i>	Cicalp	0.2	0.2	0	0	0	0	0	0	0	0.07	0.02	0	0	0	0	0	0
<i>Circaea intermedia</i>	Cirint	0	0	0	0	0	0	0	0	0	0	0.02	0.05	0.05	0.33	0.33	0.05	0
<i>Dentaria bulbifera</i>	Denbul	0	0	0	0	0	0	0	0	0	0	0	0	0.02	0.13	0.07	0.05	0.07
<i>Dentaria enneaphyllos</i>	Denenn	0	0	0	0	0	0	0	0	0	0.13	0.02	0.02	0	0.52	0.52	0.2	0.17
<i>Deschampsia caespitosa</i>	Descae	0.1	0.1	0.8	0.8	0.8	0.66	0.77	1.15	1.49	0.13	0	0	0	0	0	0	0
<i>Deschampsia flexuosa</i>	Desfle	9.8	9.9	8.3	7.8	5.6	23.17	20.09	20.58	20.6	6.4	4.48	5.42	3.58	2.3	2.3	4.15	4.15
<i>Digitalis purpurea</i>	Digpur	0	0	0	0	0	0	0	0	0	0	0	0.05	0.02	0	0	0	0
<i>Dryopteris dilatata</i>	Drydil	11.4	11.4	11.4	8.2	6.9	4.22	4.03	2.27	2.49	9.47	7.58	5	4.05	4.1	2.27	1.8	3.3
<i>Dryopteris filix-mas</i>	Dryfil	0	0	0	0	0	0	0	0	0	1.13	0.73	0.52	0.33	9.73	8.05	5.08	5.08
<i>Epilobium montanum</i>	Epimon	0	0	0	0	0	0	0.05	0.02	0.01	0	0	0	0	0	0	0	0
<i>Eriophorum angustifolium</i>	Eriang	0	0	0	0	0	0.43	0.13	0.13	0.13	0	0	0	0	0	0	0	0
<i>Eriophorum vaginatum</i>	Erivag	0	0	0	0	0	0.02	0.02	0.02	0.02	0	0	0	0	0	0	0	0
<i>Fagus sylvatica</i>	Fagsyl	0.1	0.1	0.1	0	0	0	0	0	0	6.77	6.77	4.18	9.42	1.07	1.15	1.52	6.08
<i>Fraxinus excelsior</i>	Fraexc	0	0	0	0	0	0	0	0	0	0	0	0	0	0.13	0.33	0.33	0.33
<i>Galeopsis speciosa</i>	Galspe	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	0	0	0
<i>Galium hercynicum</i>	Galher	0.4	0.4	0.6	0.9	0.8	0.25	0.22	0.11	0.11	0.05	0.05	0.05	0.02	0	0	0	0
<i>Galium saxatile</i>	Galsax	0	0	0	0	0	0	0	0	0	0.05	0.05	0	0	0	0	0	0

Continuation of Tab. II

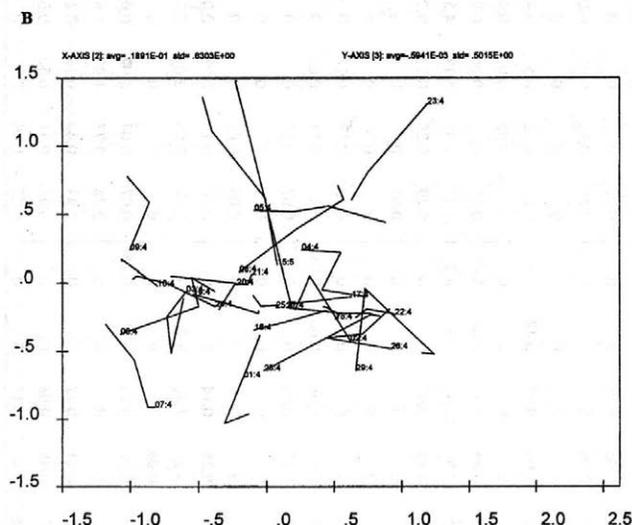
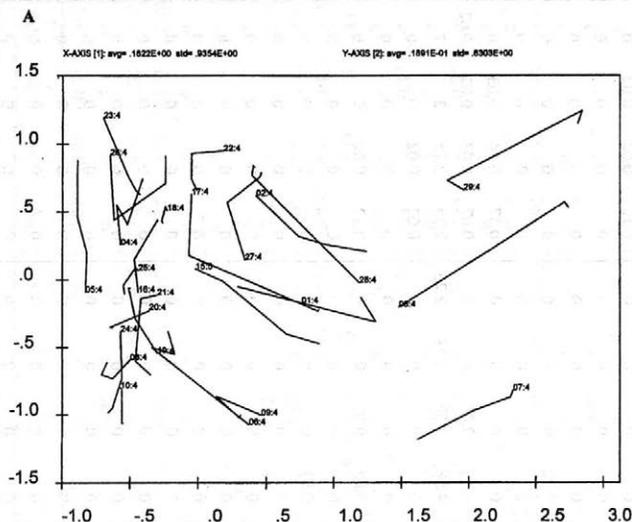
	Abbe- viation	ss 1	ss 2	ss 3	ss 4	ss 5	Pi 1	Pi 2	Pi 3	Pi 4	x 1	x 2	x 3	x 4	Fa 1	Fa 2	Fa 3	Fa 4
<i>Galium sylvaticum</i>	Galsyl	0	0	0	0	0	0	0	0	0	0	0	0	0	0.13	0.02	0	0
<i>Gentiana asclepiadea</i>	Genasc	0	0	0.1	0.8	0.8	2.79	2.87	2.37	3.41	1.13	1.33	1.33	2.08	0	0	0	0
<i>Gymnocarpium dryopteris</i>	Gymdry	0.2	0.2	0.2	0	0	0.01	0.01	0	0	4.72	4.52	1.73	1.53	1.4	1.6	1.05	1.05
<i>Hieracium laevigatum</i>	Hielae	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	0.02	0	0
<i>Hieracium sylvaticum</i>	Hiesyl	0	0	0	0	0	0.09	0.01	0	0	0.07	0.05	0	0	0.42	0.13	0.05	0.02
<i>Homogyne alpina</i>	Homalp	0.8	0.8	0.7	0.9	0.8	1.36	1.37	0.84	0.83	1.6	1.6	0.6	0.6	0.47	0.38	0.33	0.33
<i>Huperzia selago</i>	Hupsel	0.1	0.1	0.1	0	0	0.23	0.05	0	0	0	0	0	0	0	0	0	0
<i>Chaerophyllum hirsutum</i>	Chahir	0.4	0.4	0.1	0	0	0	0	0	0	0.02	0.02	0	0	0	0	0	0
<i>Chamaenerium angustifolium</i>	Chaang	0	0	0	0.1	0.4	0	0.51	0.53	0.45	0	0	0	0	0	0	0	0
<i>Juncus effusus</i>	Juneff	0	0	0	0	0	0	0	0	0	0	0	0.05	0.02	0	0	0	0
<i>Juncus filiformis</i>	Junfil	0	0	0	0	0	0.02	0.13	0.13	0.43	0	0	0	0	0	0	0	0
<i>Lamium galeobdolon</i>	Lamgal	0	0	0	0	0	0	0	0	0	0	0	0	0	0.27	0.07	0.02	0.02
<i>Lamium maculatum</i>	Lammac	0	0	0	0	0	0	0	0	0	0.05	0.05	0	0	0.47	0.13	0.05	0
<i>Lamium montanum</i>	Lammmon	0	0	0	0	0	0	0	0	0	0.05	0	0	0	0	0	0	0
<i>Larix decidua</i>	Lardec	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0	0	0	0
<i>Lilium martagon</i>	Lilmar	0	0	0	0	0	0	0	0	0	0	0	0	0	0.13	0.13	0.13	0.05
<i>Lunaria rediviva</i>	Lunred	0	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0.05	0.02	0
<i>Luzula nemorosa</i>	Luznem	0	0	0	0	0	0.34	0.39	0.43	0.56	0.5	0.42	0.52	0.67	0.27	0.18	0.1	0.05
<i>Luzula pilosa</i>	Luzpil	0.2	0.2	0	0	0	0.21	0.07	0	0	0	0	0	0	0.02	0.02	0	0
<i>Luzula sylvatica</i>	Luzsyl	0	0	0	0	0	0.18	0.15	0.21	0.28	0	0	0	0	0	0	0	0
<i>Lycopodium annotinum</i>	Lycann	0.2	0.2	0.2	0.4	0.4	0.48	0.19	0.17	0.27	0.33	0.33	0.33	0.33	0	0	0	0
<i>Lysimachia nemorum</i>	Lysnem	0	0	0	0	0	0	0	0	0	0.13	0.13	0.05	0	0.35	0.35	0.05	0.05
<i>Maianthemum bifolium</i>	Maibif	0.2	0.2	0.2	0.1	0.1	0.38	0.35	0.17	0.13	1.4	1.4	0.67	0.35	2.08	2.1	2.18	2.15
<i>Milium effusum</i>	Mileff	0	0	0	0	0	0	0	0	0	0	0	0.02	0.05	10.88	10.88	6.97	6.97
<i>Moehringia trinervia</i>	Moetri	0	0	0	0	0	0	0	0	0	0	0	0	0	0.13	0.05	0	0
<i>Mycelis muralis</i>	Mycmur	0	0	0	0	0	0	0	0	0	0.18	0.18	0.38	0.33	0.45	0.2	0.1	0.03
<i>Oxalis acetosella</i>	Oxaace	2.1	2.1	0.9	0.5	0.4	0.56	0.56	0.31	0.25	1.75	0.92	1.12	1.08	1.6	1.32	0.92	1.08
<i>Paris quadrifolia</i>	Parqua	0	0	0	0	0	0	0	0	0	0.05	0.05	0.05	0.05	0.17	0.15	0.13	0.13
<i>Phegopteris connectilis</i>	Phecon	0.8	0.9	0.9	0.9	0.9	0.05	0.02	0.01	0	1.33	0.93	0.38	0.33	0	0	0	0
<i>Phyteuma spicatum</i>	Physpi	0	0	0	0	0	0	0	0	0	0.02	0.02	0	0	0.05	0.05	0.05	0
<i>Picea abies</i>	Picabi	0.8	0.8	0.6	0.7	2.4	0.79	1.23	1.01	1.67	3.37	3.23	1.18	1.43	0.15	0.08	0.12	0.17
<i>Picea pungens</i>	Picpun	0	0	0	0	0	0	0.43	0.13	0	0	0	0	0	0	0	0	0
<i>Poa remota</i>	Poarem	0	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0.02	0	0
<i>Polygonatum multiflorum</i>	Polmul	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0.02	0.02	0.05
<i>Polygonatum verticillatum</i>	Polver	0	0	0	0	0	0.06	0.02	0.01	0.01	0.83	0.83	0.67	0.53	2.07	2.07	1.23	2.15
<i>Prenanthes purpurea</i>	Prepur	0.1	0.1	0	0	0	0	0	0.03	0.14	1.12	1.03	0.95	1.1	2.47	2.38	1.07	1.07

Continuation of Tab. II

	Abbreviation	ss 1	ss 2	ss 3	ss 4	ss 5	Pi 1	Pi 2	Pi 3	Pi 4	x 1	x 2	x 3	x 4	Fa 1	Fa 2	Fa 3	Fa 4
<i>Pulmonaria officinalis</i>	Puloff	0	0	0	0	0	0	0	0	0	0	0	0	0	0.05	0.05	0.05	0.02
<i>Ranunculus platanifolius</i>	Ranpla	0	0	0	0	0	0	0	0	0	0.05	0.02	0	0	0	0	0	0
<i>Rubus fruticosus</i>	Rubfru	0	0	0	0	0	0	0	0	0	0	0	0.02	0.05	0.18	0.33	0.33	0.33
<i>Rubus idaeus</i>	Rubida	0	0	0	0.1	0.4	0.07	0.16	0.07	0.14	0.62	0.82	0.52	0.43	3.23	3.07	2.92	2.92
<i>Rumex acetosa</i>	Rumace	0	0	0	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0
<i>Rumex arifolius</i>	Rumari	0	0	0	0	0	0.08	0.03	0.02	0.02	0	0	0	0	0	0	0	0
<i>Salix caprea</i>	Salcap	0	0	0	0	0	0	0.13	0.05	0.01	0	0	0	0	0	0	0	0
<i>Sambucus racemosa</i>	Samrac	0	0	0	0	0	0	0	0	0	0.1	0.02	0	0	0.38	0.1	0.18	0.35
<i>Senecio fuchsii</i>	Senfuc	0	0	0	0	0	0	0	0	0	0.05	0.05	0	0	0.05	0.02	0.05	0.02
<i>Senecio nemorensis</i>	Sennem	0.2	0.2	0	0	0	0.07	0.57	0.47	0.63	0.78	0.42	0.35	0.18	0.85	0.43	0.45	0.37
<i>Sorbus aucuparia</i>	Sorauc	0.3	0.4	0.1	0.1	0.1	0.62	0.52	0.33	0.32	0.65	0.23	0.33	0.35	1.2	0.75	0.27	0.17
<i>Stachys sylvatica</i>	Stasyl	0	0	0	0	0	0	0	0	0	0	0	0	0	0.13	0	0	0
<i>Stellaria nemorum</i>	Stenem	0.2	0.2	0.1	0	0	0.09	0.05	0.02	0	0.52	0.43	0.4	0.35	0.4	0.4	0.37	0.35
<i>Streptopus amplexifolius</i>	Stramp	0.5	0.5	0.4	0.3	0.3	0.25	0.16	0.07	0.04	0.02	0.02	0	0.02	0	0	0	0
<i>Trientalis europaea</i>	Trieur	0.7	0.8	1.3	2	2	1.19	1.01	1.01	1.13	0.13	0.05	0	0	0.33	0.33	0.05	0.02
<i>Trichophorum caespitosum</i>	Tricac	0	0	0	0	0	0.05	0.13	0.13	0.43	0	0	0	0	0	0	0	0
<i>Urtica dioica</i>	Urtdio	0	0	0	0	0	0	0.02	0.02	0.01	0	0	0	0	0.13	0.05	0	0
<i>Vaccinium myrtillus</i>	Vacmyr	13.1	13.1	11.9	16.6	17.5	15.77	14.71	17.76	21.47	11.35	9.27	10.03	10.02	21.92	21.92	14.85	12.77
<i>Vaccinium vitis-idaea</i>	Vacvit	0.2	0.2	0.1	0	0	0.24	0.16	0.08	0.15	0	0	0	0	0	0	0	0
<i>Veratrum lobelianum</i>	Verlob	0.2	0.2	0.1	0	0	0.15	0.11	0.05	0.03	0.1	0.03	0	0	0	0	0	0
<i>Viola biflora</i>	Viobif	0.6	0.6	0.1	0	0	0.05	0.02	0	0	0	0	0	0	0	0	0	0
<i>Viola sylvatica</i>	Viosyl	0	0	0	0	0	0	0	0	0	0.05	0	0	0	0	0	0	0
<i>Anastrepta ocradensis</i>	Anaocr	0	0	0	0	0	0.05	0	0	0	0	0	0	0	0	0	0	0
<i>Barbilophozia attenuata</i>	Baratt	0.15	0.06	0.06	0	0	0.02	0.02	0.02	0	0	0	0	0	0	0	0	0
<i>Barbilophozia floerkei</i>	Barflo	0.02	0.02	0.02	0	0	0.05	0.01	0.01	0	0	0	0	0	0	0	0	0
<i>Barbilophozia lycopodioides</i>	Barlyc	0	0	0	0	0	0.05	0.01	0	0	0	0	0	0	0	0	0	0
<i>Bazania trilobata</i>	Baztri	0	0	0	0	0	0.02	0.02	0.01	0.01	0	0	0	0	0	0	0	0
<i>Blepharostoma trichophyllum</i>	Bletri	0.15	0.06	0.02	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0
<i>Bryum erythrocarpum</i>	Bryery	0.14	0.06	0	0	0	0.03	0.01	0	0	0	0	0	0	0	0	0	0
<i>Calypogeia integristipula</i>	Calint	0.06	0.02	0	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0
<i>Calypogeia neesiana</i>	Calnee	0	0	0	0	0	0.02	0.01	0	0	0	0	0	0	0	0	0	0
<i>Calypogeia trichomanis</i>	Caltri	0.42	0.29	0.04	0.02	0	0.18	0.09	0.01	0	0	0	0	0	0	0	0	0
<i>Cephalozia biscupidata</i>	Cepbis	0.18	0.14	0.08	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0
<i>Cetraria islandica</i>	Cetisl	0.02	0.02	0.02	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0
<i>Dicranella heteromalla</i>	Dichet	0	0	0	0	0	0	0	0	0	0.33	0.29	0.23	0.2	0.29	0.29	0.26	0.24
<i>Dicranodontium denudatum</i>	Dicden	0.15	0.15	0.06	0.06	0.02	0.41	0.31	0.13	0.19	0	0	0	0	0.14	0	0.02	0

Continuation of Tab. II

	Abbreviation	ss 1	ss 2	ss 3	ss 4	ss 5	Pi 1	Pi 2	Pi 3	Pi 4	x 1	x 2	x 3	x 4	Fa 1	Fa 2	Fa 3	Fa 4
<i>Dicranum scoparium</i>	Diesco	2.56	2.56	2.9	2.9	2.9	3.47	2.82	3.03	2.34	1.77	1.56	0.57	0.75	1.46	0.84	0.56	0.35
<i>Diplophyllum obtusifolium</i>	Dipobt	0.06	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0
<i>Hylocomium splendens</i>	Hylspl	0.17	0.06	0.04	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Jungermania sphaerocarpa</i>	Junsph	0.06	0.02	0	0	0	0.02	0.02	0	0	0	0	0	0	0	0	0	0
<i>Lepidozia reptans</i>	Leprep	0.12	0.08	0.02	0	0	0.01	0.01	0	0	0	0	0	0	0	0	0	0
<i>Lophozia ventricosa</i>	Lopven	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0
<i>Mnium hornum</i>	Mnihor	0.15	0.15	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Mylia taylorii</i>	Myltay	0.48	0.52	0.52	0.06	0.1	0.18	0.15	0.11	0.1	0	0	0	0	0	0	0	0
<i>Orthodicranum montanum</i>	Ortmon	0.06	0.02	0	0	0	0.02	0.02	0	0	0	0	0	0	0	0	0	0
<i>Pellia neesiana</i>	Pelnec	0.63	0.63	0.27	0.08	0.04	0.39	0.29	0.33	0.22	0.1	0.1	0.02	0	0	0	0	0
<i>Philonotis fontana</i>	Phifon	0.15	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Plagiomnium affine</i>	Plaaff	0.55	0.55	0.21	0.08	0.15	0.25	0.25	0.16	0.14	0	0	0	0	0	0	0	0
<i>Plagiothecium curvifolium</i>	Placur	0.06	0.06	0.02	0	0	0	0	0	0	0	0	0	0	0.12	0.02	0.02	0
<i>Plagiothecium denticulatum</i>	Pladen	0.15	0.06	0.06	0	0	0.02	0	0	0	0	0	0	0	0	0	0	0
<i>Plagiothecium laetum</i>	Plalae	0.06	0.06	0	0	0	0.17	0.14	0.02	0.02	0	0	0	0	0	0	0	0
<i>Plagiothecium roeseanum</i>	Plaroe	0	0	0	0	0	0	0	0	0	0.02	0	0	0	0	0	0	0
<i>Plagiothecium undulatum</i>	Plaund	1.66	1.66	0.63	0.48	0.46	0.8	0.72	0.5	0.46	0	0	0	0	0	0	0	0
<i>Pleurozium schreberi</i>	Plesch	0	0	0	0	0	0.06	0.05	0.02	0.02	0	0	0	0	0.37	0.24	0.03	0
<i>Pohlia nutans</i>	Pohnut	0.45	0.45	0.21	0.08	0.06	0.11	0.1	0.06	0.05	0.15	0.07	0	0	0.07	0.07	0.03	0.02
<i>Polytrichastrum formosum</i>	Polfor	17.55	17.55	15.3	8.7	8.7	12.48	7.37	5.17	4.57	4.58	4.58	2.54	2.23	2.75	2.75	2	2
<i>Polytrichum commune</i>	Polcom	0.15	0.15	0.02	0	0	0.29	0.26	0.13	0.2	0	0	0	0	0.02	0.02	0	0
<i>Polytrichum gracile</i>	Polgra	0	0	0	0	0	0.01	0	0	0	0	0	0	0	0	0	0	0
<i>Polytrichum juniperinum</i>	Poljun	0	0	0	0	0	0	0	0	0	0.05	0.05	0	0	0.12	0.02	0	0
<i>Polytrichum strictum</i>	Polstr	0	0	0	0	0	0.13	0.13	0.13	0.13	0	0	0	0	0	0	0	0
<i>Rhizomnium punctatum</i>	Rhipun	0.55	0.55	0.08	0.08	0.08	0.2	0.17	0.02	0	0.07	0	0	0	0	0	0	0
<i>Rhynchostegium confertum</i>	Rhycon	0.21	0.15	0.08	0	0	0.07	0.01	0	0	0	0	0	0	0	0	0	0
<i>Rhytidadelphus loreus</i>	Rhylor	0	0	0	0	0	0.05	0	0	0	0	0	0	0	0	0	0	0
<i>Rhytidadelphus squarrosus</i>	Rhysqu	0	0	0	0	0	0.04	0.01	0	0	0	0	0	0	0	0	0	0
<i>Scapania undulata</i>	Scaund	0.36	0.36	0.14	0	0	0.26	0.12	0.03	0.01	0	0	0	0	0	0	0	0
<i>Sphagnum girgensohnii</i>	Sphgir	9.8	9.8	7.46	3.06	3	2.47	2.06	2.6	2.57	0	0	0	0	0	0	0	0
<i>Sphagnum nemoreum</i>	Sphnem	0.06	0.06	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
<i>Sphagnum quinquefarium</i>	Sphqui	0.21	0.21	0.15	0.15	0.15	0.07	0.04	0.02	0.02	0	0	0	0	0	0	0	0
<i>Sphagnum recurvum</i>	Sphrec	0	0	0	0	0	2.75	2.75	1.92	1.92	0	0	0	0	0	0	0	0
<i>Sphagnum robustum</i>	Sphrob	0.21	0.21	0.12	0	0	0.43	0.43	0.13	0.13	0	0	0	0	0	0	0	0
<i>Sphagnum squarrosus</i>	Sphsqu	0.52	0.48	0.08	0	0	0.91	0.71	0.52	0.54	0	0	0	0	0	0	0	0
<i>Tetraphis pellucida</i>	Tetpel	0.46	0.4	0.15	0	0	0.25	0.06	0	0	0	0	0	0	0	0	0	0



3. DCA ordination of the relevés without the most different plots (11–14, 30–32), axis 1 and 2 (A), 2 and 3 (B) respective. Position of the relevé at single periods 1 to 4, 5 respective is distinguished by breaking points of the trajectory, its end is described by number of the plot and the last period of sampling

hypothesis is rejected if the variable χ^2 has been higher than the relevant critical value of χ^2 -distribution:

$$\chi^2 = \sum_y \frac{(1 - P(c_{py}))^2}{P(c_{py})}$$

The probability $P(c_{py})$ was calculated on the basis of frequencies of types within the whole set of plots, and for a set of plots from which plot p was excluded. The presented algorithm is a part of new program DVCLASS for sequence study.

RESULTS AND DISCUSSION

VEGETATION CONDITIONS

Vegetation of the groups of forest stands (spruce, beech and mixed ones) was described on the basis of

“mean“ relevés for each of the periods evaluated (Tab. II). These data were used for classification (Fig. 1) and ordination (Fig. 2). Variability of the separate plots under observation with respect to their development is indicated by ordination of the relevés (Fig. 3). The results document evident differences between the groups of plots and their different behaviour during succession.

The smallest changes in the herb layer were observed in spruce stands (except PRP 11–15) while the changes in mixed stands were greater, and they were greatest in beech stands (assessed from the classification of mean relevés) (compare V a c e k et al., 1996c).

Spruce stands

Relatively small changes can be commonly described e.g. by a decrease in the coverage of *Dryopteris*

dilatata or by complete disappearance of *Blechnum spicant*, *Luzula pilosa* and *Viola biflora*. On the contrary, the coverage of *Vaccinium myrtillus* and *Deschampsia caespitosa* was increasing, some clear-cut species appeared as the new ones (*Chamaenerium angustifolium* or *Prenanthes purpurea*).

Development on PRP 11–15 was different: e.g. *Adenostyles alliariae* and *Dryopteris dilatata* receded while the representation of *Athyrium distentifolium* increased (cf. Vacek, 1984b).

Mixed stands

Changes in the average coverage of many species were relatively low – they involved e.g. reduction in the

coverage of *Blechnum spicant*, *Dentaria enneaphyllos*, *Dryopteris dilatata*, *Dryopteris filix-mas*, *Gymnocarpium dryopteris* and *Trientalis europaea*, and complete disappearance of *Asperula odorata* and *Ranunculus platanifolius*. On the other hand, the species occurring more frequently on clear-cut areas or in open forest stands increased their coverage or appeared as new species: *Digitalis purpurea*, *Juncus effusus*, *Rubus fruticosus* agg., *Carex pilosa*. The coverage of *Fagus sylvatica* in the herb layer augmented.

Beech stands

The average coverage of many species decreased (*Dentaria enneaphyllos*, *Dryopteris filix-mas*, *Hiera-*

III. Development of the species diversity ('H) and its components during single periods of sampling according to the groups of plots, herb layer. avg – average, std – standard deviation

Period	1			2			3			4			5		
	S	e	'H												
TVP 11–15															
avg	16.0	0.545	1.480	16.6	0.541	1.504	16.6	0.581	1.598	14.8	0.637	1.704	14.6	0.661	1.774
std	4.5	0.050	0.172	3.5	0.054	0.156	4.0	0.076	0.122	2.0	0.094	0.237	1.9	0.075	0.257
<i>Picea</i>															
avg				15.1	0.550	1.473	15.9	0.559	1.537	14.1	0.581	1.525	12.8	0.597	1.504
std				4.5	0.099	0.350	3.3	0.078	0.268	3.1	0.091	0.291	3.3	0.110	0.338
Mixed stands															
avg				22.7	0.661	2.056	21.7	0.652	2.001	18.2	0.666	1.932	16.8	0.674	1.907
std				2.9	0.063	0.190	2.4	0.045	0.145	2.3	0.098	0.338	2.4	0.094	0.341
<i>Fagus</i>															
avg				21.8	0.585	1.778	21.7	0.565	1.700	19.2	0.615	1.793	18.3	0.687	1.963
std				8.5	0.141	0.530	9.4	0.141	0.540	7.9	0.101	0.496	7.2	0.070	0.390

S – number of species, e – equitability, 'H – Shannon-Wiener index of diversity

IV. Development of the species diversity ('H) and its components during single periods of sampling according to the groups of plots, moss layer. avg – average, std – standard deviation

Period	1			2			3			4			5		
	S	e	'H	S	e	'H	S	e	'H	S	e	'H	S	e	'H
TVP 11–15															
avg	15.2	0.525	1.432	14.0	0.497	1.319	11.4	0.478	1.169	6.2	0.597	1.044	5.2	0.654	1.007
std	4.0	0.085	0.364	4.0	0.096	0.398	3.6	0.104	0.408	1.9	0.063	0.222	1.9	0.105	0.228
<i>Picea</i>															
avg				10.1	0.585	1.353	8.3	0.647	1.310	6.4	0.610	1.075	5.1	0.629	0.954
std				2.3	0.150	0.417	2.9	0.118	0.326	2.5	0.098	0.319	2.6	0.086	0.435
Mixed stands															
avg				4.3	0.629	0.895	3.7	0.598	0.761	3.0	0.601	0.621	2.8	0.665	0.672
std				0.7	0.126	0.142	0.5	0.151	0.174	0.6	0.157	0.094	0.4	0.160	0.150
<i>Fagus</i>															
avg				5.2	0.691	1.113	4.0	0.637	0.868	3.5	0.548	0.686	2.8	0.577	0.591
std				1.2	0.034	0.156	0.8	0.088	0.156	1.3	0.169	0.307	0.7	0.195	0.214

S – number of species, e – equitability, 'H – Shannon-Wiener index of diversity

V. Results of the classification of relevés by TWINSpan procedure. Used cut-levels: 0.00, 1.00, 10.00, 31.60, 56.20, 75.00. Indicator species of the classification groups are presented (number by the species abbreviation represents cover according to the number of the cut-level)

0 Des fle 2 Tri eur 1	00 Ath dis 1 Str amp 1 Des cae 1 Gal her 1 Ver lob 1	000 Car lim 1	001	0010 Ade all 1	00100 Cha hir 1	001010 Cic alp 1		
					00101	001011		
			0011	00110 Vac myr 3	001100 Gal her 2	0011000 Gen asc 2	0011001	
					001101 Des fle 3 Des cae 1	0011010 0011011		
				00111	001110 Ath dis 1	001111		
					01000 Sen nem 1	010000 010001 Des cae 1		
					01001	010010 Luz syl 1		
						010011	0100110 Gal her 1 Str amp 1 Cal vil 5 Des cae 1	
							0100111 Vac myr 3 Gen asc 1	
					0101 Vac myr 3 Cha ang 1	01010 Oxa ace 1 Lyc ann 1	010100 010101 Ath dis 1	0101000 Lyc ann 1 Mai bif 1
					01011 Sen nem 1 Bet pen 1 Cha ang 2 Cal vil 4	010110 010111 Mai bif 1	0101010 0101011 Abi alb 1 0101100 0101101 Ath dis 1	
			011 Pre pur 1 Fag syl 1 Pol ver 1	0110	01100	011000 011001 Gym dry 1	0110000 Luz pil 1 0110001	
				0111 Gen asc 1	01101 Bet pen 1 01110 Pre pur 2	011011 011110 Des fle 1	0110111 0111100 0111101 Ble spi 1	
					01111			
	1 Gym dry 1 Dry fil 1 Ath fil 1	10	100 Des fle 1 Fag syl 3	1000 Ace pse 1	1001			
			101	1010 1011 Ble spi 1				
			110	1100 Asp odo 1 11001				
		11 Bra syl 1	111 Act spi 1	1101 Lam gal 1				

cium sylvaticum, *Luzula nemorosa*, *Milium effusum*, *Vaccinium myrtillus*) or some species fully disappeared (*Circaea intermedia*, *Lamium maculatum*). The coverage of some grasses increased (*Brachypodium sylvaticum* and *Calamagrostis villosa*). The coverage of *Acer pseudoplatanus* in the herb layer also augmented.

GENERAL TRENDS

Changes in species diversity and vegetation richness were great (Tabs. III, IV): the numbers of species in herb (E_1) and moss (E_0) layers declined in all groups of plots, overall species diversity of moss layer was also substantially reduced. Its changes in herb layer were specific of the groups of plots: it increased on PRP 11–15 (spruce stands with subalpine elements) and partly in beech stands, but it decreased in mixed stands or there were basically no changes in the other spruce stands. Both components of diversity influenced species equitability which was mostly improved.

The relevés were classified and evaluated by ordination analysis (Fig. 3). Classification by TWINSpan program provided identification of typical groups of species (Tab. V). Such classification indicated the existence of a "core" consisting of relevés from spruce stands. Other relevés were 'chain-linked' to the core, among them those from mixed and beech stands. Vegetation on PRP 14 was the most different of all.

Ordination (DCA) of all relevés indicated perceptibly different vegetation of some beech stands (PRP 30, 31, 32; localities in the 6th altitudinal zone) in a direction of the first ordination axis. The second ordination axis pointed to the different character of herbaceous vegetation at four localities with spruce stand (PRP 11–14; these were stands with significant participation of species occurring in sub-alpine tall-herbaceous meadows).

In order to evaluate vegetation changes without influence of the ordination space distortion caused by these most differing relevés, second run of ordination was carried out excluding the seven mentioned localities. It was possible to draw a conclusion from positions of various species in the resultant ordination space and from their requirements that the first axis applied to soil richness (nutrient contents, mainly of nitrogen) while the second showed soil moisture.

The question about the existence of any general trends in species composition on the plots was answered by analysis of variance used to interpret ordination scores of plots along the first to the fourth axis when the year of relevé was taken as a parameter of classification to groups. It is to state that no trends were determined that would support the hypothesis of the existence of these trends – either within the whole group of plots or for the separate groups (with spruce, beech and mixed stands).

Moss layer composition and its changes during succession were evaluated separately. The changes in the

species composition of moss layer did not appear to be so important as the changes in its whole coverage that were great in some cases (but it was necessary to compare trends of the species diversity in moss layer). After the tree layer of spruce stands was disturbed (where E_0 coverage was primarily high), E_1 coverage increased – mainly grasses vehemently competed with bryophytes, which can fully disappear as a result of this competition (cf. PRP 26). The species composition of bryophytes in spruce stands and in beech stands did not differ significantly. Water supply and potential peatification are the most important factors of moss layer development (see PRP 23 with E_0 coverage 65–75%). Classification indicated the existence of two large and relatively homogeneous groups of relevés, and of several very different relevés (except the above-mentioned PRP 23, these were relevés from localities PRP 13, 14, 16, 21 and 25 – with spruce stand and moss layer of great cover; the greatest changes in E_0 occurred at these localities after tree layer destruction). Ordination of relevés on the basis of moss layer composition confirmed an important role of peatification (PRP 23) largely distorting the ordination space. This is the reason why changes during succession should be evaluated when the position in the projection of second and third ordination axes was available: the second axis indicated moderate differences between five spruce stands (PRP 4, 5, 13, 14 and 21). A greater change in the moss layer structure was established from a shift along the third ordination axis in spruce-beech and beech stands on PRP 2, 8, 9, 30 and 32.

Tab. VI documents the level of changes in herb layer from the early eighties to 1995 (evaluation by variability in ordination space while various numbers – 1 to 4 – of the first ordination axes were used). Not absolute values of the indexes but their mutual comparison was important. Differences between the groups of forest ecosystems were perceptible – the greatest changes were determined in beech stands while changes in spruce stands were smaller. With respect to the position of these groups in ordination space, a change evaluated only from a shift along the first axis of ordination space could be considered as conclusive in beech stands but it was necessary to evaluate at least the first two axes in spruce stands. Changes in vegetation evaluated by these indexes reflected global processes, scarce processes (including fluctuation) may have been neglected.

Similar comparisons were made to evaluate changes in the moss layer composition (Tab. VII). Comparison of the two tables showed that the level of successive changes was comparable in the two layers. Greatest changes in moss layer occurred mostly in mixed stands, that means in stands with relatively smaller changes in herb layer. Moss layer was more stable in the spruce stands.

Forest stands were divided into four groups with regard to the above-mentioned level of changes in the respective layers using medians calculated from all values (Tab. VIII). Plots with tree layer destruction were marked for illustration; they occur in all these groups.

TREE LAYER DAMAGE

Spruce stands

Tab. IX shows foliage dynamics of spruce stands, and it is plotted in Fig. 4 using an example of PRP 11 and 13. The values document that foliage trends on the separate plots were largely different, in relative terms corresponding to environmental pollution stress, state, structure and origin of forest stand, and to cultural and protective treatments of forest managers. While the average regional foliage of spruce stands was about 83.9% in 1976, it was only 39.7% in 1997, it implied an average annual increase in defoliation of 2.1%. Three typical periods were identified on the basis of different trends of foliage dynamics. In the period of first signs of damage (1976–1980), average foliage ranged from 84.9 to 85.4% at lower mountain altitudes (below 1100 m above sea level) and between 79.2 and 81.8% at higher altitudes (above 1100 m a.s.l.) (average annual

increase in defoliation of 0.1% and 0.6%, respectively). In the period of great damage (1981–1988), average foliage ranged from 48.2 to 79.3% at lower altitudes and from 46.7 to 73.8% at higher altitudes (average annual increase in defoliation of 4.4 and 3.0%). If damage caused by bark beetle was excluded, average annual defoliation increase would be 2.3% at lower altitudes and 2.8% at higher altitudes. In the period of damage abatement (1989–1997), average foliage ranged from 19.9 to 47.5% at lower altitudes and from 42.8 to 52.6% at higher altitudes (average annual increase in defoliation was 3.5 and 1.2%), if damage caused by bark beetle was excluded, 0.2 and 0.4%. The years 1981, 1983, 1984 and 1986 were critical due to the environmental pollution stress, when defoliation was severe. PRP 3, 17 and 26 were felled in the period of heavy environmental pollution stress. PRP 15 was liquidated by bark beetle invasion and PRP 16 and 18 suffered great damage (cf. V a c e k , 1996).

Maximum average defoliation of spruce-trees was 30–100% (average value 55.5%) in spruce stands and

VI. Change in the herb layer of the single plot groups expressed as sum of variances of ordination scores of several (one to four) first DCA axes

Axes DCA	[1]	[1]+[2]	[1]+[2]+[3]	[1]+...+[4]
<i>Fagus</i>				
Average	27.07	31.35	42.97	49.95
min	3.84	5.22	5.70	25.24
max	38.13	42.08	80.14	84.18
<i>Fagus + Picea</i>				
Average	21.10	25.89	26.76	32.84
min	5.59	7.47	7.81	16.22
max	40.92	46.95	49.73	52.47
<i>Picea</i>				
Average	5.64	24.09	25.28	36.17
min	0.43	1.30	1.54	9.60
max	21.01	103.17	103.25	103.50

VII. Change in the moss layer of the single plot groups expressed as sum of variances of ordination scores of several (one to four) first DCA axes

Axes DCA	[1]	[1]+[2]	[1]+[2]+[3]	[1]+...+[4]
<i>Fagus</i>				
Average	14.7	24.4	40.1	46.2
min	2.0	11.2	16.7	26.8
max	40.0	59.3	110.5	116.8
<i>Fagus + Picea</i>				
Average	2.5	14.9	48.1	55.7
min	1.2	4.5	14.1	19.7
max	3.8	38.3	120.5	134.9
<i>Picea</i>				
Average	9.0	19.8	27.0	38.7
min	0.8	5.5	11.4	18.7
max	24.7	58.9	62.1	76.3

VIII. Division of the stands into groups according to the change in the herb and moss layer, respectively (see Tabs. VI and VII); μ is median. Bold pressed are numbers of stands with total destruction of the tree layer

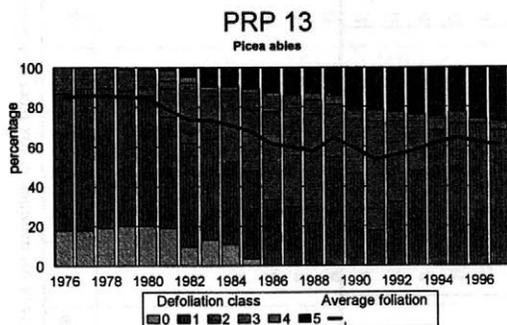
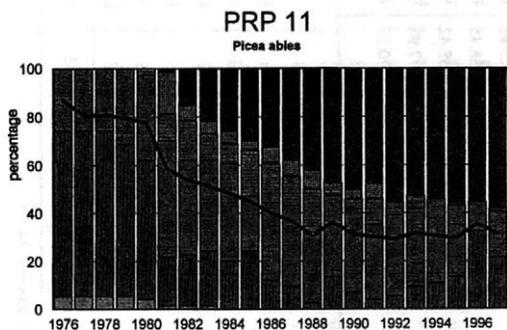
A. Herb layer (E_1); $\mu = 25.81$			
Change degree	Spruce stands	Mixed stands	Beech stands
$<\mu/2$	18 20 25		
$\mu/2.. \mu$	16 22 19 3 26 10 23 21 24	6 9	28
$\mu.. 2\mu$	4 5	1 7 8	27 29
$>2\mu$	15 12 17 13 14 11	2	31 32 30
B. Moss layer (E_0); $\mu = 35.59$			
Change degree	Spruce stands	Mixed stands	Beech stands
$<\mu/2$			
$\mu/2.. \mu$	23 13 5 4 21 12 19 25 24	1 7 6	30 29 27 28
$\mu.. 2\mu$	17 10 22 15 20 16 11 18 14 3	2 9	31
$>2\mu$	26	8	32

35–100% (average value 55.7%) in mixed spruce-beech stands in the period of observation.

Tab. X shows types of transition matrices for the separate localities. A typical sequence of transition matrices was (E)-A,F-(B)-(C)-(B)-G-A while the last two members were very doubtful, bracketed symbols indicated repetition of the type for two or more years (Fig. 5). Regular deterioration of spruce health was observed since 1981 or 1982, another turning point was the years 1986 and 1987 with peak of deterioration (the peak of damage was determined on some plots later), then the changes were not so abrupt, development was quite continuous [both in the sense of health improvement (e.g. PRP 5, 7) or total destruction of stand (PRP 14 or 1)].

Types of transition matrices can be divided into these groups (Fig. 6):

- decrease in average defoliation accompanied by a lower level of tree dieback (types M, K, B, D, F) – they describe the stages of stand health improvement;
- increase in average defoliation accompanied by a lower level of tree dieback (types A, E), the “first” stage of stand state deterioration;
- decrease in average defoliation accompanied by a higher level of tree dieback (types O, G, C) – the trees that cannot tolerate the given level of stress are eliminated in these stages; in case the period is not very long, there is a hope of conserving the stand without its total destruction if the number of surviving trees is sufficiently high;



4. Trend of average foliage and particular defoliation class of spruce on plots 11 and 13

- increase in defoliation accompanied by parallel intensive dieback of trees (types I, H, N) – stages of extreme damage to stand ending in its total destruction.

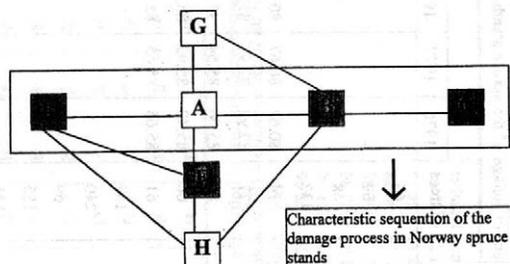
The presence of data on defoliation assessment at localities PRP 11–15 from the mid-seventies can allure to generalise the results from these localities for the whole area of the Krkonoše Mts. It is however necessary to be aware of the fact that these autochthonous spruce stands in extreme conditions of the Mumlava anemo-orographic system (cf. V a c e k, 1981) differ in some parameters from the other spruce stands – see vegetation evaluation.

Beech stands

Tab. XI shows foliage dynamics of beech stands, and it is plotted in Fig. 7 using an example of PRP 27 and 31. The average regional foliage of beech stands was 94.2% in 1980 and only 62.7% in 1997, i.e. an average annual increase in defoliation was 1.9%. In the period of great damage (1981–1988) the average foliage of beech-trees ranged from 64.5 to 88.6%, i.e. an average annual increase in defoliation was 3.4%. If damage caused by beech scale and wood-destroying fungi were excluded, it would be about 2.0%. 1981, 1984 and 1986 were the years with critical environmental pollution stress, when an annual decrease in foliage was about 5.2–9.6% (cf. V a c e k, 1993). In the period of damage abatement (in 1989–1997), average beech foliage ranged between 62.7 and 68.0%, i.e. an average annual increase in defoliation was 0.7%. If damage caused by beech scale and lignicolous fungi were excluded, an annual increase in foliage would be about 0.4%.

Maximum average defoliation of beech was 24–46% (average 33.5%) in beech stands and 22–38% (average 33.2%) in mixed spruce-beech stands in the period of observation.

Tab. XII shows the types of transition matrices for the separate localities. It was not possible to identify any typical sequence of transition matrices. The initial period could be described by matrices of types A and C, types C and D were typical of the intermediate period, and heterogeneity increased in the last years (e.g. types H and F were additional ones). The beginning of



5. Graph of the main sequences of the transition matrix types by *Picea abies*. Types in grey fields have a frequent repetition

IX. Dynamics of foliage of the spruce stands (average values in per cent; totally defoliated trees are included)

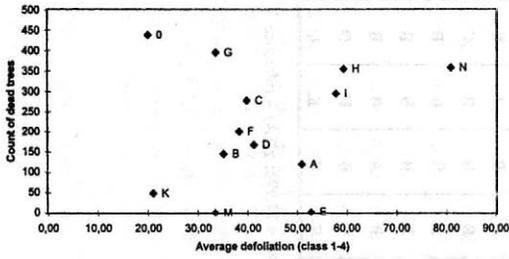
Permanent plot	Number of trees	Year																					
		1976	1977	1978	1979	1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997
3	65						72.00	68.15															
4	69						78.70	76.16	70.73	64.06	60.58	51.81	45.14	47.40	47.50	45.00	42.10	46.38	42.90	45.15	43.62	43.48	41.66
5	48						82.19	78.44	73.75	68.44	65.00	57.92	50.00	53.44	58.60	53.12	49.27	56.04	58.44	63.02	59.48	56.87	57.29
10	132						71.67	69.77	66.97	61.48	57.42	51.63	47.65	42.35	48.60	47.50	46.59	46.14	50.87	48.26	47.95	50.19	50.30
11	74	80.67	80.67	80.67	79.59	77.50	58.58	53.85	51.42	48.38	44.93	40.07	36.82	30.88	36.15	31.22	29.86	29.26	31.55	30.41	29.46	35.47	31.28
12	77	82.92	82.92	82.92	82.14	80.84	68.23	64.48	64.87	60.78	57.27	47.29	44.35	38.12	41.69	36.75	34.42	33.72	34.36	30.26	31.04	27.59	25.78
13	84	85.06	85.54	85.71	84.88	84.64	78.69	73.39	73.39	70.65	67.62	61.49	59.76	57.76	64.52	59.40	53.45	56.01	58.33	62.08	64.58	62.76	60.77
14	60	85.42	85.92	86.17	86.08	86.08	80.50	80.67	77.75	70.50	67.67	62.25	61.83	56.33	63.92	58.25	53.17	50.92	53.92	58.92	52.17	43.92	39.08
15	61	85.65	85.65	85.90	84.26	83.93	76.31	75.16	60.98	56.56	54.67	46.80	44.59	35.16	31.17	28.36	26.48	26.39	10.66	2.79	2.29	1.32	0.73
16	187						79.01	75.59	69.12	61.87	53.58	50.96	53.77	50.16	56.00	53.26	51.98	45.99	43.66	39.41	24.70	16.31	
17	240						83.46	75.29	68.88	58.29	50.21	46.08	44.33										
18	69						75.87	69.42	62.17	52.32	47.68	44.64	42.32	38.12	44.50	39.49	34.71	35.07	32.68	31.52	31.30	4.63	3.11
19	155						77.13	74.19	69.68	64.23	61.77	54.84	50.58	48.71	55.40	51.19	49.10	52.10	48.32	54.03	54.48	54.55	55.10
20	121						76.98	74.26	70.17	63.43	61.94	55.70	52.69	47.98	49.30	47.73	46.61	51.74	45.94	50.61	49.42	47.77	40.95
21	140						70.14	71.43	67.71	63.86	62.29	56.07	57.36	49.89	58.10	57.96	57.64	52.46	57.04	58.64	57.57	59.86	59.39
22	145						79.31	76.90	73.34	68.45	66.66	59.72	56.38	53.41	61.10	60.03	56.59	56.69	54.72	57.97	60.27	58.86	62.58
23	199						71.93	68.72	64.60	58.42	54.45	46.01	43.89	37.96	42.80	42.39	42.51	42.11	40.23	47.49	46.63	48.92	
24	132						80.77	79.66	76.10	72.95	72.23	65.34	64.09	60.08	66.40	66.78	67.12	63.67	62.50	64.09	65.30	64.34	64.81
25	106						80.66	78.35	74.43	70.57	67.92	60.99	55.57	53.58	58.40	64.15	57.92	55.52	45.55	56.42	59.67	19.57	3.20
26	136						66.07	55.70	41.54														

X. Sequence of the transition matrix types of defoliation for Norway spruce

Permanent plot:	01	02	04	05	06	07	08	09	10	11	12	13	14	15	16	18	19	20	21	22	23	24	25
76:										E	E	E	E	E									
77:										E	E	E	E	E									
78:										E	E	E	E	E									
79:										E	E	E	E	E									
80:	A	E			E	E	N	H		H	H	C	E	H									
81:	E	E	F	F	E	E	F	F	F	F	F	C	F	F	C	C	C	C	F	C	F	A	C
82:	F	A	A	A	A	A	A	A	F	F	F	F	F	F	C	C	C	C	H	F	C	C	F
83:	A	G	C	A	G	A	B	C	F	C	F	F	G	F	C	C	C	C	C	A	E	C	A
84:	B	A	B	B	I	B	B	C	C	B	C	F	F	A	C	I	B	F	B	C	F	F	F
85:	B	G	C	C	G	B	F	B	A	B	C	A	A	G	B	B	B	C	B	C	F	A	A
86:	B	B	B	B	B	B	N	B	B	B	B	B	B	B	B	B	B	B	B	C	F	B	B
87:	H	G	B	B	B	B	M	B	D	B	B	B	A	C	B	D	B	B	B	C	I	C	B
88:	C	B	C	B	B	B	M	C	B	B	B	B	B	B	B	M	B	C	B	L	B	B	B
89:	G	B	B	B	L	B	N	B	B	B	B	G	B	B	B	B	B	B	B	B	B	B	B
90:	O	B	B	B	B	B	M	B	B	B	B	B	B	C	C	B	B	B	B	B	B	A	B
91:	D	B	B	B	G	B	C	C	B	C	C	B	B	B	B	C	B	B	C	B	B	B	C
92:	D	B	C	B	B	B	J	B	B	B	B	B	B	B	B	B	B	B	B	B	B	C	B
93:	D	A	B	B	B	B	B	B	C	C	C	B	B	O	A	C	B	B	B	B	B	B	B
94:	B	A	C	B	B	K	B	B	B	B	C	K	B	O	G	B	A	B	B	B	A	B	B
95:	B	E	B	B	B	A	B	G	B	B	G	B	B	M	G	O	B	B	B	C	B	H	O
96:	B	E	C	B	B	E	B	B	B	B	B	B	B	G	O	B	H	C	A	B	B	K	O

Statistical evaluation of the difference between the matrix type sequence in the single plots and common (average) sequence during 1981/1982 to 1996/1997 ($f = 15$, $\chi^2[95\%] = 24.996$; $\chi^2(A)$ - reference set of all plots, $\chi^2(B)$ - reference set of all plots without the comparative plot)

TVP:	$\chi^2(A)$	$\chi^2(B)$	21:	17.80	30.70	25:	13.23	16.65
01:	37.74	57.56	16:	20.27	8.35	14:	11.25	15.00
15:	35.70	51.59	13:	16.42	22.76	12:	9.51	11.82
08:	31.45	45.36	19:	14.54	22.66	09:	8.38	10.50
24:	27.47	42.86	22:	14.57	22.09	04:	5.50	6.37
23:	25.26	40.38	02:	17.29	21.46	05:	5.04	6.11
06:	23.11	34.55	07:	14.98	20.19	20:	4.53	5.29
18:	23.21	33.74	10:	12.86	17.08	11:	3.79	4.43



6. Evaluation of the transition matrix types by spruce according to a simulation model (ten year period of the change of initial stand using average transition matrix of the respective type). Initial stand consists of 100 trees of each defoliation classes 0 to 4

the period of observation was characterised by continuous deterioration of stand health (Fig. 8). There was a change in this trend in 1986–1987 (see the changes in average defoliation and frequency of the types of transition matrices), when deterioration slowed down or on the contrary, the health state of beech stands started improving. The improvement was interrupted in some years with increasing defoliation or dieback of other trees.

With respect to the effect of transition matrices of the separate types on a ten-year development model of forest stand (with 100 individuals in each class of defoliation 0 to 4 at the beginning of modelled period; see Fig. 9) types A to I were divided into the following groups:

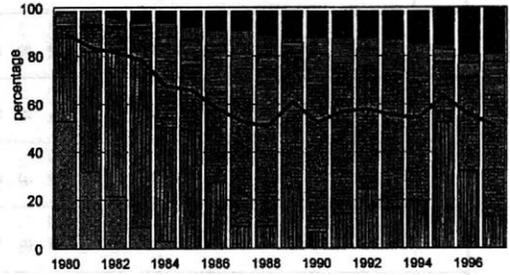
- improvement or stabilisation of stand state – types H, D, or C (applying to maintenance of stand state);
- deterioration of stand state (increasing defoliation or tree diebacks) – types F, E, B, A, I.

It is possible to distinguish three groups of plots (PRP designated by bold digits represent stands in which no statistically significantly different sequence of the types of transition matrices was determined in comparison with the whole set of remaining PRP):

- Stands with low defoliation, without any continual several-year period of health state deterioration – PRP 2, 29 (mixed spruce-beech stands).

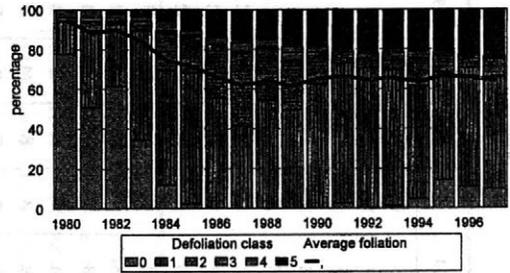
PRP 27

Fagus sylvatica



PRP 31

Fagus sylvatica

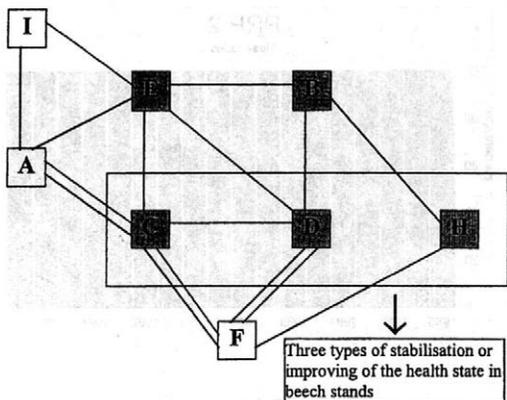


7. Trend of average foliage and particular defoliation class of beech on plots 27 and 31

- Stands with continuous deterioration of health state (increasing defoliation and tree diebacks) – PRP 6, 7, 8, 9 (mixed spruce-beech stands); maximum average defoliation was about 35% in the whole period of observation, maximally 30% of trees died.
- Stands with great damage and a continual several-year period of health state deterioration – PRP 1, 31, 32 (mixed stands) and 27, 28, 30 (beech stands); maximum average defoliation was about 35–45% in the whole period of observation, lower percentage was exceptional (PRP 30), 17–23% of trees died within 17 years, sometimes diebacks were concentrated in a single period (36% of trees died on PRP 1 within a year).

XI. Dynamics of foliage of the beech stands (average values in per cent; totally defoliated trees are included)

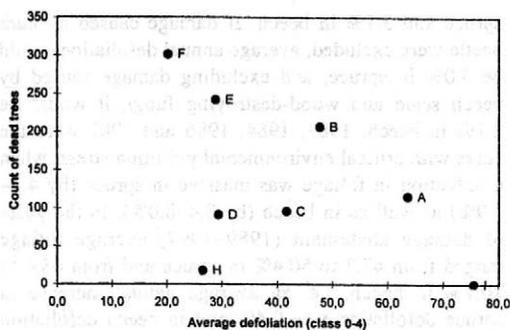
Permanent plot	Number of trees	Year																	
		1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997
27	275	89.26	83.45	81.34	78.51	67.48	65.90	58.30	52.90	50.44	60.88	52.35	56.85	57.78	55.25	54.18	63.36	55.68	51.03
28	137	95.11	90.83	89.01	84.27	72.41	71.57	66.28	63.94	64.93	69.12	63.58	68.61	69.05	67.55	60.66	69.05	61.81	57.63
29	69	99.45	94.91	96.27	90.00	84.64	83.64	80.18	79.10	80.18	80.91	80.55	80.09	81.48	80.27	86.72	79.73	72.94	75.18
30	82	94.39	89.02	89.70	84.57	75.49	73.84	71.40	70.43	67.56	69.45	71.71	69.94	68.78	67.44	72.01	67.16	73.29	69.88
31	85	94.76	88.41	89.94	84.18	74.47	71.65	66.18	61.82	64.06	61.47	65.24	66.41	64.59	65.53	63.47	66.76	65.94	64.06
32	105	92.14	84.81	85.44	79.03	68.50	68.01	62.38	61.02	63.88	66.36	69.46	67.77	63.16	64.71	65.44	65.44	63.60	58.91



8. Graph of the main sequences of the transition matrix types by *Fagus sylvatica*. Types in grey fields have a frequent repetition

Mixed stands

Tab. XIII shows foliage dynamics of spruce-beech stands, and it is plotted in Fig. 10 using an example of PRP 2. The average regional foliage of spruce-beech stands was 86.0% in spruce and 95.3% in beech in



9. Evaluation of the transition matrix types by beech according to a simulation model (ten year period of the change of initial stand using average transition matrix of the respective type). Initial stand consists of 100 trees of each defoliation classes 0 to 4

1980, and only 47.3 and 63.4% in 1997, i.e. an average annual increase in defoliation of 2.3% in spruce and 1.9% in beech. In the period of great damage (1981–1988), average foliage ranged between 50.5% and 81.4% in spruce and 68.2% and 89.5% in beech, i.e. an average annual increase in defoliation was 4.1% in

XII. Sequence of the transition matrix types of defoliation for beech

Permanent plot	01	02	06	07	08	09	27	28	29	30	31	32
80:	A	C	C	C	A	C	B	A	C	B	A	B
81:	B	C	C	C	F	C	B	H	C	E	E	C
82:	B	C	C	C	C	C	E	B	C	E	A	B
83:	B	C	F	F	C	C	E	E	C	E	I	E
84:	E	C	C	C	E	H	B	B	C	C	E	B
85:	C	C	C	C	E	B	D	B	C	C	C	B
86:	C	C	D	D	C	B	D	D	D	D	D	D
87:	D	C	D	D	D	D	D	D	D	D	D	D
88:	F	A	E	F	D	E	D	D	D	D	D	D
89:	C	C	E	C	A	D	D	D	D	E	E	D
90:	F	F	D	E	D	D	D	D	D	C	A	D
91:	D	D	C	D	F	F	D	B	D	C	E	D
92:	D	C	B	E	D	D	D	B	D	H	D	E
93:	C	C	D	D	D	D	D	E	H	H	C	B
94:	C	C	F	F	D	D	D	H	F	H	C	B
95:	D	C	H	C	D	H	D	E	C	F	F	E
96:	C	C	B	C	D	C	D	D	A	A	C	C

Statistical evaluation of the difference between the matrix type sequence in the single plots and common (average) sequence during 1981/1982 to 1996/1997 ($f = 16$, $\chi^2[95\%] = 24.296$; $\chi^2(A)$ – reference set of all plots, $\chi^2(B)$ – reference set of all plots without the comparative plot)

Permanent plot:	$\chi^2(A)$	$\chi^2(B)$
31:	20.30	38.04
09:	18.34	31.62
30:	19.83	31.01
08:	17.54	30.22
06:	17.09	27.88
28:	15.78	24.74

Permanent plot:	$\chi^2(A)$	$\chi^2(B)$
01:	14.99	22.38
02:	8.93	20.95
07:	11.30	18.03
29:	9.31	15.12
32:	8.57	11.80
27:	6.83	9.97

spruce and 3.1% in beech. If damage caused by bark beetle were excluded, average annual defoliation would be 3.0% in spruce, and excluding damage caused by beech scale and wood-destroying fungi, it would be 2.1% in beech. 1981, 1984, 1986 and 1987 were the years with critical environmental pollution stress, when a reduction in foliage was massive in spruce (by 4.7–7.9%) as well as in beech (by 3.4–8.0%). In the years of damage abatement (1989–1997) average foliage ranged from 47.3 to 50.4% in spruce and from 63.4 to 70.7% in beech, i.e. an average annual increase in spruce defoliation was 0.4% and in beech defoliation 0.9% (it would be 0.2 and 0.4%, respectively, if the effects of biotic pests were excluded). The effect of biotic pests was very harmful on PRP 1 (bark beetle attack to spruce-trees and attacks of beech scale and wood-destroying fungi to beech-trees) – see Vacek (1995).

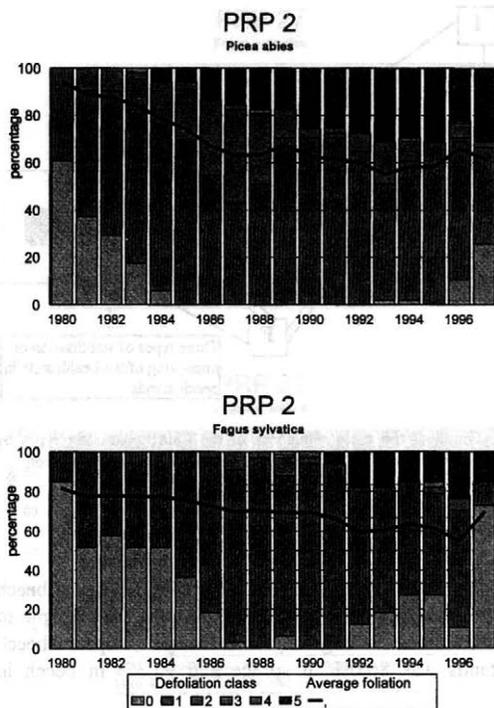
General comments on defoliation trends

Six plots with mixed stands were used to compare damage of two main species under identical conditions (cf. Vacek, 1989b). Such a low number of plots is not obviously sufficient for this purpose. In general, no substantial differences were observed between the species growing in mixed stand or in pure stand. Therefore the above evaluation by transition matrices was made with respect to tree species not to the group of plots.

There was a decrease in standard deviation of defoliation (STD) in *Picea abies* on the separate plots. Low STD values can be insignificant if the number of trees is small. The zero STD values cannot be taken as actually zero ones because defoliation variability within a classification class and variability neglected due to classification error should be taken into account. Only a homogeneous group of resistant individuals will survive stress impacts. First, STD will increase to ca. 20%, then it will decrease. The stand appears to have a chance of survival when STD decreases to 5–10% and the number of living trees is sufficiently high at the same time. A similar mechanism of decrease in standard deviation of defoliation was observed on some plots with beech. It was more difficult to be discerned because the level of stress to this species was likely relatively lower (higher tolerance of the species).

Damage to *Sorbus aucuparia* was evaluated on two plots (PRP 27 and 29) to obtain additional data. Processes of damage seem to be similar to those in beech.

With regard to the management practices, damage to individual trees or a decrease in their foliage were investigated in relation to tree classes as it can be very important for evaluation of changes in stand structure (Vacek, Lepš, 1995). The degree of defoliation can be considered as a criterion of impacts of external factors: crucial importance is ascribed to air pollutants



10. Trend of average foliage and particular defoliation class of spruce and beech on plot 2

in an air-pollution area while a tree class describes growth constitution resulting from natural development of trees in the framework of cenotic relationships in forest stand (Vacek, 1987b).

In the period of the first signs of damage (1976–1980), i.e. in forest stands almost free of damage with declining tree class, average foliage is mostly reduced, probably as a result of diminishing light intensity. This relation is most explicit at differentiated age structure, and it is better perceptible in spruce than in beech. This state is disturbed as a result of pollutant impacts in the period of great damage (1981–1988), and foliage in tree classes assumes the values conditioned by a large complex of external and internal factors. The degree of defoliation is related to tree class but the closeness of this relation is associated with the degree of stand damage. Not only subdominant and co-dominant trees but also dominant trees suffer from defoliation with an increasing intensity of pollutant impacts (cf. Vacek, 1986b).

Dominant spruce-trees had lower foliage than co-dominant ones on PRP 3, 11, 12, 21 and 23 with great pollution stress as early as in 1984. A similar situation was observed in spruce-trees of spruce-beech stands with pollution stress (PRP 6). This relatively small difference substantially increased as a result of growing pollution stress over the years of observation.

XIII. Dynamics of foliage of the mixed stands (average values in per cent; total defoliated trees are included)

Permanent plot and species (sm – Norway spruce, bk – beech)	Number of trees	Year																	
		1980	1981	1982	1983	1984	1985	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997
1 sm	31	84.19	77.74	79.03	77.10	73.87	69.35	63.39	50.65	50.48	36.45	17.42	17.50	15.32	16.09	17.26	13.71	12.71	10.96
1 bk	47	91.60	84.79	83.94	82.77	77.45	76.38	72.45	64.79	69.15	67.23	63.87	38.15	29.36	36.74	35.85	40.00	40.43	39.26
2 sm	51	94.11	88.63	87.45	84.12	78.53	74.12	67.06	63.04	63.33	67.65	62.55	61.67	60.10	55.10	58.33	58.34	58.63	61.76
2 bk	33	97.73	92.73	93.64	92.73	92.73	90.45	87.12	83.64	84.39	82.87	83.18	79.85	71.36	72.27	76.21	73.64	78.03	83.03
6 sm	51	93.43	91.08	86.47	78.82	62.55	57.25	43.33	38.53	40.78	35.49	33.53	36.90	35.20	33.50	32.70	37.50	38.70	37.40
6 bk	158	95.54	90.60	89.40	86.46	77.22	73.20	68.99	58.26	60.63	57.75	58.38	61.51	61.32	57.23	54.87	59.94	59.21	50.31
7 sm	30	96.50	95.50	90.67	82.50	74.67	72.67	65.17	65.17	70.33	69.83	69.17	67.00	69.67	69.67	69.17	74.17	73.18	72.83
7 bk	215	94.98	90.30	88.72	86.81	81.88	79.00	74.51	65.49	75.84	76.23	73.88	72.58	74.05	67.56	71.98	76.70	76.49	73.79
8 sm	9	58.33	53.89	48.89	51.11	51.11	48.33	46.11	31.67	31.67	36.67	29.47	38.33	45.56	34.44	38.89	47.78	45.56	42.77
8 bk	97	96.24	91.86	92.01	85.72	77.11	77.53	75.52	68.56	66.91	69.69	64.02	65.21	65.67	62.89	61.65	68.25	71.50	71.75
9 sm	57	89.74	81.49	76.84	73.07	66.75	67.11	65.26	53.85	56.93	56.32	52.11	55.53	57.37	54.56	53.77	60.08	56.75	57.11
9 bk	57	95.88	86.84	84.82	84.04	78.95	80.79	78.42	68.51	65.09	70.35	64.91	65.51	62.63	59.04	59.83	64.12	70.61	62.28

The level of vegetation changes was expressed by indexes calculated as a sum of variances of ordination scores for one to four first axes of DCA (calculated for herb layer). The level of stand damage at a locality was expressed as a proportion of trees of a given species that died in 1985–1997, or as maximum determined average defoliation of trees of a given species (in case more than 75% of trees died, maximum defoliation equalled 100%). Relations between these variables were tested by indexes of linear correlation (Tab. XIV).

Data show that the degree of damage can be influenced by the level of change in herb layer structure in the ecosystem; such a result was expected. But the character of this effect was very surprising – changes in herb layer were smaller at greater stand damage. There were relatively small changes in vegetation even in the case of the plots with totally destroyed tree layers (cf. Vacek et al., 1996c). This fact should be analysed in greater detail to avoid its misinterpretation. It is necessary to take into account the following facts and assumptions:

1. Forest stand without cultural treatments that does not suffer any damage increases its density in the process of growth, and this can be a cause of great changes in herbaceous vegetation.
2. It is evident that the herbaceous vegetation of spruce stands that suffered relatively severe damage is similar to clear-cut vegetation after the tree layer was destroyed by decline or felling.
3. Mechanisms leading to damage of the woody component of an ecosystem rarely act as factors of herbaceous vegetation succession.
4. It is to assume that mountainous forest ecosystems are continually disturbed by natural factors; a slightly open character of tree layer is maintained in this way.

The analyses were also aimed at a relation between the level of change in vegetation (in this case expressed as a shift of the plot in DCA four-dimensional space at the beginning of the period of observation and at its end) and height above sea level. The relation was statistically insignificant in general, but a trend of diminishing level of change in vegetation was observed with increasing height above sea level. This trend was least perceptible in beech and mixed stands ($r = -0.328$) while in spruce stands (group of plots designated by Pi) the relation was significant at a 5% level ($r = -0.539$).

A similar attempt to describe the relation between vegetation damage and change was made for moss layer. All results were statistically insignificant, but there was a trend of increase in the level of change in moss layer in forest stands with greater damage (with respect to variability calculated on the basis of the first four or three ordination axes), especially in spruce stands. The trend was more perceptible if the proportion of dead trees was confronted with maximum defoliation

CONCLUSION

Three typical periods were distinguished with respect to the dynamics of health state of the stands (mainly defoliation) during years 1976–1995. Moderate changes in foliage were determined in the period of the first signs of damage (1976–1980) (average annual increase in defoliation in spruce stands was 0.4%). A perceptible increase in defoliation occurred in the period of great damage (1981–1989) (3.0–4.4% per annum in spruce and 3.1–3.4% per annum in beech). Stabilisation or improvement of stand state were observed in the period of damage abatement (1987–1997) in those stands that were not attacked by pests, mainly by bark beetle.

Average defoliation was increasing in the period of the first signs of damage as the tree class was declining. This relationship was fully disturbed as a result of pollution stress in the period of great damage, and not only subdominant and co-dominant trees were affected by defoliation but also dominant ones. When stand damage is evaluated, it is to be aware of the fact that pollution stress is regularly accompanied by higher sensitivity to some biotic and abiotic factors, and this is explicitly reflected in accelerated (even many times) dynamics of damage and in subsequent stand destruction. The highest tolerance was observed in beech stands, spruce-beech stands showed lower tolerance, and spruce stands were the least tolerant of all.

Trends of stand health can be described by transition matrices. If permanent observation of stand defoliation is used to define up-to-date transition matrices, it is possible to estimate further trends of forest ecosystems quite exactly on the basis of a constructed model. Transition matrices can also be used to identify developmental stages of stand damage and regeneration. The knowledge will be applied to plan forest management measures in forest stands.

XIV. Linear correlation coefficients for variables of change in the herb layer (sum of variances of ordination score for the first one to four DCA axes) and degree of the tree layer damage. All values are not significant except those designated by * ($\alpha \leq 5\%$) and *** ($\alpha \leq 0.5\%$)

Sum of variance of the DCA axes	Maximal defoliation (%)	Share of dead trees (%)
<i>Picea abies</i>		
[1]	+0.1110	+0.1157
[1]+[2]	-0.3362	-0.1534
[1]+[2]+[3]	-0.3084	-0.1287
[1]+[2]+[3]+[4]	-0.2862	-0.1581
<i>Fagus sylvatica</i> (mixed stands included)		
[1]	-0.7572***	-0.2904
[1]+[2]	-0.7517***	-0.2446
[1]+[2]+[3]	-0.6612*	-0.3521
[1]+[2]+[3]+[4]	-0.5346	-0.4213

The evidence of forest ecosystem destruction and regeneration based on tree component or stand health evaluation within 21 years was complemented by data on the dynamics of herb and moss layers. The analyses provided information acquired by the study of ground vegetation as an indicator responding to changes in light conditions in the process of defoliation and natural development of stand, limiting the potential of regeneration. The cyclical character of some species that influence, and even prevent in many cases, natural or artificial regeneration (e.g. grass communities with dominance of *Calamagrostis villosa* and high ferns *Athyrium distentifolium*) was confirmed. Reforestation should not be undertaken in the peak developmental stages of these communities. On the other hand, it is worth waiting for about 3–5 years before dominance and sociability are reduced, i.e. until more favourable conditions for reforestation are created. This will largely decrease a risk of reforestation failure as a result of weeds and bracken.

It is to state at the very end of the paper that regeneration targets and methods should be based not only on relative tolerance of tree species in the types of stands but also on their ecological valence, ground layer state and economic possibilities while respecting the present functional importance of the site.

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STAV LESNÍCH POROSTŮ NA VÝZKUMNÝCH PLOCHÁCH V KRKONOŠÍCH V LETECH 1976–1997

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Podle dynamiky zdravotního stavu, zejména pak defoliace, byla v průběhu sledovaných let 1976–1997 vylišena tři charakteristická období. V období prvních příznaků poškození (léta 1976–1980) došlo k nepatrné dynamice olistění (průměrné roční zvýšení defoliace ve smrkových porostech 0,4 %). V období silného poškození (léta 1981–1989) byl zaznamenán výrazný vzrůst defoliace (3,0–4,4 % ročně u smrku a 3,1–3,4 % ročně u buku). V období ústupu poškozování (léta 1989–1997) naopak došlo ke stabilizaci až zlepšení stavu ve sledovaných porostech, které nebyly napadeny škůdci, zejména kůrovcem.

V období prvních příznaků poškození s ustupující stromovou třídou vzrůstala průměrná defoliace. Vlivem imisí v období silného poškození byl tento vztah zcela porušen a defoliace postihovala nejen podúrovňové a úrovňové stromy, ale i jedince nadúrovňové. Při posuzování poškození porostů je nutné brát v úvahu to, že imisní stres je zpravidla provázen zvýšenou citlivostí vůči některým biotickým a abiotickým činitelům, což se mnohdy výrazně (až několikanásobně) projevuje v urychlení dynamiky poškození a následné destrukci porostů. Největší toleranci projevily bukové, nižší smrkobukové a nejnižší smrkové porosty.

Vývoj zdravotního stavu porostů lze dobře popsat pomocí přechodových matic. Při využití permanentního sledování defoliace porostů pro definici aktuálních přechodových matic lze pomocí vytvořeného modelu poměrně přesně odhadnout další vývoj sledovaných lesních ekosystémů. Přechodové matice je možné užít i pro rozlišení vývojových etap poškození a regenerace porostů. Tyto poznatky jsou pak základem pro plánování hospodářsko-úpravnických opatření v porostech.

Poznatky o postupu destrukce i regenerace lesního ekosystému podle hodnocení stromové složky, resp. zdravotního stavu porostů v průběhu 21 let doplňovaly údaje o dynamice bylinného a mechového patra. Z těchto analýz vyplynula i řada poznatků limitujících možnosti obnovy, vyplývajících ze sledování přízemní vegetace jako indikátoru reagujícího na změnu světelných poměrů v průběhu defoliace a přirozeného vývoje

porostu. Bylo zjištěno, že u řady druhů, které ovlivňují a mnohdy až znemožňují přirozenou či umělou obnovu (např. travní společenstva s dominancí *Calamagrostis villosa* a vysokých kapradin *Athyrium distentifolium*), dochází k výrazné cykličnosti. Realizovat obnovu není vhodné zejména v kulminační fázi rozvoje těchto společenstev. Naopak se vyplatí počkat asi 3–5 let, než dojde ke snížení jejich dominance a sociability, tj. k vytvoření podmínek pro obnovu mnohem příznivějších. Značně se tím snižuje riziko obnovního neúspěchu daného vlivem buňeně.

V závěru lze konstatovat, že při volbě obnovních cílů a postupů je nutné vycházet nejen z relativní tolerance dřevin v jednotlivých typech porostů, ale i z jejich ekologické valence, stavu přízemního patra a hospodářských možností při respektování současného funkčního významu stanoviště.

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LITTER DECOMPOSITION AND SOIL NITROGEN DYNAMICS (1995-1996) IN THE KRKONOŠE NATIONAL PARK

ROZKLAD OPADU A DYNAMIKA PŮDNÍHO DUSÍKU V KRKONOŠSKÉM NÁRODNÍM PARKU (1995-1996)

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ABSTRACT: This paper presents the results of experiments in the Krkonoše National Park to study decay dynamics of organic matter and nitrogen dynamics in the ectorganic (holorganic) layer of forest soils. Hypotheses are formulated about changes in organic matter upon decay as well as the importance of factors involved. Results are discussed and compared with the literature. Main conclusions from the litterbag experiments are that the results conform well with those generally reported in the literature, and that Ca and Mg leach faster in forests than on clear-cuts, probably due to elevated levels of acidifying substances in canopy throughfall, while more N is immobilised in forests than on clear-cuts, probably due to higher N contents in canopy throughfall. The study of N dynamics revealed that relations between the properties of organic layers, N pools and N transformation rates were poor and that both net nitrogen mineralisation and nitrification rates were highly variable. Possibilities for their spatial interpretation were limited, so that only an estimation of the general level of net nitrogen mineralisation and nitrification was possible.

organic matter decomposition; dynamics of soil nitrogen; humus layer

ABSTRAKT: Příspěvek uvádí výsledky experimentů prováděných na území Krkonošského národního parku týkající se dynamiky rozkladu organické hmoty a dynamiky dusíku v humusové vrstvě lesních půd. Byly formulovány hypotézy o změnách organické hmoty v průběhu rozkladu a o významu sledovaných faktorů. Hlavní výsledky „litterbag“ experimentů byly v dobré shodě s literárními údaji a týkají se rychlejšího uvolňování Ca a Mg v lesní půdě proti půdě pasek, což pravděpodobně souvisí se vzrůstajícím obsahem kyselých složek ve srážkách pod korunami. Více dusíku je imobilizováno v lesích proti pasekům, což je pravděpodobně způsobeno vyšším obsahem dusíku v podkorunových srážkách. Studium dynamiky dusíku ukazuje, že vztah mezi vlastnostmi organických vrstev půdy, zásobou dusíku a rychlostí transformace dusíku je slabý a že mineralizace dusíku a rychlost nitrifikace byly vysoce proměnlivé. Možnosti interpretace jejich variability v území byly omezené, protože jsou uskutečnitelné pouze odhady obecné hladiny pro čistou mineralizaci dusíku a nitrifikaci.

rozklad organické hmoty; dynamika půdního dusíku; humusová vrstva

INTRODUCTION

As part of the research programme *Restoration of Forest Ecosystems in the Krkonoše National Park*, the Department of Physical Geography and Soil Science of the University of Amsterdam started field surveys of vegetation and soil in 1992. Rehabilitation practice in the Krkonoše Mts. deals with the reforestation of extensive clear-cuts, and the propagation of natural succession and establishment of pioneer species with an ameliorative effect. The monitoring of litter decomposition

and soil nitrogen dynamics should provide a basic insight into soil processes in relation to these aspects.

The assessment of N transformations is a prerequisite for proper evaluation of nutrient availability and acidity in forest and clear-cut ecosystems. Working hypotheses were:

1. Rates of net nitrogen mineralisation and nitrification depend on site conditions (including temperature and soil moisture, see Emmer, Tietema, 1990; Tietema et al., 1992a; Clarholm et al., 1981).
2. During stand development these rates may change, mainly due to changes in element stocks (depending

on litterfall, throughfall, and nitrogen uptake) and environmental factors, such as micro-climate (Carlyle, 1986; Emmett et al., 1993).

Organic matter composition is a key-parameter in the study of changes in chemical soil properties and humus forms. The in-situ incubation of fresh litter in litterbags is an appropriate method to study the first stages of decomposition (related to the L and F1 horizons), providing information on the potential contribution of litter types to the nutrient cycling. Litterbag experiments cannot cover the dynamics of deeper organic horizons (F2 and H) which may take up to several decades to develop (Emmer, Sevink, 1994). Working hypotheses were:

1. Clear-cuts have conditions more conducive to litter decomposition and will therefore show higher rates of litter mass loss and element release.
2. Differences between the sites are due to differences in soil temperature.
3. Litters with higher element concentrations decompose faster and release elements at higher rates than those with low element concentrations.

In this paper, the results of the measurements will be presented and discussed.

MATERIALS AND METHODS

SELECTION OF PLOTS (GENERAL)

Plots were selected along the gradient of forest damage and at various altitudes in the Krkonoše National Park. Tab. I gives an overview of selected plots. The plots near Kamenec (Kam), Vosecká bouda (VoB), Fučíkovy boudy (FuB) and Černá hora (CeH) represent a gradient from severely damaged stands in the north-west to less damaged in the south-east of the National Park. These plots

were located on Histosols, at Vosecká bouda and Fučíkovy boudy on Histosols alternating with peaty Podzols.

The altitudinal zonation of the Krkonoše Mts. comprises (with increasing altitude) Dystric Cambisols, Podzols and Histosols. Lithic Leptosols (block fields) are mainly found at higher altitudes, but are spread throughout the area at lower altitudes as well (Emmer, 1996). These sites have particular importance because of their shallow soils, poor site conditions and erosion hazard. Vosecká bouda and Fučíkovy boudy represent Histosols as well as (peaty) Podzols, while Cambisols and Leptosols are represented by Svätý Petr (SP) and Čertova struha (CSt), respectively. The latter plot is a FGMRI Jíloviště-Strnady, RS Opočno monitoring plot (No. 18). To prevent disturbing this plot, soil and vegetation inventories were done in an adjacent plot located nearer towards Důl Bílého Labe (DBL). Except for the plot Černá hora, which was located on Schist, the plots were on Granitic parent material. Because of the limited rooting depth in combination with thick peat layers and small slope angles a significant effect of parent material on nutrient cycling and forest vitality on these plots was regarded unlikely.

LITTERBAG EXPERIMENTS

General

The method utilises polyethylene bags with a mesh size of 1 mm, containing several grams of litter material. These bags are installed in the field on top of the organic layer and are retrieved at various time intervals. An appropriate length for the experiment is about 2–5 years, depending on biotic and abiotic conditions for decomposition. Details on this method are provided

I. General information on selected plots

Location	Plot code	Forestry map*	Forest	Clear-cut	Health state*
Kamenec	Kam	(H)1201C17	<i>Picea</i>	–	–
Vosecká bouda	VoB	(H)1511C17	<i>Picea</i>	–	+/-
Fučíkovy boudy	FuB	(V)2105A10	<i>Picea</i>	+	+
Černá hora	CeH	(M)3105C15	<i>Picea</i>	–	+/-
Svätý Petr	SP	(V)2240A8	<i>Picea</i>	+	+
Čertova struha	CSt/DBL	(V)2211B14	<i>Picea</i>	+	+/-
Pevnost	Pev	(V)2111D9	<i>Fagus</i>	–	+
	Soil type	Substrate	Aspect	Altitude (m)	Slope (°)
Kamenec	Histosol	Granite	WNW	1150–1200	7–10
Vosecká bouda	Histosol/Podzol	Granite	WSW	1200	3–12
Fučíkovy boudy	Histosol/Podzol	Granite	SSE	1200	5–7
Černá hora	Histosol	Schist	S	1150–1200	2–7
Svätý Petr	Cambisol	Schist	NNW	850–925	17–34
Čertova struha	Leptosol	Granite	SE	1150	18
Pevnost	Cambisol	Granite	SSE	850	20

*) Applies to the forest stand

e.g. by Berg (1986), Gosz et al. (1973), Herlitzius (1983), Lousier, Parkinson (1978), Tietema (1993), Wieder, Lang (1982).

Sampling

Only leaf and needle litter from major plant species was included in the experiments. Major tree species are *Picea abies* and *Fagus sylvatica* (stand forming), and *Sorbus aucuparia* and *Betula pendula* (pioneers, but potentially stand forming as well). Of the understorey vegetation, the main grass species (*Calamagrostis villosa*) and the major shrub species (*Vaccinium myrtillus*) were selected. *Sorbus aucuparia* litter included the leaves and the main leaf stem.

Sampling intervals were chosen to obtain more detailed information on weight loss and elemental composition during the early stages of decay, when the highest rates of change can be expected, i.e. during the first year. Due to the lack of other litterbag experiments carried out in the area in the past, it is not certain whether the best choice of intervals was made. Triplicate sampling and chemical analyses were done throughout the experiment to reduce the effects of spatial variance within the sub-plots. Because of the high variation in drainage conditions in the peat-podzol complexes of VoB and FuB, three moisture states (wet, dry and intermediate) were distinguished in which the replicates were sampled.

VoB was included for litterbag experiments because of the extensive monitoring of deposition and soil water chemistry. A stand with dominant *Fagus* near Pevnost (Pev) was selected for comparison of beech litter decay.

As the accumulated effort connected to this set-up would exceed funding, some of the experiments were simplified by increasing sampling intervals, and reducing the number of litter types per plot.

Just before the last sampling on 12 October 1996, the CSt forest plot was clear-felled because of a bark beetle infestation. The litterbags had to be transported to an adjacent area (approximately 30 m away) and reinstalled. The *Calamagrostis* bags were covered with saw-dust and could no longer be used.

Sample processing and chemical analyses

Litter samples were collected by shaking trees and collecting the fallen needles or leaves, by cutting dry grass or by hand-picking shrub leaves, in October 1995. Air-dried samples of about 5–11 g were put in polyethylene bags with a mesh size of 1 mm. *Picea* needles were slightly moistened with demineralised water to prevent any loss during transport to the field. The bags were attached to the surface of the organic layer in the selected plots, using wooden pins, on October 30, 1995.

Sample processing and chemical analysis were done in the Laboratory at FMGRI, RS Opočno. After removal from the field the incubated samples were dried at 70 °C for the determination of the remaining weight.

Loss-on-ignition was determined after heating at 100, 200, 300 and 400 °C for 1 hour, and 500 °C for 4 hours. The carbon content of organic matter was estimated as 50% of the ash-free material.

After grinding, sub-samples were digested repeatedly with a mixture of concentrated H₂SO₄ and 30% H₂O₂ at 300 °C during 30 minutes, until a colourless liquid remained. P and base elements were analysed photometrically. Total N was analysed by means of a Kjeldahl procedure.

NITROGEN DYNAMICS

General

N mineralisation and nitrification can be assessed quite satisfactorily using a simple technique with PVC tubes, as described by Tietema et al. (1990) and Tietema, Verstraten (1991). On each sample date, pairs of intact soil cores within a 10 cm distance of each other were sampled utilising PVC tubes with a 7 cm internal diameter, length of 25 cm, and one sharpened end. One core was taken to the laboratory immediately for analysis (reference sample), while the other was left in the field for several weeks for incubation (incubation sample). During incubation the tube was closed at the top and bottom to prevent loss or gain of nitrogen solutes. Aeration was through two 14 mm holes in the top of the tube.

The difference between the nitrate concentrations of the reference and incubation samples thus obtained was taken as a measure for the rate of net nitrification. Net nitrogen mineralisation was calculated as the difference between the inorganic nitrogen (ammonium + nitrate + nitrite) concentration of both samples, related to the incubation period.

Sampling

Sample locations included forest and clear-cut sub-plots of FuB, SP and CSt. These plots represent the 3 major soil conditions distinguished: Histosol-peaty Podzol complex, Cambisol and Leptosol, respectively. The health state of the *Picea* stands was relatively good. Additionally, VoB and 6 plots which represent an age series of *Picea* stands from 0 (a recently replanted clear-cut) to 91 years old (Tab. II) were selected. The age sequence was located in a Leptosol/Podzol complex. Triplicate samples were taken in all plots. Because of the high variation in drainage conditions in the peat-podzol complexes of VoB and FuB, three moisture states (wet, dry and intermediate) were distinguished in which the replicates were sampled. In the young stands of the age series, samples were taken right under the canopy, and on the edge of the canopy, and outside the canopy.

The first incubation period was between 30 October 1995 and 6 June 1996, including a period of snow cover

II. Nitrogen incubation experiments: plot information

Plot	Forestry map	Age (year)	Soil type	Differentiation*	Altitude (m)	Aspect	Temp. rec.
<i>Picea</i> forest and clear-cut							
VoB	1511C17		Histosol/Podzol	W/I/D	1200	WSW	
FuB	2105A10		Histosol/Podzol	W/I/D	1200	SSE	+
SP	2240A8		Cambisol	-	850-925	NNW	+
CSt	2211B14		Leptosol	-	1150	SE	+
Age series (only forest plot)							
211A1	2211A1	0	Leptosol/Podzol	-	1070	W	
208A2	2208A2	18	Leptosol/Podzol	U/E/O	1060	W	
202G3	2202G3	26	Leptosol/Podzol	U/E/O	1090	SW	
209A4	2209A4	36	Leptosol/Podzol	U/E/O	1150	WSW	
209B7	2209B7	68	Leptosol/Podzol	-	1100	W	
209B9	2209B9	91	Leptosol/Podzol	-	1100	W	

* W: Wet; I: Interm.; D: Dry; U: Under canopy; E: Under edge of canopy; O: Outside canopy

(until the beginning of May 1996) and one subsequent month (totalling 220 days). The next 3 incubation periods were: 6 June - 8 August (63 days), 8 August - 11 September (34 days) and 11-23 September 1996 (12 days).

Sample processing and chemical analyses

Sample processing and chemical analysis were done by the Laboratory at FMGRI, RS Opočno. The organic layer was separated from the mineral part of the soil core, and subsequently large wood particles were removed. To assure a proper sub-sampling, sample processing proceeded with material sieved to pass a 5 mm mesh. Dry weights were deduced from moist sample weights and gravimetric moisture contents of sub-samples. Total nitrogen (without nitrate and nitrite) was determined by means of a Kjeldahl procedure. The organic samples were extracted over-night with 1% K_2SO_4 (25 : 200 w/v) and analysed for dissolved organic nitrogen, ammonium, nitrate and nitrite. The pH was determined in a 1 : 5 w/v 1 M KCl extract.

Dissolved organic nitrogen (DON) was analysed as ammonium after elimination of nitrate and nitrite, by means of a modified Kjeldahl procedure involving mineralisation in boiling sulphuric acid with HgO and Se as catalysts, and correction for the original ammonium concentration. Ammonium in the soil extracts was determined photometrically after adding KOH (and $Na_2S_2O_3$ for elimination of Hg in the case of DON analysis) and distillation into a H_2SO_4 solution. Nitrate was estimated by means of an ion-selective electrode. Nitrite was estimated photometrically after adding of sulphanic acid and N-(1-naphthyl)ethylene-diamine dihydrochloride as colour reagent.

Measurements of soil temperatures have been discussed in detail by Emmert (1997).

RESULTS AND DISCUSSION

LITTER DECOMPOSITION

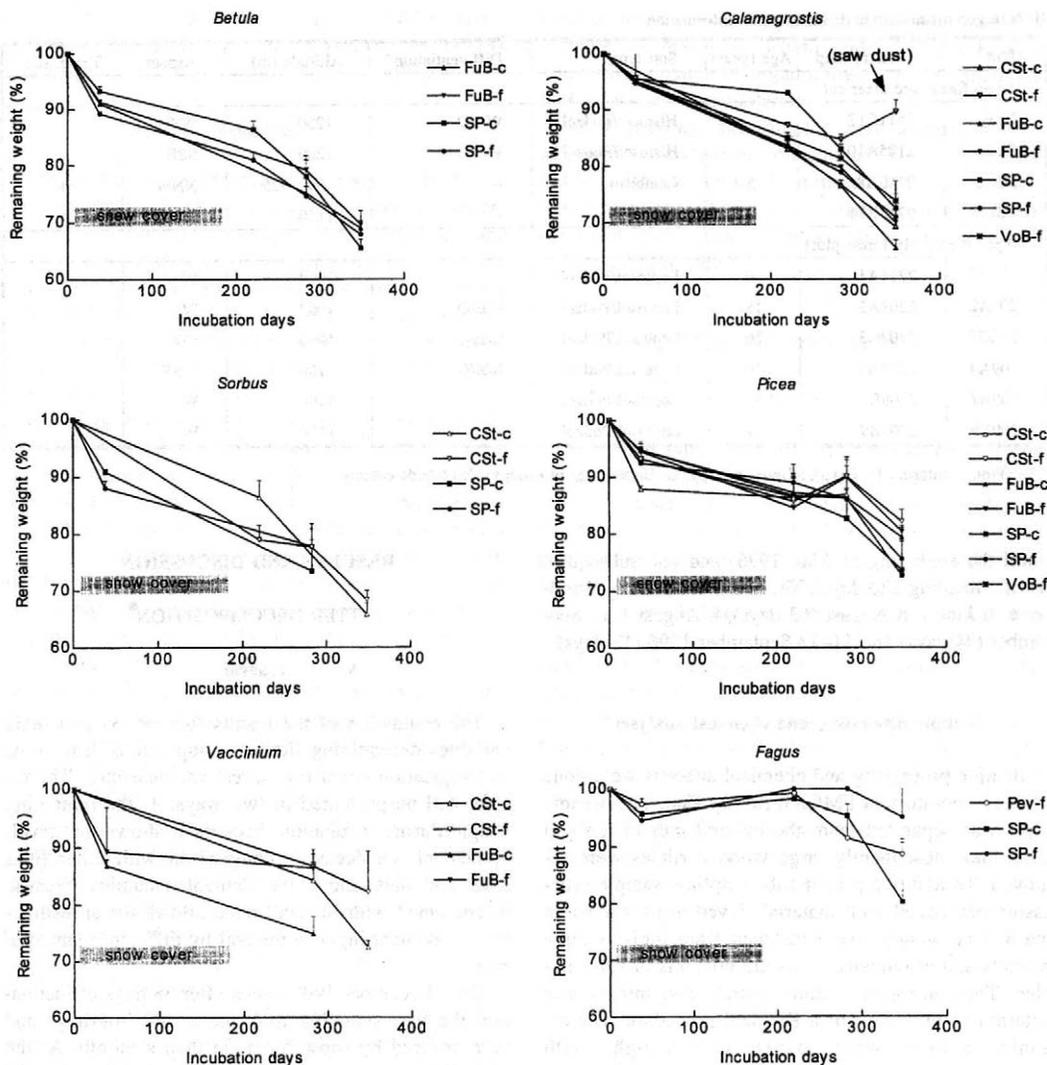
General

The evaluation of the results focused on two main variables determining litter decomposition: litter type and vegetation cover (i.e. forest vs. clear-cut). The results will be presented in two ways: 1. the remaining weight during incubation, because it shows the development of the decay in comparison with other litter types and sites, and 2. the element dynamics, because it, combined with litterfall data, allows for an estimation of element input to the soil by different plant species.

On 7 December 1995, when after 38 days of incubation the first sampling took place, the litterbags had been covered by snow for more than a month. At the CSt clear-cut the bags could not be found under the snow, so that values for the initial stage of decay for this sub-plot are missing.

Regarding soil temperature, as an important determinant of organic matter decay, 3 main factors controlling temperature dynamics could be tentatively extracted from the results (see Emmert, 1997):

1. Soil moisture content: Wet soils show a small difference in average soil temperatures between forest and clear-cut.
2. Exposition: more exposure to the sun leads to higher average temperatures and larger amplitudes.
3. Vegetation cover: forests have lower average soil temperatures and smaller amplitudes. Differences between averages of forests and clear-cuts depend on other factors (moisture and exposition) as well, but the amplitude in forests seems to be consistently approximately 50% lower.



1. First-year weight loss dynamics of various litter types. T-bars indicate standard deviations. For explanation of codes, see Tab. 1 (f: forest; c: clear-cut)

Litter weight loss

Besides differences between sites and vegetation covers, there was a clear difference between the decay rates of litter types. Taking the remaining weight after one year of decomposition as a measure of the relative decay rate, the order of increasing decay is:

1. *Fagus sylvatica* (68–100%), 2. *Vaccinium myrtillus* (72–89%), 3. *Picea abies* (68–84%), 4. *Calamagrostis villosa* (63–76%), 5. *Betula pendula* (64–72%), 6. *Sorbus aucuparia* (64–70%). This order is in agreement with results generally found in literature where coniferous, deciduous and herbal litters are compared (e.g. Swift et al., 1979).

However, the first year of the experiment generally provides the most important information for the study of the first phase of decomposition, which is associated with the highest rates of element dynamics. Moreover, the method becomes increasingly unreliable over the years, due to the contamination of the contained organic matter with non-conforming materials, such as roots, fine humus and mineral particles.

Regarding the shape of the decomposition curves, there is a conspicuous difference in results from other litterbags experiments in Western European temperate forests. Usually, decomposition approximately follows a one-phase exponential decay curve according to: $mass_t = mass_0 e^{-kt}$, where t is time. An inherent feature

of this equation is parameter k , which is the decay constant (y^{-1}). The equation applies to conditions in which the weight loss rate is initially high and decelerates as decay continues. Weight loss in this experiment followed a different trend in all cases (Fig. 1). Initially, there was a rapid decline in the weight of the samples during the first month. This can be attributed to leaching of elements from the organic tissue (cf. Berg, Cortina, 1995). During the following 5 months with snow cover and soil temperatures around freezing point, weight loss was small. Upon disappearance of the snow, decay rates increased and led to a rather consistent weight loss during the vegetation season. It is likely that decay rates would decrease during the next period of snow cover and increase again during the subsequent vegetation season. This would result in a stepwise trend of decay. Evidently, a one-phase exponential curve does not apply to the current results and thus a value for the decay constant k cannot be obtained. However, as the decay rate will decrease over the years, a long-term experiment may allow for the use of exponential curve fitting and calculation of k , despite strong seasonal variations.

Betula, *Calamagrostis* and *Sorbus* leaves decomposed at a more or less constant pace during the 1996 vegetation season, without much difference between forests and clear-cuts or between sites. A more or less similar picture can be seen for *Picea* needles, although these showed a less constant loss rate. *Vaccinium*, with a small number of data points, was the only example where clear-cuts consistently showed larger weight losses than forests. *Fagus* leaves in general decomposed very slowly (around 15% after one year), with a somewhat higher rate (at high variability) on the SP clear-cut.

High temperature and sufficient moisture are preconditions for high decomposition rates. Clear-cuts, with higher average temperatures, did not consistently show a larger weight loss. This may be due to low moisture contents, counteracting the more favourable temperature conditions. Very dry conditions occurred regularly on clear-cuts, during prolonged periods of warm and dry weather.

In long-term litterbag experiments generally an asymptotic limit for mass loss can be observed, reflecting a final stage of low or absent microbial activity

(Berg, Ekbohm, 1993; Berg et al., 1995). In their experiments, *Betula* and *Alnus* leaves initially decomposed more rapidly than *Pinus* needles. However, after about 50% mass loss was reached, *Betula* and *Alnus* stabilised, while *Pinus* needles continued losing weight until about 60–70% of the initial. This proves that no conclusions can be drawn as to the dynamics during later stages of decay, based on dynamics over the first year. Assuming that the conditions in the litterbags mimic those in the organic layer, the contribution of litter to the formation of the organic layer can be calculated, by multiplying the remaining mass after long-term decay with the litter input. However, as stated by Berg et al. (1995), the lack or strong reduction of faunal activity in the litterbags may invalidate this approach. This may be particularly the case when comparing *Picea* and *Betula*. In a study on the relation between tree species and soil development in reclaimed lake soils in the Netherlands (Sevinck et al., 1989), *Betula* was found to produce humus forms exhibiting a much higher degree of faunal activity (Mull to Moder) than *Picea* (Mor to Mormoder), partly attributed to differences in faunal activity.

Element loss from litter

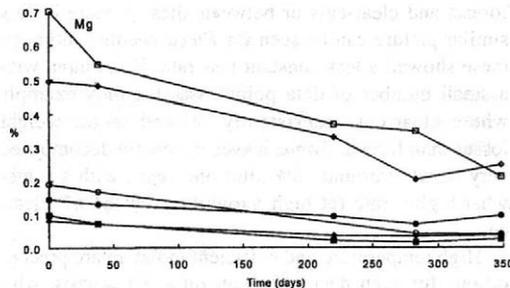
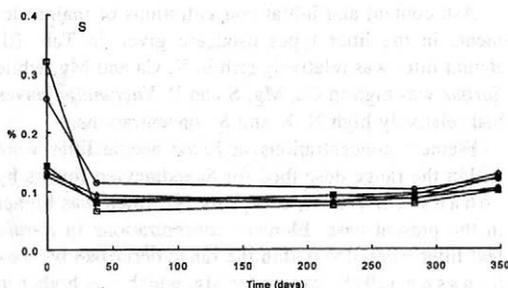
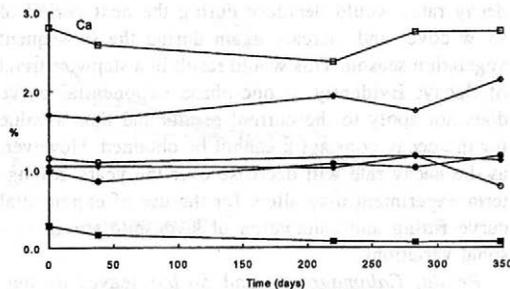
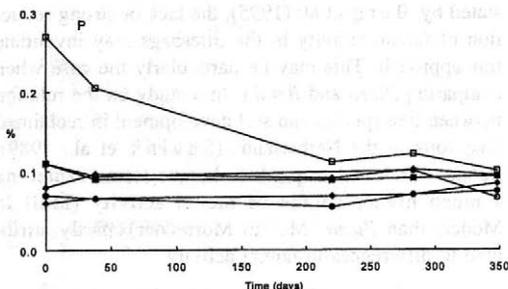
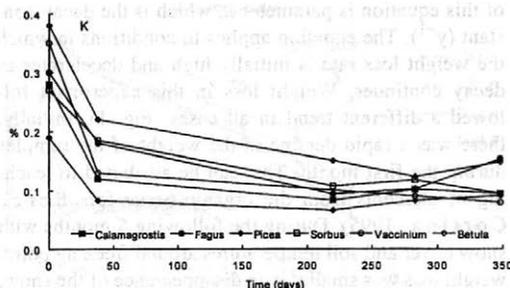
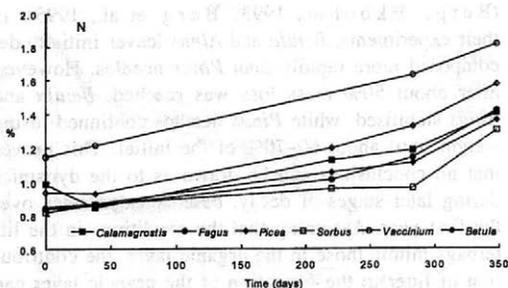
Ash content and initial concentrations of major elements in the litter types used are given in Tab. III. *Betula* litter was relatively rich in K, Ca and Mg, while *Sorbus* was high in Ca, Mg, S and P. *Vaccinium* leaves had relatively high N, K and S concentrations.

Element concentrations in *Picea* needle litter were within the range described for Scandinavian forests by Johansson (1995), except for N, which was higher in the present case. Element concentrations in *Betula* leaf litter were also within the range described by Johansson (1995), except for Mg, which was higher in the present case. The elemental composition of the *Fagus* leaf litter was very similar to that reported by Staf (1980) for beech stands in southern Sweden. In *Vaccinium* leaves, N, P and Mg concentrations were comparable to those reported for Swedish forests by Johansson (1993), while K was lower and Ca higher.

Element loss rates from the litter have been deduced from element concentrations and weight loss after

III. A: Ash and element concentrations in litter types used in litterbags experiments. Relatively high concentrations are indicated by grey shades. B: Species order of increasing loss rate per element. High initial element concentrations are indicated by grey shades

	A: N	P	K	Ca	Mg	S	Ash	B: P	K	Ca	Mg	S
	(%)											
<i>Sorbus</i>	0.82	0.27	0.27	2.80	0.70	0.32	8.6	1	5	1	1	1
<i>Vaccinium</i>	1.15	0.11	0.35	1.14	0.19	0.26	4.7	2	2	2	3	2
<i>Betula</i>	0.94	0.08	0.38	1.70	0.50	0.13	5.5	4	1	3	2	4
<i>Calamagrostis</i>	0.99	0.11	0.28	0.28	0.10	0.14	3.8	3	3	4	4	3
<i>Fagus</i>	0.84	0.07	0.30	0.98	0.15	0.13	5.1	5	4	5	5	5
<i>Picea</i>	0.85	0.06	0.19	1.05	0.08	0.13	11.2	6	6	?	6	6



2. First-year dynamics of element concentrations (averages) in litterbags containing various litter types

348 days of incubation in comparison with initial values. The species order of increasing loss rate per element is shown in Tab. III. This does not involve N, as this element has a different fate than the others. Below, loss rates as well as percentual losses will be discussed. It should be noted that high loss rates do not necessarily coincide with large percentual losses. This is for instance the case if litter with high element concentrations rapidly releases elements. More details are given by Emm er (1997).

P, Ca, Mg and S were leached from *Sorbus* leaves at relatively high rates. A great part of the initial P (65–81%), K (74–80%), Mg (67–93%) and S (74–85%) was released, while this was much less for Ca (29–44%). Loss rates for *Vaccinium* leaves were relatively high as well. They lost most of their K (80–85%) and Mg (69–94%). For P (45–63%), Ca (25–62%) and S (56–71%) a smaller percentual loss was recorded. *Betula* leaves

released Ca and Mg relatively rapidly. They retained much of the Ca (4–26% loss) and P (8–53% loss), but were largely depleted of Mg (62–88%), except for SPc (29–51%) and K (61–80%). *Calamagrostis* leaves lost much of their K (47–82%), Ca (57–97%) and Mg (50–93%) but less of P (8–69%) and S (15–47%), at an intermediate pace. Element losses from *Fagus* leaves, both absolute and percentual, were relatively low and involved negative values. Negative values of release suggest immobilisation by micro-organisms. *Fagus* litterbags retrieved from the Pev-forest site (a pure *Fagus* stand) contained large amounts of white fungal mycelia. Net accumulation of Ca as oxalate salts by fungi was suggested in other research (Berg, Cortina, 1995, and references therein). K (16–79%), Mg (14–81%) and S (1–52%) showed positive, but highly variable values. This variation cannot be attributed to site differences. Element loss rates from *Picea* needles rank low-

est among the litter types selected. Loss rates were quite consistent, except for Ca. Much (46–92%) of Mg was lost, while this was less for K (50–71%) and S (23–57%). Losses of Ca and P were highly variable and included negative values (–7–+93% and –80–+22%, respectively).

First year dynamics revealed an initial heavy weight loss due to leaching, after which element concentrations stabilised (Fig. 2). Mg and K are not structurally bound to the organic tissue, thus not limiting to decomposers, and in general there is a linear relation between accumulated mass loss and decrease in concentration (Berg, Cortina, 1995). After initial heavy leaching, probably of excess S accumulated due to canopy interception, S concentrations stabilised. Berg, Cortina (1995) indicated an increase in S concentration in the long term and this may be the case in this study as well, as concentrations in 348 days showed a slight increase. P concentrations were stable, except for *Sorbus*. As shown by Berg, Cortina (1995), litters high in P (>0.1%) were initially heavily leached of this element. The generally increasing trend in their results cannot be seen in ours, but as for S this may be due to the short duration of the experiment. Ca concentrations were rather constant, but Berg, Cortina (1995) showed that after a certain accumulated weight loss (about 50% in their study) Ca is released faster than organic matter decomposes, leading moreover to a narrower range of concentrations among litter types. In the final stage of their experiment Ca concentrations seemed to level off again. The persisting wide range of Ca concentrations in Fig. 2 is therefore also likely to narrow upon further decay.

Nitrogen generally shows particular dynamics in fresh litter and is therefore treated separately here. In general, 3 phases can be distinguished during litter decay (cf. Berg, Staaf, 1981). At the onset of decay, easily soluble N compounds leach rapidly from the organic tissue, resulting in a net loss of nitrogen (lower concentration as well as lower absolute weight). As N supply is limited compared to C, micro-organisms entering the litter immobilise N, thereby reducing the C/N ratio. This results in an increase of the N concentration and the absolute amount, possibly even exceeding the initial values. After this immobilisation phase, N is released at a rate related to the weight loss of the litter. Fig. 2 shows the dynamics of element concentrations over the first year of decay. During the leaching phase the mass loss is high as well, so that N concentrations remained approximately constant or even increased. The duration of the experiment was too short to reach the end of the immobilisation phase. Immobilisation of N is quite variable, either species and site-wise. The results provide insufficient information about this phase, so that conclusions as to the eventual amount of immobilised N would be dubious.

Differences between forests and clear-cuts involve the dynamics of N, Ca and Mg. Under forest canopies N immobilisation is higher, whereas Ca and Mg release

is higher than on clear-cuts. As decay rates do not differ much between forests and clear-cuts, it is very likely that the differences can be attributed to canopy throughfall. A greater supply of N may result in a greater amount of N immobilised. Furthermore, a greater portion of acidifying compounds in throughfall may cause a higher leaching rate of Ca and Mg from organic tissue.

Element dynamics in the selected litter types resembled the general pattern apparent from other studies. Elements that were present in high concentrations leached from the organic tissue at relatively high rates during the initial stages of decay (first year), with *Sorbus*, *Vaccinium* and *Betula* as good examples. Similar results were found by Berg, Cortina (1995) in Swedish forests involving *Pinus sylvestris*, *Betula pubescens* and *Alnus incana*, and by Staaf (1980) for *Fagus* stands. Nevertheless, they also showed that initial and final (i.e. after 5 years of decay) concentrations of N, P and S were highly correlated. Applying these findings to our results, it may be tentatively concluded that *Sorbus* and *Vaccinium* produce humus high in these elements. Berg, Cortina (1995) also found (linear) relations between accumulated mass loss and concentrations, as well as between initial and final concentrations of K and Mg (not Ca). Thus, initial element concentrations appear to have a good predictive value for element loss rates and final composition of the humus produced.

N DYNAMICS

General

Nitrogen mineralisation rates and nitrification rates were assessed on selected plots representing 3 main soil types (Cambisols, Podzols and Leptosols), and along the chronosequence of *Picea* stands (0–91 years old) on a Podzol/Leptosol complex.

Nitrite concentrations were low and generally below the detection limit of 0.1 mg.kg⁻¹. Therefore, this paper will not deal further with this N species. The lack of information on denitrification in oxygen-limited conditions is a significant shortcoming of this study, as will be discussed further below.

N transformations in 'terrestrial' humus forms will be considered only. Although in some plots Histosols were present (see Tab. II), these were for the greater part covered by Mor-type humus forms of varying thickness.

Total N concentrations were between 2 and 2.5%, except for the first sampling in the SP forest, which shows an erroneously high concentration. In the age sequence, N concentrations were similar, with no clear differentiation between plots. Values for pH-KCl were generally higher in the poorly drained Histosol/Podzol complexes FuB and VoB (around 3–3.5) than on the well drained plots SP and CST.

Dry weights of the organic layer were calculated on an ash-free basis to obtain a better view on accumulated

IV. Pearson correlations between soil properties, N pools and N transformations rates. The data set consisted of plot averages of triplicates ($n = 52$)

	Total N	Dry weight	pH-KCl	LOI	Moisture	N-NH ₄ ⁺	N-DON	N-NO ₃	N-inorg.	N-dyn.	N min.	Nitrificat.
Total N	1											
Dry weight	-0.26	1										
pH-KCl	0.17	-0.30	1									
LOI	-0.65	0.30	-0.05	1								
Moisture	-0.52	-0.01	0.23	0.81	1							
N-NH ₄ ⁺	0.19	-0.23	0.29	0.03	0.09	1						
N-DON	0.14	-0.39	0.56	0.01	0.23	<i>0.38</i>	1					
N-NO ₃	0.04	-0.07	-0.06	-0.02	0.09	0.14	-0.07	1				
N-inorganic	-0.16	-0.11	0.21	<i>0.47</i>	<i>0.47</i>	0.85	<i>0.29</i>	<i>0.34</i>	1			
N-dynamic	-0.23	-0.21	<i>0.43</i>	0.55	0.63	0.63	0.72	0.11	0.80	1		
Net nitrogen mineralisation	-0.17	-0.17	-0.14	0.21	<i>0.43</i>	-0.01	-0.18	<i>0.34</i>	0.17	0.05	1	
Nitrification	-0.11	0.09	-0.24	0.00	0.02	0.05	-0.39	0.25	0.11	-0.18	<i>0.44</i>	1

Italics and bold: $p < 0.001$

Italics: $p < 0.05$

organic matter. Dry weights at CSt (Leptosol) were higher than at other sites. At SP and FuB the clear-cuts had lower dry weights of the organic layer than the forests. There appeared to be an age-related trend, involving an increase with age of approximately 40%.

Relations between soil properties (reference samples), N pools and N transformation rates were generally poor. N pools and N transformations did not depend on total N concentrations and organic matter weight (Tab. IV). Significant correlations were found between dynamic N pool (including all distinguished extractable N species) and pH, LOI and moisture. The inorganic N pool (ammonium and nitrate) significantly correlated with LOI and moisture. Net nitrogen mineralisation rate was correlated with moisture and nitrate content only (in the reference sample), while nitrification was correlated with DON and net nitrogen mineralisation rate only.

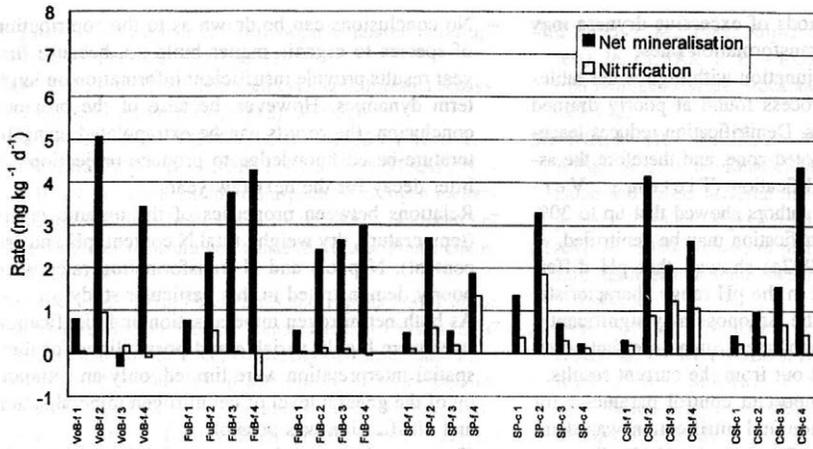
Sites and N transformations

Due to a combination of the facts that a considerable short-distance variation in N transformation rates may occur, and that reference and incubation samples are taken at different locations (although close to each other), there is a possibility that negative rates may be obtained. This will occur in cases where NH₄⁺ in the reference samples is much higher than initial NH₄⁺ in the incubation samples while net nitrogen mineralisation rates are low; the same applies to NO₃⁻ and nitrification. Low concentrations in reference samples and high initial ones in incubation samples have an opposite effect. With this in mind the results in Figs. 3 and 4 should be interpreted.

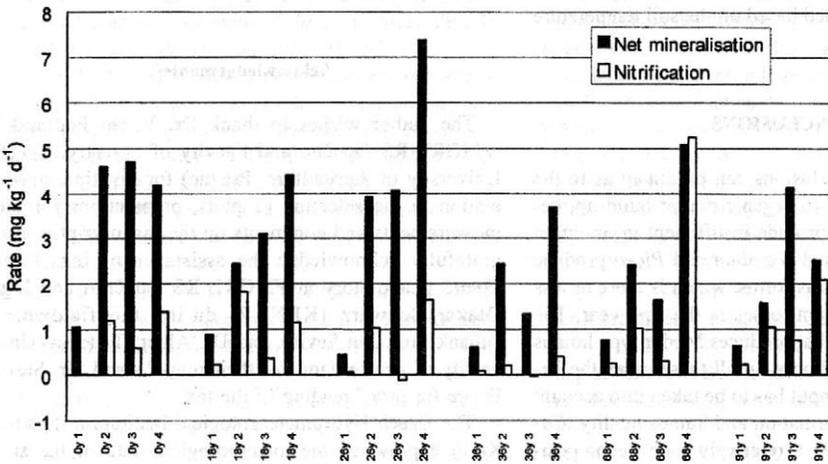
During the first incubation period, the tubes were almost permanently covered by a thick layer of snow.

Probably due to low temperatures and possibly reduced oxygen levels, N transformation rates were lower than during the subsequent vegetative season. Both net nitrogen mineralisation and nitrification rates were highly variable. Nitrification rates in the plots SP and CSt, i.e. the well-drained sites, were higher than in the poorly drained plots VoB and FuB. The conclusion is that with these results no clear general trends can be extracted regarding site conditions to describe N transformation rates. The noise in the results, whether an artefact or due to the complexity of N transformation processes, limits their spatial interpretation. The results therefore allow for the estimation of a general level of net nitrogen mineralisation and nitrification only. Regarding the latter, no further inferences can be made from the result obtained at FuB and VoB, where a distinction was made between wet, dry and intermediate soil moisture conditions.

In the age series, the plots with open canopies showed slightly higher net mineralisation rates in the forest floor than the closed stands. No clear differences in nitrification could be observed. This is in contrast with results from Stevens et al. (1992) and Emmett et al. (1993), who found increasing net mineralisation and nitrification rates with increasing age of Sitka spruce stands in Wales. This cannot be explained by the higher levels of N flux in throughfall. It increased with stand age at the Welsh sites (9–25 kg inorganic N ha⁻¹.year⁻¹) compared to the Krkonoše (up to 20–25 kg inorganic N ha⁻¹.year⁻¹ for the growing season only, according to Hošek, Kaufman, 1995). A better explanation might be the fact that the Welsh sites involved new plantations, whereas the Krkonoše sites were reforestations of spruce clear-cuts, where organic matter quality was dictated by antecedent soil formation; N gain via throughfall may have been a less important factor.



3. Net N-mineralisation and nitrification rates in forests and on clear-cuts. For explanation of codes, see Tab. I (f: forest; c: clear-cut)



4. Net N-mineralisation and nitrification rates in an age series of forests. For explanation of codes, see Tab. II

The general levels of net mineralisation and nitrification, calculated as averages including all plots and scaled to 1 year, were 508 and 125 mg.kg⁻¹.year⁻¹, respectively. Taking 12.5 kg.m⁻² as an average for the ash-free weight of the organic layer (Emmer, 1997), 6.35 g N.m⁻².year⁻¹ net mineralisation and 1.56 g N.m⁻².year⁻¹ nitrification were recorded during the period of measurement. This is lower than the approximately 10 to 35 and 0.3 to 20 g N.m⁻².year⁻¹, respectively, reported by Tietema et al. (1992b), for deciduous and coniferous forests in the Netherlands, and also lower than the average 11 g N.m⁻².year⁻¹, calculated for Central-European deciduous forests by Melillo (1981).

It is questionable whether the relation between moisture content apparent from the statistical analyses agrees with other studies, because moisture content highly correlated with ash content (Tab. IV). Moisture contents calculated for samples rich in mineral particles were generally lower than for pure organic matter (e.g.

in peaty soils), but water availability to micro-organisms operating in the organic matter need not differ to the same extent. Tietema et al. (1992b) show that at low gravimetric moisture content (< 50% of moist weight or < 100% of dry weight) a linear increase in net nitrogen mineralisation rate with increasing moisture content may occur, with increases between 100 and 500%, depending on the type of organic matter. At higher moisture contents, rates levelled off. Average gravimetric moisture contents in this study range from 20 to 90% (of moist weight). Values below 50% only occurred at SP, related to the high ash content of the organic layer. Because of the small differences between the plots, it can be concluded that the high moisture contents recorded did not cause any considerable reduction of net nitrogen mineralisation rates. Under certain circumstances organic layers may be excessively drained. Such cases did not influence the current results, as the sampling of organic layers during dry periods is a matter of chance and did not occur during the experiment.

In reality, however, periods of excessive dryness may significantly reduce N transformation rates.

Denitrification in conjunction with high water tables may be an important process found at poorly drained sites in the Krkonoše Mts. Denitrification reduces leaching of nitrate from the rooted zone, and therefore the associated subsequent acidification (Tietema, Verstraten, 1991). These authors showed that up to 30% of nitrate produced by nitrification may be denitrified.

Tietema et al. (1992a) showed that pH differences of less than 1 unit in the pH range characteristic of the organic layers in the Krkonoše may significantly influence N transformation rates. Such an effect could not, however, be singled out from the current results.

Temperature as an important control parameter for net nitrogen mineralisation and nitrification was demonstrated by Emmer, Tietema (1990). Because of the methodological inaccuracies and the considerable noise in the results due to other effects, no clear relations could be established based on the soil temperature data (Emmer, 1997).

CONCLUSIONS

No quantitative conclusions can be drawn as to the contribution of species to organic matter build-up, because first year results provide insufficient information on long-term dynamics. *Vaccinium* and *Picea* produce Mor or Moder type humus forms, which is more or less indicated by lower weight losses in the first year. The same applies to *Fagus* that produces Moder type humus forms and deep forest floors. In all these cases the below-ground root litter input has to be taken into account to understand humus formation and humus quality. *Calamagrostis* leaves have a relatively high decomposition rate and rapid element depletion of organic tissue (except for N). This is favourable for the species as it is able to regain the nutrients by densely rooting the organic layer.

Summarised conclusions are:

- High element concentrations relate to high initial mass loss rates.
- Initial element concentrations have a good predictive value for element loss rates and final composition of the humus produced.
- Except for N and Ca, the ranges of concentrations narrow considerably upon litter decay during the incubation period.
- There are only small differences between sites and vegetation covers regarding weight loss dynamics.
- Ca and Mg leach faster in forests than on clear-cuts, probably due to elevated levels of acidifying substances in canopy throughfall.
- More N is immobilised in forests than on clear-cuts, probably due to higher N contents in canopy throughfall.
- Element contents and dynamics conform with general trends found in Nordic forests.

- No conclusions can be drawn as to the contribution of species to organic matter build-up, because first year results provide insufficient information on long-term dynamics. However, because of the previous conclusion, the results can be extrapolated using literature-based knowledge to produce projections of litter decay for the next few years.
- Relations between properties of the organic layers (temperature, dry weight, total N content, pH and ash content), N pools and N transformation rates were poorly demonstrated in this particular study.
- As both net nitrogen mineralisation and nitrification rates were highly variable and possibilities for their spatial interpretation were limited, only an estimation of the general level of net nitrogen mineralisation and nitrification was possible.
- The general levels of net mineralisation and nitrification were 508 and 125 mg.kg⁻¹.year⁻¹, or 6.35 and 1.56 g N.m⁻².year⁻¹, respectively.

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VLIV APLIKACE HERBICIDU NA PŮDNÍ VLASTNOSTI IMISNÍCH HOLIN S POROSTY TRAV V BESKYDECH

EFFECT OF HERBICIDE APPLICATION ON SOIL PROPERTIES OF AIR-POLLUTION INDUCED CLEARCUTS WITH GRASS COVER IN THE BESKYDY MTS.

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ABSTRACT: Herbicide applications to grass stands (*Calamagrostis arundinacea*, *C. villosa*) of clearcuts induced by air pollution resulted in an increase of soil organic matter in the upper soil layers of most studied sites. Soil acidity was usually lower by 0.1 to 0.3 pH in the soil of grass stands than in the soil without grass cover. The highest differences in the soil layer of 0–0.1 m represented 0.65 pH-H₂O and 0.56 pH-KCl. A decrease in the contents of base cations was recorded in nearly all soil layers clearcuts. In the 0–0.1 m soil layer, the amounts of Ca²⁺ and Mg²⁺ were mostly by 20 to 60% lower. Lower contents of base cations coincided with higher amounts of Al³⁺ in soil at sites after herbicide application and with more unfavourable Ca²⁺/Al³⁺ ratios. In comparison with grass stands, an increase in contents of ammonium and nitrate ions was recorded in the soil clearcuts at the end of the second growing season after herbicide application, especially at localities with *C. arundinacea*. The application of herbicide and removal of grass cover resulted mostly in worse soil properties. Thus a wide application of herbicide to air-pollution induced clearcuts appears rather questionable.

soil pH; base cations; Al³⁺; soil organic matter; soil nitrogen; *Calamagrostis arundinacea*; *C. villosa*

ABSTRAKT: Po aplikaci herbicidu na travinné porosty imisních holin (s převažující *C. arundinacea* nebo *C. villosa*) došlo ke zvýšení obsahu půdní organické hmoty na většině ploch. Hodnoty aktivního i výměnného pH byly většinou o 0,1 až 0,3 stupně pH nižší než v půdě porostů obou druhů trav. Největší rozdíly představovaly nižší pH až o 0,65 pH-H₂O a 0,56 pH-KCl ve vrstvě 0–0,1 m. Snížení obsahu bazických kationtů, vápníku i hořčíku bylo zaznamenáno téměř ve všech vrstvách půdy na holých plochách bez vegetace. Množství Ca²⁺ a Mg²⁺ ve vrstvě 0–10 cm bylo většinou o 20 až 60 % nižší. Nižší obsahy bazických kationtů se odrážely ve větším množství hlíníku v půdě po aplikaci herbicidu a v nepříznivějších poměrech Ca²⁺/Al³⁺ iontů. Na konci druhé vegetační sezony po aplikaci herbicidu byl zaznamenán vzrůst koncentrace amonných a nitrátových iontů na plochách bez vegetace ve srovnání s porosty obou druhů trav, nejvýrazněji na plochách s porosty *C. arundinacea*. Po aplikaci herbicidu a odstranění aktivního působení travinného porostu došlo tak ke zhoršení sledovaných půdních vlastností. Široké použití herbicidů na imisních holinách může být proto velmi problematické.

pH půdy; bazické kationty; hlíník; půdní organická hmota; půdní dusík; *Calamagrostis arundinacea*; *C. villosa*

ÚVOD

Význam travinné vegetace na odlesněných plochách v imisních oblastech je značný, byť její úloha není ve všech směrech jednoznačně pozitivní. Husté porosty trav se silnou vrstvou opadu ztěžují uchycení sazenic stromů při novém zalesňování imisních holin. Vytvářejí podmínky pro rozvoj populací drobných hlodavců, které mohou poškozovat vysazené dřeviny. Jednotvárnost potravní nabídky pro býložravce může vést k okusu dřevin. Při tání sněhu a jarních deštích, kdy slehlá stařina snižuje průsak vody, dochází ke zvýšenému povrchovému odtoku, což může způsobovat i větší nebezpečí povodní.

Na druhé straně význam travinných porostů pro obnovu narušené krajiny je dán tím, že po odlesnění – díky jejich rychlé expanzi a vytvoření vegetačního krytu – mohou jednak efektivně zvýšit odolnost svrchních vrstev půdy proti vodní erozi, jednak přispět ke snížení ztrát živin z ekosystému po odlesnění (Fiala et al., 1998). Procesy acidifikace a změněné dostupnosti půdních živin pro vegetaci v imisních oblastech souvisejí zejména s vyplavováním bazických kationtů (Ca, Mg) z půdy (Kaupenjohann et al., 1989). Ve srovnání s blízkými smrkovými porosty byla však v travinných porostech imisních holin Beskyd ve svrchních vrstvách půdy často zaznamenána nižší acidita, vyšší obsah ba-

zických kationtů a nižší obsahy toxického hliníku i množství dusíku (Ježková, Tůma, 1995; Zelená et al., 1996; Fiala, 1996c). Nadbytek dusíku v těchto ekosystémech mohou travinné porosty vázat v biomase a imobilizovat v pomalu se rozkládajícím opadu (Tůma, 1996b) a eliminovat tak i živinovou nevyužitost, která je na stanovištích pod vlivem imisí (Schulze et al., 1989).

V lesnické praxi se již řadu let používají selektivní herbicidy pro ochranu lesních kultur a k přípravě půdy pro zalesnění. Sledovala se většinou doba, po kterou byla buřeň na holinách potlačena, a stav kultur sazenic (Švestka, Balek, 1987; Wagner, 1987). Snížení nadzemní biomasy *C. villosa* bylo při vyšších dávkách herbicidu vysoce průkazné ještě v šestém roce po aplikaci a při použití nižších dávek se udržela nižší hmota nadzemní biomasy jen dva roky (Wagner, 1987; Morávková-Lipnická, 1991). Reakce rostlin na aplikaci herbicidu se většinou detailně nezkuňují a stejně tak i změny v půdních vlastnostech.

Uvedené skutečnosti nás vedly k založení terénního experimentu, který spočíval v odstranění porostu trávy imisní holiny pomocí herbicidu; cílem bylo studovat, zda zničení rostlinného krytu je doprovázeno nižšími obsahy vyplavovaných bazických kationtů a nižšími hodnotami pH půdy.

MATERIÁL A METODIKA

Studium vlivu aplikace herbicidu na půdní vlastnosti probíhalo na trvalých výzkumných plochách v Moravskoslezských Beskydách v rámci širších grantových

projektů (Fiala, 1996a). Sledování se uskutečnila na osmi lokalitách: na čtyřech byl porost s dominantní *Calamagrostis arundinacea* (třtina rákosovitá) a na čtyřech převažovala *C. villosa* (třtina chloupkatá). Plochy se nalézaly v různých nadmořských výškách (od 625 m do 1 250 m) a byly charakterizovány širokým rozmezím průměrné roční teploty vzduchu (3,6–7,1 °C) (tab. I). Množství srážek vzrůstalo od lokalit nižších poloh (údolí Černé Ostravice, roční úhrn srážek 869 mm) k lokalitám poblíž vrcholů hor (Malý Smrk, Kněhyně – až 1 146 mm). Plochy také charakterizoval gradient vzrůstající imisní zátěže a vlivu kyselých depozic (20,1–33,5 kg·ha⁻¹ SO₄²⁻ a 5,7–20,6 kg·ha⁻¹ dusíku v mokré depozici za vegetačním obdobím při pH srážkové vody 3,8–4,28), daný většinou vzrůstající nadmořskou výškou (tab. I). Půdy na studovaných lokalitách v Beskydách se vyvinuly z původních hnědých lesních půd po jejich degradaci a podzolizaci v podmínkách vlhkého klimatu a zvláště díky akumulaci surového kyselého humusu v sekundárních jehličnatých lesních porostech (Klimo, Vavříček, 1991). Na studovaných plochách půdy náležejí nejčastěji mezi podzoly typické nebo podzoly kambizemní (tab. I). Snížení půdní acidity o 0,5 pH/H₂O a 0,2 pH/KCl (v povrchových minerálních vrstvách) bylo zaznamenáno po leteckém vápění několika imisních ploch v Beskydách prováděných před více než deseti lety a současně množství CaO v půdě je na stejné úrovni jako před aplikací (Vavříček, Betušová, 1994).

Většina studovaných porostů trav byla na odlesněných plochách, jen některé se nalézaly v prořídlem lese s vývraty smrků (údolí Černé Ostravice) nebo v původ-

I. Základní charakteristiky studovaných ploch – Basic characteristics of studied sites

Lokalita ¹	Poloha ²	Nadmořská výška ³ (m)	Expozice ⁴	Porost ⁵	T ⁶ (°C)	DS ⁷ (mm)	Dešťové srážky ⁸			Půdní typ (klasifikace FAO) ⁹
							pH	SO ₄ ²⁻	N	
Černá Ostravice (ČO)	49°28' N 18°31' E	625	SE	CV	7,1	869	4,28	20,1	19,6	pseudoglej typický ¹⁰
Černá Ostravice (ČO)	49°28' N 18°31' E	633	SE	CA	7,1	869	4,28	20,1	19,6	kambizem pseudoglejová kyselá ¹¹
Bílý Kříž (BK)	49°31' N 18°32' E	890	SW	CA	6,5	948	4,06	24,6	21,1	podzol typický ¹²
Bílý Kříž (BK)	49°28' N 18°31' E	945	SW	CV	6,5	948	4,06	24,6	21,1	podzol typický ¹²
Smrk (SM)	49°30' N 18°19' E	1 140	NW	CA	5,6	1 111	3,93	27,7	21,8	podzol kambizemní ¹³
Malý Smrk (MS)	49°31' N 18°22' E	1 175	NE	CA	5,6	1 107	3,82	33,5	25,3	podzol kambizemní ¹³
Kněhyně (KN)	49°30' N 18°19' E	1 240	SW	CV	3,7	1 102	4,02	25,3	22,8	podzol kambizemní ¹³
Smrk (SM)	49°31' N 18°22' E	1 250	N	CV	3,6	1 146	3,8	21,2	5,7	podzol typický ¹²

CA – *Calamagrostis arundinacea*, CV – *Calamagrostis villosa*

Poznámka: Teploty vzduchu (T) a úhrn dešťových srážek (DS) jsou roční průměry za období 1990–1992 (Hadaš, 1993). Množství SO₄²⁻ a N (v kg·ha⁻¹) a průměrné hodnoty pH byly zaznamenány ve srážkách ve vegetačním období 1995 (Fiala et al., 1996a) – Note: Air temperatures (T) and precipitation (DS) are annual means for 1990–1992 (Hadaš, 1993). SO₄²⁻ and N amounts (kg·ha⁻¹) and average pH values were recorded in precipitation in the growing season 1995 (Fiala et al., 1996a)

¹locality, ²longitude, latitude, ³altitude, ⁴exposure, ⁵type of grass stand (CA – *Calamagrostis arundinacea*, CV – *Calamagrostis villosa*), ⁶temperatures, ⁷precipitation, ⁸wet depositions, ⁹soil type (FAO classification), ¹⁰dystric-planosol, ¹¹stagno-gleyic cambisol, ¹²ferro-humic podzol, ¹³spodo-dystric cambisol

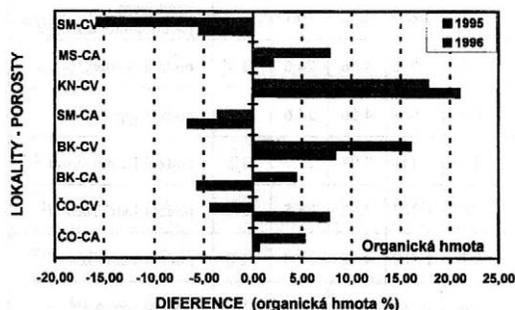
ním rozvolněném a odumírajícím smrkovém lese (poblíž vrcholu hory Smrk) (tab. I). V Moravskoslezských Beskydech převažuje *C. villosa* v náhradních společenstvech, která se vytvořila na plochách zničených původních smrkových lesů ve vyšších polohách nebo na místech kyselých podmaččených smrčtin v nadmořské výšce okolo 600 m. Společenstva s *C. arundinacea* pokrývají rozsáhlé plochy mýtin na místech původních horských bučin nebo smrko-jedlových bučin (Zelená, 1996). Většina těchto lesů byla přeměněna na sekundární smrkové monokultury. Ve studovaných porostech s převažující *C. arundinacea* se nejčastěji vyskytovaly druhy *Vaccinium myrtillus*, *Maianthemum bifolium*, *Rubus idaeus*, *Dryopteris dilatata*, *Oxalis acetosella* a *Senecio fuchsii*. Druhové složení porostů *C. villosa* bylo charakterizováno výraznou dominancí tohoto druhu, který většinou pokrýval celou plochu a byl doprovázen jen několika druhy (především *Vaccinium myrtillus*, *Maianthemum bifolium*, *Rubus idaeus* – Zelená, 1996; Fiala et al., 1996a). Oba druhy trav jsou schopné tolerovat silně kyselé půdy a vysoký obsah iontů hliníku.

V roce 1994 (8. června) byla část porostů trav na studovaných lokalitách ošetřena herbicidem (2% Roundup) a mrtvé nadzemní části rostlin byly z plochy odstraněny. Na konci druhé a třetí vegetační sezony po aplikaci herbicidu (17. 10. 1995 a 15. 10. 1996) byly odebrány půdní vzorky jak z původního porostu trav (kontrola), tak i z ploch, kde byl rostlinný kryt odstraněn (holá plocha). Půdní vzorky představovaly směsný vzorek z pěti náhodně odebraných vzorků pomocí odběrového válce o průměru 50 mm z vrstev 0–0,05, 0,05–0,1 a 0,1–0,2 m (v roce 1995) a 0–0,1 a 0,1–0,2 m (v roce 1996), tj. z kořenové vrstvy trav. Hodnoty aktivního pH byly stanoveny v půdní suspenzi po smíchání 10 g na vzduchu usušené jemnozemi a 25 ml destilované vody po 24 h na pH-metru Radelkis. Ke stanovení výměnného pH bylo použito 1 N roztoku

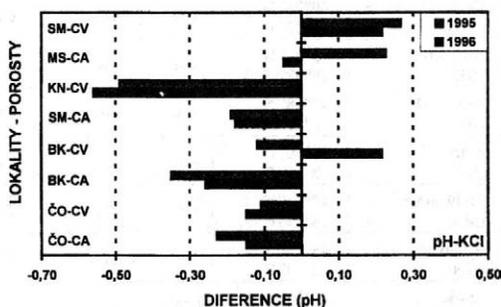
KCl. Obsah organické hmoty v půdě byl určen gravimetricky po spálení vzorků při teplotě 550 °C. Obsah výměnných kationtů (Ca^{2+} , Mg^{2+} , Al^{3+}) byl stanoven komplexometricky (Moravec, 1960). Obsah amonných a dusičnanových iontů byl stanoven v čerstvých půdních vzorcích odebraných stejným způsobem z vrstvy 0–0,1 m, které byly uchovávány při teplotě 4 °C a zpracovány do tří dnů. Ze směsného vzorku (50 g čerstvé hmotnosti) byl získán půdní výluh do 200 ml 1% roztoku K_2SO_4 . Amonné ionty byly stanoveny metodou spektrofotometrické absorbance po reakci s Nesslerovým činidlem (Javorský et al., 1987). Stejně byl také stanoven obsah dusičnanů po reakci se salicylanem sodným (Horáková et al., 1986).

VÝSLEDKY A DISKUSE

Po dvou letech po aplikaci herbicidu došlo ke zvýšení obsahu půdní organické hmoty na většině ploch (tab. II–V, obr. 1). Avšak na stanovištích s vysokou imisní zátěží s porosty *C. arundinacea* (Smrk) a *C. villosa* (Smrk) byly zjištěny nižší obsahy organické hmoty v půdě. Rozdíly zaznamenané po třech letech od zničení porostu byly menší než po dvou letech od aplikace herbicidu. Na plochách bez rostlinného krytu dochází k výrazné změně v mikroklimatických podmínkách – především ve vodním a teplotním režimu půd (Tůma, 1998). Odumřelé oddenky a kořeny trav jsou v příznivějších teplotních podmínkách pravděpodobně rychleji rozkládány a v prvním období po aplikaci herbicidu dochází k výraznému obohacování půdy o organickou hmotu. V dalších letech na holých plochách nedochází již k tvorbě a rozkladu nově vytvořené rostlinné hmoty a k dalšímu výraznému obohacování půdy o organickou hmotu a živiny. Doba obratu podzemní rostlinné hmoty v porostech *C. villosa* byla odhadnuta na tři roky



1. Srovnání rozdílů mezi obsahem organické hmoty v půdní vrstvě 0–0,1 m v porostech trav a na holé ploše po aplikaci herbicidu stanovených v roce 1995 a 1996 na různých lokalitách v Moravskoslezských Beskydech. Označení lokalit a porostů trav v tab. I – Comparison of differences between the contents of soil organic matter in the soil layer of 0–0.1 m in grass stands and bare plots after herbicide application as recorded at different localities in the Moravian-Silesian Beskydy Mts. in 1995 and 1996. Description of localities and stands in Tab. I



2. Srovnání rozdílů mezi výměnným pH (pH-KCl) půdní vrstvy 0–0,1 m v porostech trav a na holé ploše po aplikaci herbicidu stanovených v roce 1995 a 1996 v Moravskoslezských Beskydech. Označení lokalit a porostů trav v tab. I – Comparison of differences between pH (pH-KCl) in the soil layer of 0–0.1 m in grass stands and bare plots after herbicide application as recorded at different localities in the Moravian-Silesian Beskydy Mts. in 1995 and 1996. Description of localities and stands in Tab. I

II. pH půdy, obsah organické hmoty (OH, %), Ca^{2+} , Mg^{2+} , Al^{3+} (v $\text{mg}\cdot 100\text{ g}^{-1}$), N-NH_4^+ , N-NO_3^- a celkového minerálního N (v $\text{mg}\cdot 1000\text{ g}^{-1}$) v půdě na lokalitách s porosty *Calamagrostis arundinacea* a na holých plochách po aplikaci herbicidu. Stanoveno 17. 10. 1995 – pH values, contents of organic matter (OH, %), Ca^{2+} , Mg^{2+} , Al^{3+} (mg.100 g^{-1}), N-NH_4^+ , N-NO_3^- and mineral nitrogen (mg.1000 g^{-1}) in soil of plots either with *Calamagrostis arundinacea* stands (grass cover) or without cover (herbicide) after the application of herbicide. Recorded on October 17, 1995

Varianta ¹	Lokalita ²	Černá Ostravice		Bílý Kříž		Smrk		Malý Smrk	
		tráva ³	herbicid ⁴						
	Vrstva půdy ⁵ (m)								
OH	0,00–0,05	19,4	28,8	27,7	35,9	60,2	59,8	63,2	71,9
	0,05–0,10	8,8	10,0	19,5	20,3	30,9	24,1	22,6	29,6
	0,10–0,20	6,7	6,9	8,9	7,6	11,0	11,2	12,0	14,2
pH-H ₂ O	0,00–0,05	3,92	3,77	3,60	3,85	4,64	4,36	3,93	4,07
	0,05–0,10	3,86	3,82	4,05	3,87	3,92	3,65	3,69	3,60
	0,10–0,20	3,98	3,90	4,11	4,39	3,94	3,70	3,81	3,71
pH-KCl	0,00–0,05	3,97	3,63	3,56	2,99	3,86	3,66	3,88	4,33
	0,05–0,10	4,05	3,93	3,10	2,97	2,90	2,73	3,53	3,53
	0,10–0,20	4,22	4,20	3,08	3,22	2,81	2,62	3,78	3,50
Ca^{2+}	0,00–0,05	94,0	52,0	180,0	165,0	540,0	360,0	390,0	634,0
	0,05–0,10	36,0	18,0	50,0	55,0	122,0	70,0	60,0	152,0
	0,10–0,20	24,0	13,0	28,6	24,0	38,0	18,0	24,0	30,0
Mg^{2+}	0,00–0,05	17,4	8,5	49,9	42,6	145,9	122,8	115,5	170,2
	0,05–0,10	10,5	6,7	23,1	18,2	37,1	26,6	23,1	47,4
	0,10–0,20	4,86	3,04	11,92	9,73	12,8	15,2	9,1	9,1
Al^{3+}	0,00–0,05	87,6	114,6	43,2	95,1	20,2	47,2	60,7	33,7
	0,05–0,10	87,6	114,6	70,1	86,3	80,9	107,8	114,6	132,1
	0,10–0,20	114,6	98,4	80,9	83,6	128,1	80,9	121,3	101,1
$\text{Ca}^{2+}/\text{Al}^{3+}$	0,00–0,05	0,48	0,45	4,16	1,74	26,69	7,63	2,89	18,80
	0,05–0,10	0,41	0,16	0,71	0,64	0,62	0,65	0,24	1,15
	0,10–0,20	0,05	0,13	0,35	0,29	0,13	0,22	0,09	0,29
N-NH_4^+	0,00–0,10	61,05	43,98	80,52	106,41	80,90	83,29	76,79	104,76
N-NO_3^-	0,00–0,10	1,81	8,13	0,87	7,39	2,72	12,18	2,31	12,37
N – min	0,00–0,10	62,86	52,11	81,39	113,80	83,62	95,47	79,10	117,13

¹variant, ²locality, ³grass cover, ⁴herbicide, ⁵soil layer

(Fiala, 1995). Na stanovištích plně exponovaných vůči škodlivinám ve vyšších polohách Beskyd byly v porostech trav zaznamenány nižší přírůstky podzemních orgánů i pomalejší rozklad rostlinného opadu (Fiala, 1996b; Tůma, 1996a), které mohou také souviset s rozdíly stanovenými po aplikaci herbicidu na těchto lokalitách.

Po aplikaci herbicidu byl na plochách bez vegetace v obou sledovaných letech zaznamenán vzrůst acidity téměř ve všech sledovaných vrstvách půdy (tab. II–V). Na těchto plochách hodnoty aktivního i výměnného pH byly většinou o 0,1–0,3 stupně pH nižší než v půdě porostů obou druhů trav (obr. 2). Největší rozdíly (nižší pH až o 0,65 pH-H₂O a 0,56 pH-KCl) byly zjištěny ve vrstvě 0–0,1 m na lokalitě Kněhyně. Výjimku představovaly jen nejvýše položené lokality nejvíce exponované vůči škodlivinám (Malý Smrk a Smrk).

Ačkoliv pH půdy dává užitečnou informaci o momentálním stavu půdy, podává jen malou informaci o její potenciální acidifikaci (Bredemeier et al., 1990). Množství výměnných bazických kationtů je naopak významným indikátorem stupně acidifikace půdy a tyto kationty mají významnou úlohu v omezení vlivu

toxického hliníku. Snížení obsahu bazických kationtů, vápníku i hořčíku, bylo zaznamenáno téměř ve všech vrstvách půdy na holých plochách bez vegetace (tab. II–V). Vzhledem k tomu, že jednotlivá stanoviště, nalézající se v širokém rozmezí nadmořských výšek s odlišnými typy půd, charakterizovaly dosti odlišné půdní vlastnosti, nemělo smysl celý soubor dat zhodnotit statisticky. Výsledky získané při opakovaných analýzách půd v roce 1996 byly však velmi podobné údajům z předchozího roku. Za účelem vzájemného srovnání ploch byly proto vypočteny jen procentuální rozdíly mezi obsahem některých prvků v porostech trav a na holé ploše po aplikaci herbicidu. Tato srovnání stanovišť ukazují lépe na zjištěné rozdíly. Množství Ca^{2+} a Mg^{2+} ve vrstvě 0–10 cm bylo většinou o 20 až 60 % nižší [jedná se o procentuální rozdíl mezi obsahem prvku v půdě porostů trav (= 100 %) a na holých plochách po aplikaci herbicidu] (obr. 3). Výrazně nižší byly v roce 1995 obsahy Ca^{2+} ve svrchní vrstvě půdy (0–0,05 m) na lokalitách Bílý Kříž a Kněhyně s *C. villosa*, které představovaly jen 74 % a 45 % množství zaznamenaného na kontrolních plochách (tab. III). Podobné rozdíly se týkaly také obsahu Mg^{2+} . Vedle sní-

III. pH půdy, obsah organické hmoty, Ca^{2+} , Mg^{2+} , Al^{3+} (v $\text{mg} \cdot 100 \text{ g}^{-1}$), N-NH_4^+ , N-NO_3^- a celkového minerálního N (v $\text{mg} \cdot 1000 \text{ g}^{-1}$) v půdě na lokalitách s porosty *Calamagrostis villosa* a na holých plochách po aplikaci herbicidu. Stanoveno 17. 10. 1995 – pH values, contents of organic matter (OH, %), Ca^{2+} , Mg^{2+} , Al^{3+} ($\text{mg} \cdot 100 \text{ g}^{-1}$), N-NH_4^+ , N-NO_3^- and mineral nitrogen ($\text{mg} \cdot 1000 \text{ g}^{-1}$) in soil of plots either with *Calamagrostis villosa* stands (grass cover) or without cover (herbicide) after the application of herbicide. Recorded on October 17, 1995

Varianta ¹	Lokalita – porost ⁶	Černá Ostravice		Bílý Kříž		Kněhyně		Smrk	
		tráva ³	herbicide ⁴						
	Vrstva půdy ⁵ (m)								
OH	0,00–0,05	54,6	52,8	52,8	54,2	64,5	53,9	71,6	53,3
	0,05–0,10	22,7	15,8	12,7	43,4	22,6	68,9	20,2	6,8
	0,10–0,20	10,2	–	6,7	19,3	9,1	13,1	4,8	5,5
pH-H ₂ O	0,00–0,05	3,71	3,75	4,14	4,05	4,09	3,49	3,66	3,79
	0,05–0,10	3,75	3,76	3,93	3,71	3,71	3,61	3,76	4,30
	0,10–0,20	3,96	3,90	3,81	3,69	4,02	3,88	3,79	4,39
pH-KCl	0,00–0,05	3,31	3,20	4,54	4,30	4,31	3,49	3,44	3,69
	0,05–0,10	3,52	3,40	3,81	3,81	3,59	3,44	3,41	3,69
	0,10–0,20	3,92	4,02	3,90	3,84	3,80	3,47	3,61	3,66
Ca ²⁺	0,00–0,05	80,0	68,0	450,0	117,0	354,0	194,0	84,0	200,0
	0,05–0,10	45,0	50,0	60,0	40,4	95,2	172,0	14,6	9,2
	0,10–0,20	15,4	60,0	24,0	9,2	41,4	46,0	10,6	9,4
Mg ²⁺	0,00–0,05	9,4	6,1	133,8	31,6	103,6	39,4	36,5	59,6
	0,05–0,10	6,1	13,4	25,5	4,9	25,5	32,3	6,4	7,9
	0,10–0,20	2,4	2,4	7,3	2,4	14,2	15,3	2,8	2,9
Al ³⁺	0,00–0,05	74,2	94,4	23,6	121,4	67,4	105,2	87,6	76,5
	0,05–0,10	128,1	78,9	64,0	80,9	101,1	114,6	90,3	59,7
	0,10–0,20	121,3	57,3	84,3	80,0	82,3	106,5	42,2	54,9
Ca ²⁺ /Al ³⁺	0,00–0,05	1,08	0,72	19,07	0,96	5,25	1,84	0,96	2,62
	0,05–0,10	0,35	0,63	0,94	0,50	0,94	1,50	0,16	0,15
	0,10–0,20	0,13	0,10	0,28	0,12	0,50	0,43	0,25	0,17
N-NH ₄ ⁺	0,00–0,10	95,41	90,28	121,21	136,87	66,24	69,24	58,75	71,47
N-NO ₃ ⁻	0,00–0,10	0,81	1,31	0,97	1,07	2,76	10,63	3,76	2,30
N – min	0,00–0,10	96,22	91,59	122,18	137,94	69,00	79,87	62,51	73,77

For 1–5 see Tab. II, ⁶stand

žení obsahu Ca^{2+} a Mg^{2+} ve svrchní vrstvě (0–0,05 m) byl však zaznamenán vyšší obsah těchto kationtů v hlubší vrstvě (0,05–0,1 m) na několika lokalitách (Kněhyně, Černá Ostravice – plochy s *C. villosa*, Bílý Kříž – *C. arundinacea*) (tab. II, III). Poměrně menší rozdíly byly zjištěny na plochách s *C. arundinacea*. Na imisní holině stanovili Fiala et al. (1998) třikrát až čtyřikrát větší množství vyplavovaných bazických kationtů z holé půdy než z porostů trav. Na nejvýše položených lokalitách, plně exponovaných vůči škodlivinám (Malý Smrk, Smrk – *C. villosa*), byl však v obou letech po aplikaci herbicidu zjištěn větší obsah výměnného Ca^{2+} (až o 74 % a 112 % ve srovnání s porostem trav) a Mg^{2+} (až o 36 % a 57 %) ve vrstvě 0–0,1 m. Byly zde také zaznamenány vyšší hodnoty pH půdy (tab. III, V). Tyto rozdíly mohou být dány i variabilitou studovaných ploch. Zjištění však byla pro nás překvapující, neboť tyto lokality byly řazeny mezi nejvíce exponované vůči škodlivinám, kde by mělo docházet k výraznému vyplavování bazických kationtů. Nicméně množství SO_4 a dusíku v mokré depozici stanovená za vegetační období na lokalitě poblíž vrcholu hory Smrk (*C. villosa*) patřila k nižším množství zaznamenaným

v oblasti (Fiala et al., 1996a). Podzoly kambizemní (Malý Smrk – *C. arundinacea*) mohou mít horší hydraulickou vodivost půdy a ve svažitých terénech vyšších poloh může tak převládat spíše povrchový odtok srážek a vyplavování živin nemusí být tak intenzivní jako na jiných lokalitách.

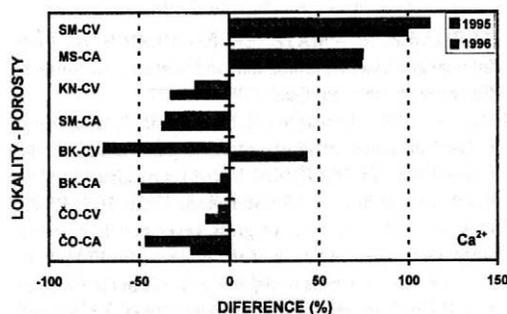
Nižší obsahy bazických kationtů se odrážely ve větším množství hliníku v půdě po aplikaci herbicidu (tab. II–V). Poměr $\text{Ca}^{2+}/\text{Al}^{3+}$ iontů je považován za velmi důležitou charakteristiku při hodnocení toxicity hlinitých iontů (Ulrich, 1987). Na plochách bez porostu trav byly tyto poměry mnohem nepříznivější. Ve svrchní vrstvě půdy (0–0,05 m) holé plochy došlo k výraznějšímu snížení těchto poměrů v roce 1995 především na lokalitách s *C. villosa* (Bílý Kříž, Kněhyně) než na lokalitách s *C. arundinacea* (tab. I, II).

Vzrůst vstupů dusíku ve srážkách do lesních ekosystémů v posledních desetiletích změnil poměry v dostupnosti dusíku v lesních ekosystémech střední Evropy. Tato skutečnost přispěla k vyplavování zásoby kationtů z půdy díky acidifikaci, která je důsledkem nitrifikace (Nihlgard, 1985). Stanovení amonných a nitrátových iontů na konci vegetační sezony 1995 ukázala na

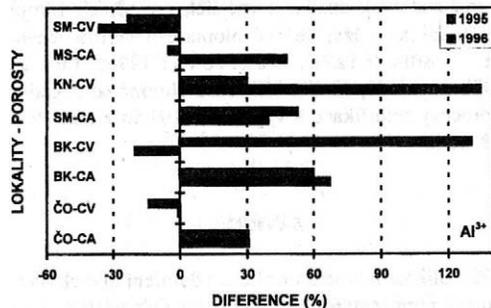
IV. pH půdy, obsah organické hmoty, Ca^{2+} , Mg^{2+} , Al^{3+} (v mg.100 g⁻¹), N-NH_4^+ , N-NO_3^- a celkového minerálního N (v mg.l 000 g⁻¹) v půdě na lokalitách s porosty *Calamagrostis arundinacea* a na holých plochách po aplikaci herbicidu. Stanoveno 15. 10. 1996 – pH values, contents of organic matter (OH, %), Ca^{2+} , Mg^{2+} , Al^{3+} (mg.100 g⁻¹), N-NH_4^+ , N-NO_3^- and mineral nitrogen (mg.1000 g⁻¹) in soil of plots either with *Calamagrostis arundinacea* stands (grass cover) or without cover (herbicide) after the application of herbicide. Recorded on October 15, 1996

Varianta ¹	Lokalita ²	Černá Ostravice		Bílý Kříž		Smrk		Malý Smrk	
		tráva ³	herbicid ⁴						
	Vrstva půdy ⁵ (m)								
OH	0,00–0,10	13,4	14,1	24,8	19,1	39,2	32,6	25,6	27,7
	0,10–0,20	6,3	7,2	8,5	7,2	11,9	10,9	10,3	12,3
pH-H ₂ O	0,00–0,10	4,18	3,96	4,32	3,96	4,28	4,20	3,92	3,92
	0,10–0,20	4,33	4,25	3,99	4,07	3,91	3,74	4,00	3,82
pH-KCl	0,00–0,10	3,32	3,17	3,40	3,14	3,40	3,22	3,05	3,00
	0,10–0,20	3,58	3,59	3,05	3,32	2,86	2,69	3,21	2,97
Ca^{2+}	0,00–0,10	28,0	22,0	78,0	40,0	256,0	160,0	62,0	108,0
	0,10–0,20	10,0	12,0	14,0	12,0	22,0	18,0	16,0	16,0
Mg^{2+}	0,00–0,10	7,3	13,4	27,9	12,2	76,6	31,6	13,4	18,2
	0,10–0,20	4,0	6,1	6,1	4,9	7,3	6,1	4,1	4,9
Al^{3+}	0,00–0,10	78,2	102,5	49,9	83,6	40,5	56,6	72,8	107,9
	0,10–0,20	94,9	111,9	82,3	82,3	107,9	101,1	94,4	105,2
$\text{Ca}^{2+}/\text{Al}^{3+}$	0,00–0,10	0,36	0,21	1,56	0,48	6,32	2,82	0,85	1,00
	0,10–0,20	0,10	0,11	0,17	0,14	0,20	0,18	0,17	0,15
N-NH_4^+	0,00–0,10	48,10	48,20	50,10	49,90	53,20	53,30	53,60	52,80
N-NO_3^-	0,00–0,10	1,58	1,12	0,47	3,87	6,39	8,11	6,01	4,94
N – min	0,00–0,10	49,68	49,32	50,57	53,77	59,59	61,41	59,61	57,74

For 1–5 see Tab. II



3. Srovnání procentuálních rozdílů mezi obsahy Ca^{2+} v půdní vrstvě 0–0,1 m v porostech trav (= 100 %) a na holé ploše po aplikaci herbicidu stanovených v roce 1995 a 1996 na různých lokalitách v Moravskoslezských Beskydách. Označení lokalit a porostů trav v tab. I – Comparison of percentage differences between Ca^{2+} contents in the soil layer of 0–0.1 m in grass stands (= 100%) and bare plots after herbicide application as recorded at different localities in the Moravian-Silesian Beskydy Mts. in 1995 and 1996. Description of localities and stands in Tab. I



4. Srovnání procentuálních rozdílů mezi obsahy Al^{3+} v půdní vrstvě 0–0,1 m v porostech trav (= 100 %) a na holé ploše po aplikaci herbicidu stanovených v roce 1995 a 1996 na různých lokalitách v Moravskoslezských Beskydách. Označení lokalit a porostů trav v tab. I – Comparison of percentage differences between Al^{3+} contents in the soil layer of 0–0.1 m in grass stands (= 100%) and bare plots after herbicide application as recorded at different localities in the Moravian-Silesian Beskydy Mts. in 1995 and 1996. Description of localities and stands in Tab. I

vzrůst koncentrace půdního dusíku na plochách bez vegetace ve srovnání s porosty obou druhů trav, nejvýrazněji však na plochách s porosty *C. arundinacea* (tab. II–V). Po odstranění porostů *C. arundinacea* byla koncentrace nitrátových iontů v půdě čtyřikrát až osmkrát vyšší. Výjimkou byly jen koncentrace amonného dusíku v půdě po odstranění *C. villosa* i *C. arundinacea* na plochách nižších poloh (údolí Černá Ostravice).

V podzimním období dalšího roku (1996) nebyly již takové výrazné rozdíly zaznamenány. Větší množství vyplavovaných nitrátů z půdy lesa ve srovnání s porosty trav na imisní holině zaznamenal v Beskydách Záhora (1997). Podobně i v průsakových vodách zachycených z variant bez porostu trav na imisní holině byly stanoveny vyšší koncentrace nitrátového dusíku (Fiála et al., 1998). Porosty trav vytvořené na imisních

V. pH půdy, obsah organické hmoty, Ca^{2+} , Mg^{2+} , Al^{3+} (v $\text{mg}\cdot 100\text{ g}^{-1}$), N-NH_4^+ , N-NO_3^- a celkového minerálního N (v $\text{mg}\cdot 1\ 000\text{ g}^{-1}$) v půdě na lokalitách s porosty *Calamagrostis villosa* a na holých plochách po aplikaci herbicidu. Stanoveno 15. 10. 1996 – pH values, contents of organic matter (OH, %), Ca^{2+} , Mg^{2+} , Al^{3+} ($\text{mg}\cdot 100\text{ g}^{-1}$), N-NH_4^+ , N-NO_3^- and mineral nitrogen ($\text{mg}\cdot 1000\text{ g}^{-1}$) in soil of plots either with *Calamagrostis villosa* stands (grass cover) or without cover (herbicide) after the application of herbicide. Recorded on October 15, 1996

Varianta ¹	Lokalita ² – porost ⁶	Černá Ostravice		Bílý Kříž		Kněhyně		Smrk	
		tráva ³	herbicide ⁴						
	Vrstva půdy ⁵ (m)								
OH	0,00–0,10	18,1	25,1	20,9	29,3	33,7	54,8	22,5	17,1
	0,10–0,20	5,6	6,2	4,6	6,8	11,3	10,6	6,6	7,5
pH-H ₂ O	0,00–0,10	3,69	3,59	4,17	4,47	4,27	3,62	3,67	3,90
	0,10–0,20	4,13	3,99	4,17	4,18	3,92	3,73	3,72	3,99
pH-KCl	0,00–0,10	2,90	2,75	3,26	3,48	3,30	2,74	2,66	2,88
	0,10–0,20	3,40	3,27	3,26	3,22	2,99	2,76	2,76	3,02
Ca ²⁺	0,00–0,10	30,0	26,0	106,0	152,0	148,0	100,0	18,0	36,0
	0,10–0,20	10,0	8,0	9,0	12,0	12,0	14,0	8,0	8,0
Mg ²⁺	0,00–0,10	8,5	8,0	27,0	37,7	48,6	24,3	7,3	10,9
	0,10–0,20	3,6	3,6	3,6	6,1	4,9	6,1	3,6	3,0
Al ³⁺	0,00–0,10	126,7	125,4	59,3	47,2	43,1	101,1	80,9	51,2
	0,10–0,20	98,4	101,1	45,8	91,7	76,8	49,9	55,3	53,9
Ca ²⁺ /Al ³⁺	0,00–0,10	0,24	0,21	1,79	3,22	3,43	0,99	0,22	0,70
	0,10–0,20	0,10	0,08	0,19	0,13	0,16	0,28	0,14	0,15
N-NH ₄ ⁺	0,00–0,10	51,80	50,80	50,12	50,80	54,30	56,90	52,50	49,13
N-NO ₃ ⁻	0,00–0,10	3,11	2,56	6,65	4,35	7,26	3,74	3,36	5,89
N – min	0,00–0,10	54,91	53,36	56,77	55,15	61,56	60,64	55,86	55,02

For 1–6 see Tab. III

holinách a na ně vázaná společenstva mikroorganismů mohou nadbytek dusíku z vnějších i vnitřních zdrojů i další živiny vázat ve své biomase a nerozloženém opadu rostlin (Fiala, Jakrlová, 1996; Tůma, 1996b) a pravděpodobně tak zčásti eliminovat negativní procesy acidifikace a vyplavování živin z půdy lesních holin.

ZÁVĚRY

Po aplikaci herbicidu došlo k odumření živých nadzemních i podzemních orgánů rostlin. Odumřelé oddenky a kořeny jsou poměrně rychle rozloženy (Tůma, 1994; Fiala, 1996a). Dochází k rozvolnění detritové vrstvy. Proto existuje značné nebezpečí zvýšené půdní eroze při velkoplošných aplikacích herbicidu na svazích s větším sklonem při přívalových deštích. Po aplikaci herbicidu a odstranění aktivního působení travinného porostu může dojít ke zhoršení půdních vlastností, např. ke zvýšení půdní acidity, snížení obsahu výměnného vápníku a hořčíku v půdě a ke zvýšení koncentrace toxického hliníku a obsahu dusíku ve svrchních vrstvách půdy. Široké použití herbicidů na imisních holinách je tak velmi problematické.

Poděkování

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EFFECT OF HERBICIDE APPLICATION ON SOIL PROPERTIES OF AIR-POLLUTION INDUCED CLEARCUTS WITH GRASS COVER IN THE BESKYDY MTS.

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In the Moravian-Silesian Beskydy Mts., higher contents of Ca and Mg and lower soil acidity and contents

of soil nitrogen in upper soil layers were recorded by several authors in grass stands developed on deforested

sites in comparison with adjacent spruce forests. Nevertheless, dense grass stands with thick layer of dead plant litter on the soil surface prevent tree saplings take roots during reforestation of air-pollution induced clearcuts. These facts motivated us to establish a field experiment based on the removal of grass stands by the application of herbicide, and to examine changes in soil properties, especially the possibility of enhanced leaching of base cations accompanied by a lower pH in plots devoid of grass cover.

Eight experimental sites (four dominated by *Calamagrostis arundinacea* and four by *C. villosa*) were chosen. Studied sites were situated at different altitudes (625 up to 1250 m a.s.l.) and they were therefore characterised by a broad range of mean annual air temperatures (3.6 to 7.1 °C) and annual precipitation (869 to 1146 mm). They were also located along a gradient of increasing air pollution and acid deposits (13.8 to 19.2 $\mu\text{g m}^{-3}$ of an average annual concentration of SO_2 , 20.1 to 33.5 $\text{kg}\cdot\text{ha}^{-1}$ of SO_4^{2-} , and 5.7 to 25.3 $\text{kg}\cdot\text{ha}^{-1}$ of nitrogen as an input of wet bulk deposits during the growing season at a rain water pH between 3.8 to 4.28). Bedrock of studied sites is represented by sandstone. Soils mostly belong to ferro-humic podzols or spodosystric cambisols which have developed from original brown forest soils after their degradation and podzolization.

A part of each grass stand was treated by the herbicide (2% Roundup) in the first half of June 1994 in order to destroy the plant cover and the dead above-ground plant parts were removed. At the end of the second (October 17, 1995) and third (October 15, 1996) growing seasons, soil samples were taken both in living grass stands (control plots) and in area devoid of plant cover (bare plot) from the soil layers 0–0.05, 0.05–0.1, and 0.1–0.2 m (in 1995) and 0–0.1 and 0.1–0.2 m (in 1996), i.e., in the root zone of grasses. Soil samples represented a mixed sample from five soil cores, taken randomly, each 50 mm in diameter. The content of soil organic matter was assessed by incineration at 550 °C. A Radelkis pH meter was used for measuring pH-H₂O and pH-KCl. The content of exchangeable cations (Ca^{2+} , Mg^{2+} , Al^{3+}) was assessed complexometrically. Nitrogen forms were determined colorimetrically in the 1% K_2SO_4 soil extract: N-NH_4^+ using Nessler's reagent and N-NO_3^- after the reaction with sodium salicylate.

The application of herbicide to grass stands of air-pollution induced clearcuts resulted in an increase of soil organic matter in the upper soil layers of most studied sites. Differences recorded three years after

stand removal (in 1996) were lower than those found one year earlier (in 1995). Soil water and temperature regimens changed considerably on bare plots. Thus dead roots and rhizomes were decomposed more rapidly, which resulted in a greater content of soil organic matter. The application of herbicide increased soil acidity in nearly all examined soil layers. Soil pH values were usually by 0.1 to 0.3 pH higher in the soil layer of 0–0.1 m of grass stands than in soil without grass cover. The highest differences represented 0.65 pH-H₂O and 0.56 pH-KCl. Similarly, a decrease in the contents of base cations was recorded in individual soil layers of bare plots without vegetation. Amounts of Ca^{2+} and Mg^{2+} in the 0–0.1 m soil layer were mostly by 20 to 60% lower [expressed as percentage differences between the contents of the respective cations in the soil of a grass stand (= 100%) and in a bare plot after herbicide application]. However, the decrease in both Ca^{2+} and Mg^{2+} contents recorded in the upper soil layer (0–0.05 m) was accompanied by an increase of these cations in the deeper soil layer (0.05–0.1 m) at several localities. Lower contents of base cations coincided with greater amounts of toxic Al^{3+} ions in soil at sites after herbicide application and in more unfavourable $\text{Ca}^{2+}/\text{Al}^{3+}$ ratios. Exceptions concerning changes in soil organic matter, pH, and exchangeable cations were found only at localities situated in the uppermost mountain zone. This fact was surprising, while these localities were expected to be the most polluted ones, where the leaching of base cations should be quite evident. On steep slopes at the higher altitudes, the surface runoff could probably prevail and leaching of nutrients was not so intensive as at other sites. At the end of the second growing season since the application of herbicide, an increase in the contents of ammonium and nitrate ions was recorded in the soil of plots without vegetation. Concentrations of nitrates increased in bare plots four to eight times especially at localities with *C. arundinacea*.

Above-ground and below-ground plant organs are damaged after the application of herbicide. The content of soil organic matter increases due to the fast decomposition of dead rhizomes and roots. However, open sites of steep slopes are endangered by soil erosion during heavy rains. The removal of grass activity resulted mostly in impaired soil properties, i.e. lower soil pH and base cation contents, and higher contents of toxic aluminium. An extensive application of herbicide to air-pollution induced clearcuts thus appears rather questionable.

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MEZINÁRODNÍ SEMINÁŘ O EVROPSKÝCH HORSKÝCH LESÍCH V TRENTU

Krátce po třetí ministerské konferenci o ochraně evropských lesů v Lisabonu byl do alpského Trenta v Itálii svolán seminář, týkající se lesních ekosystémů v horách (2nd International Workshop *A European Project for Mountain Forest*). Seminář ve dnech 24.–26. září 1998 organizovala European Observatory of Mountain Forest se sídlem ve francouzském Saint Jean d'Arvey. Na realizaci semináře (včetně finančních nákladů) se podílela i místní samospráva, lesnické a jiné organizace (Federation Europeenne des Communes Forestieres, Autonomous Region Trentino South Tyrol, Autonomous Province Trento, Municipality of Trento, Federation of Italian Forestry Communities, Azienda Forestale Trento-Sopramonte, Fondazione Cassa di Risparmio di Trento e Rovereto, Magnifica Comunita di Fiemme). Seminář se konal pod patronací italských ministerstev životního prostředí a zemědělské politiky (Ministry of Environment a Ministry of Agricultural Policy). Cílem setkání byla vzájemná výměna informací o stavu a pěstební strategii v horských lesích zastoupených evropských zemí a diskuse o aktuálních problémech. Seminář se zúčastnili zástupci 17 zemí (Albánie, Andorra, Belgie, Česká republika, Francie, Itálie, Maďarsko, Německo, Norsko, Polsko, Portugalsko, Rakousko, Řecko, Slovinsko, Španělsko, Švýcarsko, Velká Británie). Přátelský ohlas mělo i vystoupení zástupců jediné zahraniční privátní organizace – Sdružení vlastníků obecních a soukromých lesů v České republice.

Po třech dnech jednání byla prodiskutována a schválena závěrečná deklarace. V ní byly konstatovány následující skutečnosti:

- zvyšující se význam polyfunkční úlohy lesů v horských oblastech,
- zvyšující se význam horských lesů při jednání mezinárodních, evropských, regionálních a lokálních institucí,
- rozdílnost národních, regionálních a lokálních postojů a zájmů při řešení problematiky horských lesů,
- potřeba maximalizace kompletnosti a vyloučení duplicit při řešení problémů trvale udržitelného hospodaření v horských lesích různými institucemi,
- potřeba přizpůsobení ochrany životního prostředí (vývoje a udržení přírodních zdrojů a sociálně ekonomických aktivit) na úrovni přijatelné pro rozhodnutí plynoucí z mezinárodních ministerských konferencí konaných v Lisabonu, Helsinkách a Štrasburku.

Účastníci semináře vyslovili podporu realizaci Rezoluce S4 (Štrasburk, 1990) o přizpůsobení hospodaření v horských lesích novým environmentálním podmínkám, a navrhli následující aktivity:

- každé dva roky uspořádat seminář, který by byl nezávislým mezinárodním fórem pro subjekty zainteresované na trvale udržitelném hospodaření v horských lesích, kde by byly prodiskutovány aktuální problémy,
- vytvořit komunikační síť pro politiku, techniku, výzkum a praxi související s trvale udržitelným hospodařením v horských lesích,
- vytvořit síť vzorových lokalit v každé zainteresované zemi sloužící pro demonstraci výsledků lesopěstebních aktivit,

monitoring stavu a vývoje lesních ekosystémů a pro mezinárodní výměnu poznatků v daném oboru,

- sestavit mezinárodní databázi údajů ze oblasti životního prostředí, obsahující rovněž výsledky ze sítí vzorových lokalit podle kritérií a ukazatelů přijatých v Lisabonu,
- organizovat pracovní setkání zástupců zainteresovaných zemí pro řešení problematiky rezolucí přijatých v Lisabonu, Helsinkách a Štrasburku.

Ministerstva, do jejichž resortu spadá péče o lesy, jsou závěrečnou deklaraci žádána o zahrnutí závěrů Ministerské konference konané v Lisabonu do svých pracovních plánů (zmíněn bod 5 generální deklarace).

Evropská komise a ministerstva jsou žádána o přijetí článku 30, kap. VIII. (Proposals of Regulation of Rural Development), který je zaměřen na trvale udržitelné hospodaření v lesích.

Kompetentní subjekty jsou žádány o zvážení ekologických a ekonomických funkcí lesů ve státních rozpočtech (zmíněn skleníkový efekt a konference v Kjótu).

Po třídenním semináři v renesančním paláci Geremio (z počátku 16. století) byli účastníci pozváni na exkurzi do blízkého údolí v Dolomitech a města Cavalese. Diskutovalo se o lesích v oblasti Trentina. Jejich výměra zde činí 344 tisíc ha, tj. 55 % rozlohy celé oblasti. Smrk je v porostech zastoupen 60 %, modřín 17 %, jedle 11 %, borovice lesní 6 %, borovice limba 1 %, buk 4 % a ostatní listnáče 1 %. Horní hranice výskytu dřevinné vegetace se pohybuje v rozmezí 2 200–2 800 m n. m. Zásoba dřevní hmoty se odhaduje na 55 mil. m³, roční přírůstek na 1,3 mil. m³ a roční těžba činí (pouze) 0,6 mil. m³. Poměrně nízký objem těžeb je způsoben malým zájmem o dřevní hmotu, jejíž cena od padesátých let poklesla přibližně na pětinu.

Většinu lesů v Trentinu vlastní obce (70 %) a sdružení občanů (27 %), pouze malá část patří státu (2 %) a soukromníkům (1 %). Podle tradice zděděné po rakouské správě se každý rok v Trentinu konají veřejná zasedání lesníků, na kterých se připravuje program aktivit na další rok a řeší se problémy spojené s lesním hospodářstvím. Za potenciální nebezpečí pro lesy se zde považují kyselé srážky a znečištění ovzduší, klimatické extrémy, dřevokazné houby, hmyz a požáry. Lesní porosty jsou ve velmi dobrém zdravotním stavu, plní především důležité ekologické funkce – hlavně ochranu proti lavinám. Zhruba 40 % území má větší sklon než 55 %.

Součástí exkurze byla návštěva místní samosprávy Magnifica Comunita di Fiemme. Komunita sdružuje obyvatele obcí v daném horském údolí. Zčásti jim zajišťuje práci (například vlastní dřevokombinát o zpracovatelské kapacitě 35 000 m³ dřeva ročně), pomáhá řešit právní záležitosti a podporuje sociálně slabší občany.

Na semináři byl přednesen příspěvek *Trvale udržitelné obhospodařování horských lesů v České republice*, jehož zkrácenou českou verzí uvádíme. (Plné znění příspěvku bude publikováno ve sborníku ze semináře.)

Od počátku osmdesátých let se s rozvojem ekologických poznatků a současně se zhoršováním stavu lesů zvyšuje pozornost, kterou v České republice věnuje nejširší veřejnost i politikové lesům. Les je chápán jako ekosystém, kterému je

I. Charakteristiky lesních vegetačních stupňů lesů v horských lesích ČR – Forest vegetation zones and their percentage in the Czech mountains

Lesní vegetační stupeň ¹		Výměra* ²		Zastoupení ³ (%)		Nadmořská výška ⁴	Průměrná teplota ⁵	Roční srážky ⁶	Vegetační doba ⁷
		(ha)		z 6–9 LVS ¹²	z 0–9 LVS ¹³	(m n. m.)	(°C)	(mm)	(dny ¹⁴)
9	klečový ⁸	5 857		1,27	0,22	>1 350	<2,5	>1 500	<60
8	smrkový ⁹	44 522		9,69	1,69	1 050–1 350	2,5–4,0	2 300–1 500	60–100
7	bukosmrkový ¹⁰	99 824		21,72	3,79	900–1 050	4,0–4,5	1 050–1 200	100–115
6	smrkobukový ¹¹	309 367		67,32	11,75	700–900	4,5–5,5	900–1 050	115–130

*Celková výměra lesů v ČR je 2 632 tisíc ha, z toho výměra horských lesů (6.–9. LVS) je 459 570 ha, tj. 17,45 % (údaje podle ÚHÚL) – Total forest area in the Czech Republic is 2,632 thousand ha, forest area in the Czech mountains is 459 570 ha (17.45%)

¹forest vegetation zone, ²area, ³percentage, ⁴altitude, ⁵mean temperature, ⁶precipitation per year, ⁷growing period, ⁸dwarf pine, ⁹spruce, ¹⁰beech-spruce, ¹¹spruce-beech, ¹²from mountain forests, ¹³from all forests, ¹⁴days

II. Přirozená, současná a cílová (diferencovaná) druhová skladba v horských lesích ČR podle lesních vegetačních stupňů (LVS) – Tree species compositions in the mountain Czech forests – natural, actual, and recommended – according to forest vegetation zones

LVS ¹	Skladba ²	SM ³		JD ⁴		BO ⁵		KOS ⁶		Ostatní jehličnany ⁷		DB ⁸		BK ⁹		BR ¹⁰		OL ¹¹		JR ¹²		Ostatní listnáče ¹³	
		(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)	(ha)	(%)
6	Přirozená ¹⁴	114 156	36,9	88 170	28,5	0	0,0	0	0,0	0	0,0	0	0,0	106 113	34,3	619	0,2	309	0,1	0	0,0	0	0,0
	Současná ¹⁵	243 977	78,5	2 815	0,9	14 726	4,8	62	0,0	9 716	3,1	340	0,1	18 964	6,1	9 529	3,1	5 287	1,7	0	0,0	4 951	1,6
	Cílová ¹⁶	213 463	69,0	55 377	17,9	742	0,2	0	0,0	0	0,0	0	0,0	39 413	12,7	309	0,1	62	0,0	0	0,0	0	0,0
7	Přirozená ¹⁴	72 772	72,9	9 583	9,6	0	0,0	0	0,0	0	0,0	0	0,0	11 979	12,0	5 490	5,5	0	0,0	0	0,0	0	0,0
	Současná ¹⁵	76 765	76,9	519	0,5	1 557	1,6	898	0,9	8 086	8,1	10	0,0	3 065	3,1	5 410	5,4	1 118	1,1	0	0,0	2 396	2,4
	Cílová ¹⁶	88 264	88,4	1 098	1,1	0	0,0	0	0,0	0	0,0	0	0,0	8 884	8,9	1 577	1,6	0	0,0	0	0,0	0	0,0
8	Přirozená ¹⁴	41 272	92,7	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0	294	0,7	2 787	6,3	0	0,0	169	0,4	0	0,0
	Současná ¹⁵	34 349	77,2	13	0,0	107	0,2	534	1,2	2 280	5,1	4	0,0	102	0,2	1 006	2,3	160	0,4	0	0,0	5 966	13,4
	Cílová ¹⁶	44 495	99,9	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0	27	0,1	0	0,0	0	0,0	0	0,0	0	0,0
9	Přirozená ¹⁴	715	12,2	0	0,0	0	0,0	5 142	87,8	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0
	Současná ¹⁵	1 751	29,9	1	0,0	11	0,2	3 966	67,7	12	0,2	5	0,1	2	0,0	56	0,8	0	0,0	0	0,0	53	0,9
	Cílová ¹⁶	773	13,2	0	0,0	0	0,0	5 084	86,8	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0	0	0,0
6 až 9	Přirozená ¹⁴	228 915	49,8	97 753	21,3	0	0,0	5 142	1,1	0	0,0	0	0,0	118 386	25,8	8 896	1,9	309	0,1	169	0,0	0	0,0
	Současná ¹⁵	355 841	77,4	3 348	0,7	16 401	3,6	5 461	1,2	20 093	4,4	360	0,1	22 133	4,8	16 001	3,5	6 565	1,4	0	0,0	13 365	2,9
	Cílová ¹⁶	346 996	75,5	56 475	12,3	742	0,2	5 084	1,1	0	0,0	0	0,0	48 324	10,5	1 887	0,4	62	0,0	0	0,0	0	0,0

Údaje podle ÚHÚL (J. Macků)

¹forest vegetation zone, ²species composition, ³spruce, ⁴fir, ⁵pine, ⁶dwarf pine, ⁷other conifers, ⁸oak, ⁹beech, ¹⁰birch, ¹¹alder, ¹²rowan, ¹³other broadleaved species, ¹⁴natural composition, ¹⁵actual composition, ¹⁶recommended composition

III. Charakteristika a rozlohy pásem ohrožení – Characteristics and area of threat zones in Czech mountain forests

Pásmo ohrožení ¹	Životnost dospělých smrkových porostů ²	Plocha ³		
		v tis. ha ⁴	procento horských lesů ⁵	procento z pásma ohrožení v ČR ⁶
A	do 20 let ⁷	29,0	6,3	100
B	21–40 let ⁸	129,0	28,1	98
C	41–60 let	301,6	65,6	71
D	61–80 let	0	0	0

Údaje podle ÚHÚL (V. Henzlík)

¹threat zone code, ²survival of mature Norway spruce stands, ³area, ⁴thousands ha, ⁵percentage from mountain forests, ⁶percentage from all CZ forests, ⁷up to 20 years, ⁸21–40 years

nutné věnovat zvýšenou péči v zájmu jeho zachování pro budoucnost. V životě země a tím i v lesním hospodářství se postupně uskutečňují zásadní změny, které byly nastartovány změnou politického systému v roce 1989. V této převoluční době vláda České republiky vyjádřila svůj vztah k lesům schválením *Základních principů státní lesnické politiky* (11. 5. 1994) a nového zákona o lesích č. 289/95 Sb. (3. 11. 1995). Lesy České republiky, s. p., v r. 1996 doplnily tyto důležité základní dokumenty vypracováním lesnické politiky LČR. Následně byl pod koordinací Ministerstva zemědělství ČR vypracován Národní lesnický program. Tyto dokumenty jsou plně v souladu s četnými mezinárodními aktivitami – např. ministerskými konferencemi o ochraně lesů v Evropě (Štrasburk 1990, Helsinky 1993, Lisabon 1998 apod.).

Horské lesy jsou významnou krajinnou složkou našeho státu. Jsou objektem zvláštního významu z hlediska ochrany přírodního prostředí, stabilizace přírodních procesů i celkové homeostáze krajiny. Kromě toho plní řadu produkčních i mimoprodukčních funkcí. Jde nejen o produkci ekologicky cenné dřevní suroviny, zvěřiny, lesních plodů apod., ale i o celospolečensky velmi důležité plnění funkcí vodohospodářských, půdoochranných, klimatickoochranných, rekreačních apod. Cílem jejich trvale udržitelného obhospodařování je proto vytvoření stabilního, kvalitního, druhově, prostorově a věkově diferencovaného lesa.

Do horských lesů v souladu s pojetím ÚHÚL (např. Plíva, 1991) řadíme lesní ekosystémy náležející do 6.–9. lesního vegetačního stupně (LVS), tj. smrkobukového, buk-smrkového, smrkového a klečového LVS. Přehled LVS a jejich zastoupení v horských lesích ČR je uveden v tab. I. Rozdíly růstových podmínek lesů jsou podrobně zmapovány systémem lesních typů (LT). LT se pak sdružují do souboru lesních typů (SLT). SLT s podobnými podmínkami pro hospodaření se slučují do hospodářských souborů, pro ně jsou pak stanoveny optimální způsoby hospodaření.

Na podkladě zastoupení LT byla v roce 1995 nově vypracována rekonstruovaná přirozená a diferencovaná (optimalizovaná cílová) druhová skladba lesů podle SLT. Druhová skladba lesních porostů je přizpůsobena lokálním přírodním podmínkám i společenským požadavkům, mezi něž patří zvyšování biodiverzity lesních ekosystémů, která podstatně zvyšuje ekologickou stabilitu lesů. Přirozená, současná a diferencovaná (cílová) druhová skladba pro horské lesy ČR je uvedena v tab. II. Diferencovaná druhová skladba je určitým optimalizovaným kompromisem mezi požadavky na zvyšování ekologické stability lesních porostů, na nepřetržité plnění všech funkcí lesa a na požadavky na produkci dřeva.

Za uplynulých 50 let se v našich lesích podíl listnatých dřevin téměř zdvojnásobil. Také v posledním desetiletí je patrný pokles zastoupení jehličnatých dřevin (s výjimkou modřinu) a tomu odpovídající pomaly, ale trvalý růst podílu listnatých dřevin. Na těchto trendech se však zčásti podílí i ústup jedle a velký podíl listnatých dřevin při obnově smrkových porostů v oblastech silně zatížených imisně ekologickými vlivy.

Všechny horské lesy v České republice jsou víceméně zatíženy znečištěním ovzduší. Pro porozumění charakteru průběhu a rozsahu poškození lesních ekosystémů je třeba monitorovat kvalitu i kvantitu a časovou dynamiku imisní zátěže, které mohou mít zásadní vliv na volbu dřevinné skladby v obnovních cílech. Při stanovení základních principů výchovy a obnovy je respektována i předpokládaná dynamika poškození lesních porostů vlivem imisní zátěže. Komplexním vyjádřením potenciálních účinků znečištění ovzduší v lesích jsou pásma ohrožení imisemi D až A (tab. III).

Zdravotní stav lesa se u nás hodnotí pozemním šetřením již od roku 1986 na monitoračních plochách v rámci programu ICP Forests. Domácí výzkumné aktivity v tomto směru jsou však již téměř o 20 let starší. V současné době se hodnocení ICP Forests provádí na monitoračních plochách rozmístěných v síti 16 x 16 km (případně 8 x 8 km) po celém území republiky. Jedním z hlavních kritérií při sledování zdravotního stavu je defoliace.

Navzdory dlouholeté tradici a současně vysoké úrovni teoretických poznatků i praktických zkušeností je dnešní situace lesního hospodářství v České republice mnohem složitější než dříve. Více než ve většině ostatních evropských států doléhá na horské lesní ekosystémy komplexní antropogenní zátěž – působená hlavně znečištěním ovzduší – a s ní spojené doprovodné stresy (např.: přemnožení hmyzích škůdců a drobných zemních hlodavců, větrné výkyvy a teplotní zvraty). Po vytěžení poškozených porostů pak nastupují problémy s obnovou lesa na rozsáhlých kalamitních holinách. Nejvíce postižené území ve střední Evropě – nazývané „černý trojúhelník“ – zahrnuje i česká pohoří Kránské hory, Jizerské hory a Krkonoše. Je zřejmé, že velkoplošné narušení lesních horských ekosystémů má negativní ekologické důsledky i v celoevropském měřítku. Právě proto požadavek na uplatňování trvale udržitelného hospodaření v lesích nabývá v posledních letech ve všech lesnických vyspělých zemích na významu a důrazu. K tomuto trendu značnou měrou přispívá i lesnický výzkum, lesnická osvěta a každodenní kvalifikovaná práce lesníků na všech stupních řízení. V České republice se v současnosti řeší množství výzkumných projektů, které má přispět k posílení stability horských ekosystémů. Pozornost se přitom věnuje jak aplikaci klasických lesnických disciplín ve specifických horských podmínkách, tak i aktivitám novodobým, které si současný neutečený stav lesů a vzrůstající požadavky na plnění mimoprodukčních funkcí vyžadují. Jedná se zejména o úkoly na úseku genetiky a fyziologie lesních dřevin, zvyšování vitality a stability lesních porostů, sledování půdních procesů a možností jejich ovlivňování, možnosti předpovídání potenciálních pozitivních i negativních vnějších vlivů na lesy, sledování změn v lesních ekosystémech, modelování budoucího vývoje ekosystému a možnosti dlouhodobé adaptace lesních dřevin i skladby lesů na vnější vlivy, oceňování lesních ekosystémů a ekonomiky přestavby na trvale udržitelné hospodaření.

Závěrem bychom chtěli zdůraznit nezbytnost zintenzivnění mezinárodní spolupráce na poli informovanosti o řešení problémů spojených s horskými lesními ekosystémy. I když výsledky výzkumu bývají většinou publikovány, jsou často těžko dostupné, zvláště pokud jsou otištěny v regionálních

nepériodických cizích zemí. Téměř nedostupné pak bývají informace o projektech, které se teprve začínají řešit. Přitom informovanost a případná spolupráce při řešení podobných

problémů by přinesly vícestranný užitek. Proto navrhujeme vytvoření mezinárodní databáze všech projektů zabývajících se uvedenou problematikou v minulosti i současnosti.

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INTERNATIONAL WORKSHOP ON MOUNTAIN FORESTS, TRENTO, ITALY

Shortly after Ministerial Conference of Lisbon, 2nd International Workshop *A European Project for Mountain Forest* took place in Trento, Italy, 24–26 September 1998. The workshop was organized by European Observatory of Mountain Forest (Saint Jean d'Arvey), Federation of European Forest Community Councils, Autonomous Province Trento, Municipality of Trento, Foundation of the Trento and Rovereto Savings Bank, Federation of Italian Forestry Communities, and Forestry Agency Trento-Sopramonte. Goal of the workshop was to reinforce the implementation of the Resolution S4 declared in Strasbourg 1990 (Adapting of the Management of Mountain Forest to New Environmental Conditions).

Final Declaration of the 2nd International Workshop held in Trento was adopted by 200 delegates from 17 countries and a number of international institutions and organisations (European Parliament, International Institute for Environment and Development IIED, Confederation of the European Agricultural Professional Organisations COPA–COGECA, WWF-Europe, European Confederation of Paper Industries CEPI, European Federation of Local Forest Authorities FECOF). The Ministerial Conference on the Protection of Forests in Europe, FAO and the International Union of Forestry Research Organisations (IUFRO), attended as observers. The adopted Final Declaration is based on the recognition of the following points:

- the increasing importance of the multifunctional role of forests in mountain areas,
- the increasing importance of mountain forests within the positions taken by international, European, national, regional and local institutions as well as the concrete commitments made by decision-makers (Alpine Convention, European Charter of Mountain Regions, Resolution S4 *Mountain Forest* of the Ministerial Conference of Strasbourg, Resolution of the European Parliament on the Forest Strategy in the European Union, Convention on Biodiversity, Pan-European Strategy on Biological and Landscape Diversity),
- the diversity of national, regional and local situations as well as the common issues linking countries, regions and local communities,
- the diversity of interests between direct and indirect actors as well as the common goals of enhancement of knowledge, conservation, management, training and information on those forests,
- the need to maximise complementarity and avoid duplication between the many organisations and institutions concerned with sustainable mountain development,
- the need to adapt measures of environmental protection, of resources development and maintenance of a viable level of local socio-economic activities following the decisions taken at the Ministerial Conferences of Lisbon, Helsinki and Strasbourg.

At the same time, the Final Declaration identifies the content of a preliminary action plan for mountain forests. Participants decided to jointly mobilise their resources in order:

1. to strengthen and implement Resolution S4 on mountain forests of the Ministerial Conference on the Protection of Forests in Europe, taking into account the Resolutions adopted by European Ministers in Lisbon, Helsinki and Strasbourg,
2. to develop and implement an action plan for the strengthening of Resolution S4, following the principles stated at point 5 of the General Declaration of Lisbon, following the outputs of the 2nd International Workshop *A European Project on Mountain Forests*, and including five actions:
 - follow the state-of-the-art to be made by the first White Book of Mountain Forest in Europe with an up-dating and Workshop every two years,
 - establish a network of communication on policies, techniques, research and training concerning sustainable management of mountain forests,
 - establish a network of sample sites in each country for the implementation and monitoring of criteria and indicators, as well as for the exchange of experiences between countries,
 - establish a referential of socio-economic and environmental data provided by countries and the sites of the network, and following the criteria and indicators endorsed in Lisbon,
 - organise training meetings between representatives of countries, based on decisions adopted at Lisbon, Helsinki and Strasbourg.

The workshop was ended by visiting of Magnifique Communauté de Fiemme in the beautiful Alpine countryside.

Among national reports, a Czech contribution *Sustainable Management and State of the Mountain Forests in the Czech Republic* was reported. It was aimed to inform about actual silvicultural problems in the Czech mountains. The following terms were discussed: forest site type classification, forest vegetation zone, edaphic categories, management groups, natural tree species composition, optimized (prescribed) tree species composition, actual (present) tree species composition, management groups, and 4 air pollution threat zones. It was stated that, in spite of the fact the idea of sustainable forest management has 200 years old tradition in the Czech Republic, adapting of our present silviculture to new environmental conditions is a very demanding task. Its achievement is conditioned by strong human activation and by progress in science and technology. At the end of the contribution, some present forest tasks and activities were mentioned and necessity of international co-operation in forest problem solution was stressed.

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