**ČESKÁ ZEMĚDĚLSKÁ UNIVERZITA**
**V PRAZE**
FAKULTA LESNICKÁ A DŘEVAŘSKÁ
KATEDRA HOSPODÁŘSKÉ ÚPRAVY LESŮ

# PROGRESÍVNE INVENTARIZAČNÉ A MONITOROVACIE POSTUPY A NOVÉ METÓDY HODNOTENIA ŠTRUKTÚRY LESNÝCH EKOSYSTÉMOV

Habilitačná práca

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Praha
2010

## ***Poďakovanie***

*Na tomto mieste by som sa chcel v prvom rade poďakovať svojej rodine, manželke Kataríne, dcéram Veronike a Katke za trpezlivosť a pochopenie. Úprimne ďakujem svojim kolegom doc. Marušákovi a doc. Fabrikovi ako aj všetkým kolegom a kolegyniam, s ktorými som mal česť spolupracovať. Ďakujem prof. Šmelkovi za to, čo som sa pri ňom naučil.*

*Ďakujem*

# Obsah

Obsah ........................................................................................................ 3
Abstract ........................................................................................................ 4
1. Úvod ........................................................................................................ 5
2. Výberové metódy – progresívny nástroj zisťovania a monitorovania lesných ekosystémov ........................................................................................................ 6
3. Podstata biodiverzity – základu štruktúry lesných porastov a jej foriem ako ekologickej funkcie lesa ........................................................................................................ 9
    3.1. Hodnotenie druhovej diverzity lesných porastov na báze výberových matematicko-štatistických metód ........................................................................ 10
    3.2. Hodnotenie štrukturálnej diverzity lesných porastov na báze výberových matematicko-štatistických metód ........................................................................ 12
    3.3. Vplyv vybraných faktorov na diverzitu lesných ekosystémov .................... 17
4. Zložky štruktúry lesa – indikátor prirodzenosti lesných ekosystémov ....................... 22
5. Záver ........................................................................................................ 27
6. Literatúra ........................................................................................................ 28
7. Príloha ........................................................................................................ 35
    7.1. Článok v kapitole 2. Výberové metódy – progresívny nástroj zisťovania a monitorovania lesných ekosystémov ........................................................ 37
    7.2. Články v kapitole 3.1. Hodnotenie druhovej diverzity lesných porastov na báze výberových matematicko-štatistických metód .................................... 45
    7.3. Články v kapitole 3.2. Hodnotenie štrukturálnej diverzity lesných porastov na báze výberových matematicko-štatistických metód ............................... 63
    7.4. Články v kapitole 3.3. Vplyv vybraných faktorov na diverzitu lesných ekosystémov ........................................................................ 101
    7.5. Články v kapitole 4. Zložky štruktúry lesa – indikátor prirodzenosti lesných ekosystémov ........................................................................ 158

# Abstract

The assessment of forest condition and development is an important part of forest management. Mathematical quantification of forest characteristics is a tool that allows a user to evaluate ecosystem objectively and also to understand and to describe the relationships in the ecosystem. The presented work is a compilation of sixteen original scientific papers that are based on mathematical statistical sampling designs and that utilise a wide range of mathematical and statistical evaluation methods including modern multidimensional methods and geostatistical approaches.

The presented results comprise the proposal of new methods of tree species diversity evaluation and its spatial interpolation, a new algorithm for the estimation of parameters of Weibull function applied to the evaluation of forest stand diameter distribution in cases when tree diameters are measured from a threshold value, which is common in forest enumeration surveys. Furthermore, the work presents new information on the evaluation of structural diversity in natural spruce forests covering the features of natural regeneration, dynamics, and ecological stability. In addition, the findings concerning the relationship of the abiotic factors to the components of tree species and structural diversity are presented. At the end of the work, the significance of forest strutural elements as indicators of forest ecosystem naturalness is analysed, and the proposal of a new model for the evaluation of forest naturalness of the forests in 7th altitudinal vegetation zone, which can be considered as a potential tool for decision- and managment-support systems, is presented.

Based on the achieved results, theoretical and practical experience it is recommended to propose a complex information spectrum when preparing forest surveys. The characteristics to be inventoried should include the features of forest structure, because structure is an important characteristic of forest stand condition and development closely connected to all processes occurring in the ecosystem. Considering the current state of information technologies as well as the state and the size of information databases, it is recommended to apply the approaches of stratified sampling design in a wider context.

**Keywords:**
Sampling design, national inventory, biodiversity, species diversity, structural diversity, discriminant analysis, Weibull function, method of moments, virgin forest, climate change, Ellenberg indicator, indicator kriging, deer, dead wood, carbon, naturalness

# 1. Úvod

Les je zložitý ekologický systém, od ktorého ľudská spoločnosť očakáva plnenie rôznorodých funkcií. Kým v minulosti bola v popredí produkčná funkcia lesa, ktorá súvisela s rýchlo napredujúcim hospodárskym rozvojom, v poslednom období kladie spoločnosť vysoké požiadavky na environmentálne funkcie lesa. S dopytom po „lese“ sa postupne rozvíja viacero lesníckych disciplín, z ktorých mnohé sa stali aj vednými odbormi, a ktorých všeobecným cieľom je zefektívniť hospodárenie v lesoch. Uvedomujúc si, že les plní množstvo nenahraditeľných funkcií, sa stalo trvalo udržateľné hospodárenie v lesoch základným princípom moderných lesných zákonov v mnohých krajinách s vyspelým lesným hospodárstvom. Hospodárska úprava lesa je disciplína, ktorá v sebe integruje poznatky z takmer všetkých lesníckych odborov a premieňa ich v plánovité hospodárenie v lesoch tak, aby sa pri zosúladení záujmov vlastníkov, obhospodarovateľov lesov a verejného záujmu smerovalo k naplneniu cieľov trvalo udržateľného hospodárenia v lesoch.

Dôležitými činnosťami hospodárskej úpravy lesov je zisťovanie stavu a vývoja lesov. Táto činnosť je rozvíjaná najmä prostredníctvom lesníckej biometrie a dendrometrie. Zloženie a štruktúra lesa určujú a predstavujú priestorovú biodiverzitu a sú dôležité pre produkciu, mimoprodukčné funkcie a trvalú udržateľnosť lesných ekosystémov. Rozvoj metód hodnotenia štruktúry lesa má preto významné postavenie vo výskume v danej oblasti, pretože matematická kvantifikácia umožňuje nielen objektívnejšie ohodnotiť, ale v konečnom dôsledku aj lepšie pochopiť a popísať vzťahy, ktoré v ekosystéme fungujú. Štrukturálna diverzita lesných porastov je veľmi dôležitou charakteristikou stavu a vývoja porastov. Úzko súvisí so všetkými procesmi prebiehajúcimi v ekosystéme. Poznanie a rešpektovanie týchto zákonitostí môže významne napomôcť lesnému hospodárovi pri obhospodarovaní lesa v meniacich sa ekologických podmienkach a zabezpečiť jeho trvalosť.

Predkladaná práca vychádza zo súboru šestnástich z tridsiatich pôvodných vedeckých prác. Vo všetkých prácach, na ktorých som sa autorsky spolupodieľal, sa využívajú matematicko-štatistické výberové postupy pri aplikácii základných, ale aj modernejších a efektívnejších výberových dizajnov. Súhrn dosiahnutých výsledkov prezentovaný v predkladanej práci má za cieľ poukázať na možnosti komplexného hodnotenia lesného ekosystému s ohľadom na široké spektrum požiadaviek spoločnosti. Tento stav vyžaduje od inventarizácie a monitoringu komplexný pohľad na lesný ekosystém zahŕňajúci nielen drevinovú zložku a jej produkciu, ktorá doteraz stála v centre pozornosti, ale aj ďalšie súčasti a charakteristiky biodiverzity lesného ekosystému vrátane štruktúry lesného porastu, regenerácie, fytocenózy, zoocenózy, stability a prirodzenosti lesných porastov ako aj abiotických faktorov prostredia. Výsledky vybraných prác prezentujú návrh nových metód pri hodnotení druhovej diverzity a jej priestorovej regionalizácie, nový algoritmus pre odhad parametrov Weibulovej funkcie pri hodnotení hrúbkovej štruktúry lesných porastov v prípadoch, keď sú hrúbky stromov merané od určitej minimálnej hranice, čo je bežný prípad pri taxácii lesných porastov. Zároveň práca prezentuje nové poznatky z hodnotenia štrukturálnej diverzity v smrekových prírodných lesoch z pohľadu regenerácie, dynamiky a ekologickej stability. Taktiež sa prezentujú nové poznatky vzťahu abiotických faktorov ku zložkám druhovej a štrukturálnej diverzity a v závere práce sa analyzuje význam zložiek štruktúry lesa ako indikátorov prirodzenosti lesného ekosystému, čo vyúsťuje do návrhu nového modelu hodnotenia prirodzenosti lesa pre 7. lesný vegetačný stupeň ako potencionálneho nástroja pre rozhodovacie a manažérske systémy.

# 2. Výberové metódy – progresívny nástroj zisťovania a monitorovania lesných ekosystémov

Lesy sú významnou súčasťou prírodného bohatstva a rozhodujúcou zložkou životného prostredia, pričom plnia nezastupiteľné celospoločenské funkcie. Z biometrického hľadiska sú tvorené veľkým množstvom rôznych jednotiek, napr. biotopov, dielcov, stromov a pod. Súhrnom informácií o ich parametroch, zmene, kvalitatívnom stave, priestorovej polohe a pod. za väčšie objekty resp. ich časti získavame podklady o stave a vývoji lesného ekosystému, ktoré sú použiteľné a dôležité pre rôzne úrovne rozhodovacieho procesu hospodárskych ako aj environmentálnych odvetví našej spoločnosti. Je zrejmé, že celoplošné zisťovanie podrobných a plnohodnotných informácií o jednotkách tvoriacich les na tak rozsiahlych výmerách ako v súčasnosti zaberajú lesy, nie je ekonomicky únosné a možné. Dopyt po informáciách o lese, t.j. po inventarizácii a monitoringu lesa, ktorých cieľom je poskytovať objektívne informácie o stave a zmenách lesných ekosystémov, má už takmer storočnú tradíciu. V 20. rokoch minulého storočia začína v škandinávskych krajinách rozvoj týchto metód na báze štatistických princípov, keď v roku 1921 zrealizovali vo Fínsku prvú národnú inventarizáciu lesa (Kangas a Maltamo 2006). O niečo neskôr v 60. rokoch zaznamenávame rozvoj monitorovacích metód, ktorý prichádza z USA (Scott 1947) a pre európske pomery ho dobre rozpracovali a aplikovali vo Švajčiarsku (Schmid 1963, 1967, Šmelko 1985). Rozvoj týchto metód rýchlo napredoval z dôvodu čoraz väčšieho záujmu o aktuálne a objektívne informácie prevažne o produkcii lesov. Zhruba od 80. rokov sa verejnosť začína zaujímať nielen o produkčnú stránku lesov, ale do popredia sa začínajú presadzovať aj ďalšie dôležité informácie o ekológii lesných ekosystémov, čím pri inventarizačných a monitorovacích zisťovaniach dochádza k rozširovaniu informačného spektra. Dopyt po väčšom počte informácií súvisí aj s rozvojom ekologicko-produkčného výskumu a hlavne modelovania.

S avizovaným nárastom záujmu o kvalitné informácie vznikal tlak na vývoj efektívnejších a racionálnejších výberových metód. Z biometrického hľadiska je podstatou výberových metód určiť (štatisticky odhadnúť) neznámy parameter – aritmetický priemer kvantitatívneho znaku $\mu$ alebo relatívny podiel kvalitatívneho znaku $\pi$ v skúmanom objekte (základnom súbore, ktorého presnú hodnotu zvyčajne nepoznáme) (Šmelko 2000). Výberovým zisťovaním dostaneme iba výberové charakteristiky týchto parametrov $\bar{x}$ a $\bar{w}$. Tie sa vždy do určitej miery odlišujú od $\mu$ a $\pi$, pretože majú charakter náhodnej premennej. Z toho dôvodu sú výberové charakteristiky zaťažené chybami (z merania, z reprezentácie) a preto je informácia o ich presnosti dôležitým poznatkom. Číselným ukazovateľom presnosti určenia parametra $\mu$ a $\pi$ pomocou výberového $\bar{x}$, $\bar{w}$ je chyba odhadu, ktorú možno symbolicky zapísať

$$\text{pre} \quad \mu : \Delta\bar{x} = t_{\alpha/2} \cdot s_{\bar{x}} \qquad \Delta\bar{x}\% = \frac{\Delta\bar{x}}{\bar{x}} \cdot 100 \tag{1}$$

$$\text{pre} \quad \pi : \Delta\bar{w} = t_{\alpha/2} \cdot s_{\bar{w}} \qquad \Delta\bar{w}\% = \frac{\Delta\bar{w}}{\bar{w}} \cdot 100 \tag{2}$$

Táto chyba udáva teoretický rámec, ktorý by skutočná odchýlka ($\bar{x}$ - μ), resp. ($\bar{w}$ - π), nemala prekročiť so zvolenou pravdepodobnosťou $P$ (pre $P$ = 95% je $t_{\alpha/2}$= 1,96 resp. 2,0 pri $n > 30$). Stredná chyba výberového priemeru, resp. podielu $s_{\bar{x}}, s_{\bar{w}}$ predstavuje teoretický rámec pre výskyt skutočnej odchýlky ($\bar{x} - \mu = \Delta\bar{x}$) resp. ($\bar{w} - \pi = \Delta\bar{w}$) s pravdepodobnosťou 68%.

Veľkosť strednej chyby $s_{\bar{x}}, s_{\bar{w}}$ závisí od dvoch činiteľov:

- od variability (smerodajnej odchýlky) zisťovanej veličiny, s nárastom ktorej sa chyba zväčšuje;
- od rozsahu výberu $n$, t.j. od počtu jednotiek – stromov, skusných plôch, traktov, na ktorých sa zisťovanie uskutočnilo a s rastom ktorého chyba klesá úmerne k $\sqrt{n}$.

Výsledný odhad parametra $\mu$ a $\pi$ so zvolenou spoľahlivosťou *P* sa zvyčajne zapisuje vo forme intervalu spoľahlivosti platného pre pravdepodobnosť *P%*:

$$\mu = \bar{x} \pm \Delta\bar{x}\,; \qquad \pi = \bar{w} \pm \Delta\bar{w} \tag{3}$$

V biologických disciplínach vznikol celý rad výberových metód, ktoré sa medzi sebou líšia dizajnom. Ten je prispôsobený účelu zisťovanej veličiny. Výberový dizajn je súhrn rozhodnutí o druhu výberových jednotiek, ich rozmiestnení a počte, čo má rozhodujúci vplyv na presnosť, náklady a organizáciu výberového zisťovania. V lesníckych aplikáciách môže byť výberovou jednotkou strom, skusná plocha, porast, línia a pod.

Cieľom mnohých výberových dizajnov je dostatočne presne charakterizovať záujmové územie z pohľadu stavu resp. vývoja hlavných veličín. Iný variant výberových dizajnov sa odvíja od snahy získať reprezentatívne údaje ako podklad pre špeciálne analýzy a modelovanie. Oba koncepty majú svoje výhody i nevýhody. V prvom prípade získame údaje, ktoré reprezentujú prevládajúce ekologické formy na danom území, na strane druhej takéto údaje nemusia byť vhodným podkladom pre špeciálne účely, napr. modelovanie, pretože nevystihujú plnú šírku skúmaného ekologického gradienta. V druhom prípade máme relatívne homogénne údaje popisujúce stav resp. zmeny veličín pozdĺž zvoleného ekologického gradientu, ale zároveň sa stráca možnosť zovšeobecnenia informácií za hodnotené územie. Z viacerých hľadísk sa preto javí potreba integrácie uvedených konceptov a výberové dizajny pripravovať tak, aby spĺňali viacrozmerné využitie. Efektívnym spôsobom je aplikácia špeciálnych výberových postupov, ktoré sú už dnes dobre známe a dobre matematicko-štatisticky prepracované.

Jednou z možností je aplikácia stratifikovaného výberu resp. jeho rôznych modifikácií. Základným črtou tohto výberu je, že sa základný súbor rozdelí na menšie homogénne sub-populácie (sub-jednotky), ktoré sa nazývajú stratá. Tieto stratá sa navzájom neprekrývajú a teda ich suma predstavuje základný súbor. Následne sa v každom strate robí samostatný výber, t.j. každé stratum predstavuje nový základný súbor, ktorý sa hodnotí zvlášť. Výhodou stratifikovaného výberu oproti jednoduchému výberu je, že sa zníži stredná výberová chyba. Navyše sa takýmto spôsobom získajú odhady pre každé stratum ako aj pre celý základný súbor. Stratifikáciu je možné robiť viacerými spôsobmi (priamo pomocou hodnotenej veličiny, pomocou indikačných veličín, prostriedkami DPZ a GIS, v teréne a pod.), pričom k tomu môžeme využiť rôzne zdroje, ktoré budú zahŕňať aj stratifikáciu v smere hlavných ekologických gradientov, ako sú napr. primárne veličiny teplota, zrážky a pod. Výhodou je, ak je vypovedacia schopnosť strát približne rovnaká, čo môžeme zabezpečiť kombináciou viacerých monitorovacích systémov diferencovaných podľa veľkosti a charakteru strát. Dôležité je tiež, aby výberová jednotka, napr. skusná plocha, mala homogénny charakter, t.j. mala by plne reprezentovať vylíšené stratum. Výhodou takéhoto dizajnu je zachytenie informácií pozdĺž celého gradientu faktora (-ov), čo je pri ekologických štúdiách nezanedbateľné a získané údaje sú vhodným podkladom pre špeciálne analýzy. Súčasne nestrácame možnosť charakterizovať a zovšeobecniť informácie na územie ako celok. Nevýhodou je potreba detailnej analýzy hlavných ekologických faktorov, ktoré ovplyvňujú produkciu, zdravotný stav i ekológiu lesných ekosystémov a zároveň je potrebné poznať podiel jednotlivých strát.

Ďalším efektívnym výberovým dizajnom sú viacfázové výbery. Viacfázový výber sa skladá z viacerých fáz, ktoré na seba nadväzujú. Najčastejší je dvojfázový výber, kedy sa v prvej fáze zozbiera veľké množstvo informácií jednoduchým, časovo a peňažne efektívnym spôsobom, napr. informácie sa odhadnú, alebo sa meria náhradná ľahšie zistiteľná premenná. V druhej fáze sa náhodným spôsobom vyberie sub-vzorka zo vzorky z prvej fázy, na ktorej sa zisťujú informácie presnou metódou. Prvá a druhá fáza sú teda závislé. Pomocou údajov získaných z prvej a druhej fázy sa odvodí regresia alebo pomer, ktoré sa následne používajú na odhad hodnoty skúmanej veličiny, príp. na korekciu údajov z prvej fázy.

Aplikáciu výberových postupov a metód môžeme nájsť vo všetkých prácach, na ktorých sa habilitant autorsky spolupodieľal. V práci uvedenej v prílohe P01 [P01]**Šmelko – Merganič (2008)** sa prezentuje základná koncepcia Národnej inventarizácie a monitoringu lesov (NIML) Slovenska, ktorá sa po prvýkrát uskutočnila v rokoch 2005–2006. Príspevok opisuje niektoré metodické princípy terénneho zberu údajov (určovanie výšok stromov dvojfázovou metódou, regresné rovnice uplatnené pri stanovení objemu a sortimentácii stromov lesných drevín, spôsob kvantifikácie objemu mŕtveho dreva na skusných plochách) a biometrické modely pripravené pre spracovanie údajov a zovšeobecnenie výsledkov. Výberový dizajn a celá koncepcia NIML boli navrhnuté tak, aby umožňovali vo zvolených časových intervaloch poskytovať komplexné a integrované informácie o stave a zmenách produkčných a ekologických charakteristík lesných ekosystémov na celoštátnej i regionálnej úrovni.

Filozofia a uplatňovanie efektívnych výberových dizajnov bola aplikovaná aj vo viacerých projektoch habilitanta, v roku 2003 pri monitoringu diverzity horských lesov severnej Oravy (Merganič et al. 2003), v roku 2005 pri výskume reakcie diverzity lesných fytocenóz na zmenu edaficko-klimatických podmienok Slovenska (Vladovič at al. 2008), v roku 2008 pri monitorovaní stavu lesných biotopov európskeho významu v š.p. Lesy SR (Merganič a Šmelko 2008a, 2008b, 2009) a v roku 2009 pri návrhu komplexného nepeňažného a ekonomického ohodnotenia biodiverzity ako základného potenciálu funkcií lesa (Marušák et al. 2009).

# 3. Podstata biodiverzity – základu štruktúry lesných porastov a jej foriem ako ekologickej funkcie lesa

Odkedy sa pojem biologická diverzita (skrátene biodiverzita) v roku 1972 prvýkrát objavil v názvoch článkov v odbornej literatúre (Kaennel 1998), stal sa jedným z najfrekventovanejších pojmov našej planéty. Kým v 70. a začiatkom 80. rokov označoval tento termín „počet prítomných druhov“ (Christie et al. 2004), v súčasnosti existuje množstvo rôznych formálnych a neformálnych definícií pojmu biodiverzita. Už v roku 1990 Noss (1990) konštatoval, že pod pojmom biologická diverzita rozumejú rozliční ľudia rozličné veci. „Systematik môže biologickú diverzitu chápať ako zoznam druhov určitého taxónu alebo skupiny taxónov. Genetik môže uvažovať o alelickej diverzite a heterozygotnosti, kým fytocenológ sa viac zaujíma o rozmanitosť a rozmiestnenie druhov a vegetačných typov“ (Noss 1990). Najčastejšie sa však autori odvolávajú na Dohovor o biologickej diverzite (Convention on Biological Diversity), ktorý definuje biologickú diverzitu ako „rozmanitosť a rôznorodosť všetkých živých organizmov vrátane ich suchozemských, morských a ostatných vodných ekosystémov a ekologických komplexov, ktorých sú súčasťou“. Uvedená definícia v podstate vymedzuje tri základné zložky biologickej diverzity: genetickú, druhovú a ekosystémovú diverzitu (Duelli 1997 in Larsson 2001, Kaennel 1998, Subade 2005). Genetická diverzita zodpovedajúca variabilite v rámci druhu je najzákladnejšou úrovňou (Nunes a Bergh 2001) a v podstate sa týka informácií obsiahnutých v génoch DNA jednotlivých druhov (Wilson 1994 in Nunes a Bergh 2001). Môže byť však vnímaná aj ako variabilita medzi populáciami (Christie et al. 2004). Na úrovni planéty Zem predstavuje genetická diverzita sumu genetických informácií zakódovaných v génoch všetkých živých organizmov (Brown et al. 1993). Druhová diverzita označuje diverzitu medzi druhmi na určitom území. Pod termínom druh sa rozumie taká populácia, v rámci ktorej dochádza k prirodzenému toku génov (Brown et al. 1993). V situáciách, kedy je ťažké určiť hranicu medzi genetickou a druhovou diverzitou, sa odporúča rozlišovať fenotypovú a genotypovú diverzitu (Nunes a Bergh 2001). Ekosystémová diverzita sa vzťahuje na nad-druhovú úroveň a vyjadruje diverzitu spoločenstiev organizmov v určitých stanovištných podmienkach (Nunes a Bergh 2001). Väčšinou sa hodnotí na úrovni krajiny a regiónu (Tacconi a Bennett 1995).

Hoci sa táto trilógia komponentov biodiverzity stala štandardom uznávaným širokým okruhom vedcov, Noss (1990) a Redford a Sanderson (1992 in Kaennel 1998) poukázali na to, že definícia podľa Dohovoru neuvažuje s ekologickými procesmi a časovým a priestorovým hľadiskom, ktoré hrajú v procese zachovania biodiverzity dôležitú úlohu. Preto niektorí autori vylišujú aj ďalšie zložky: napr. Nunes a Bergh (2001) priraďuje k trom štandardným elementom aj funkčnú diverzitu, ktorú definujú ako schopnosť ekosystému odolať určitej hladine stresu alebo šoku bez toho, aby došlo k zmene jeho stavu a správania. Christie et al. (2004), ktorý tiež rozoznávajú funkčnú (resp. ekologickú) diverzitu, pod ňou rozumejú komplex ekosystémových procesov prebiehajúcich v spoločenstve ako dôsledok prítomnosti jednotlivých funkčných skupín organizmov. Brown et al. (1993), Pearce a Moran (1994) a iní však považujú diverzitu ekologických procesov za prvok ekosystémovej diverzity. Podobne ako Christie et al. (2004) vnímajú funkčnú diverzitu aj Hammer et al. (1993 in Smith 1996), ktorí ju definujú ako diverzitu funkcií jednotlivých druhov v ekosystéme. Títo autori navyše rozlišujú aj tzv. priestorovo-časovú diverzitu zahŕňajúcu topografiu, klímu apod. Iní rozoznávajú kultúrnu diverzitu, diverzitu správania, umelú diverzitu atď.

Ako vyplýva z uvedeného prehľadu, vytvoriť jednoduchú a pritom súhrnnú definíciu, ktorá by uspokojila všetky strany, je nemožné. Preto Noss (1990) navrhol, že namiesto definovania biodiverzity je vhodnejšie ju charakterizovať na základe jej hlavných zložiek. Franklin et al. (1981) rozlíšili tri základné atribúty ekosystémov: zloženie, štruktúra, funkcia (in Noss 1990). ”Zloženie sa týka zhodnosti a rozdielnosti prvkov v súbore a zahŕňa zoznam druhov a miery druhovej a genetickej diverzity. Štruktúra je fyzické usporiadanie systému počínajúc od komplexnosti stanovišťa meranej v rámci spoločenstva až po mozaiku spoločenstiev a iných prvkov na úrovni krajiny. Funkcia zahŕňa ekologické a evolučné procesy vrátane genetických tokov, narušení ekosystémov a cyklu živín” (Noss 1990). Zloženie a štruktúra teda určujú a predstavujú priestorovú biodiverzitu a sú dôležité pre produktivitu a trvalú udržateľnosť lesných ekosystémov, kým pod funkčnou diverzitou Larsson (2001) rozumie diverzitu ekologických funckií a diverzitu druhov, ktoré dané funkcie vykonávajú. Vychádzajúc z tohto ponímania pojmu biodiverzita sa vytvára hierarchický systém na viacerých úrovniach priestoru (populácia, druh, biocenóza, stanovište, krajina), zloženia (suma prvkov genómu, populácie, druhového spoločenstva, ekosystému alebo krajiny) a funkcií (procesy, ktoré prebiehajú na rozličných úrovniach hierarchického systému) (Büchs 2003). Podobne na základe koncepcie Noss-a (1990) navrhol Marcot (2007) vnímať biodiverzitu na troch organizačných úrovniach: gén, populácia/druh, spoločenstvo/ekosystém, a v troch dimenziách: zloženie, štruktúra a funckia (in Charnley et al. 2007).

## 3.1. Hodnotenie druhovej diverzity lesných porastov na báze výberových matematicko-štatistických metód

Kvantifikácia jednotlivých zložiek biodiverzity v lesníckych a poľnohospodárskych aplikáciách sa v prevažnej miere zameriava na vyššie rastliny. Hodnotenie druhovej diverzity je založené na minimálne jednom z troch nasledujúcich znakov (Bruciamacchie 1996), a to na:

- druhovej bohatosti, ako na najstaršom a najjednoduchšom poňatí druhovej diverzity vyjadrenej iba na základe počtu druhov;
- druhovej vyrovnanosti, ako miere rovnomernosti zastúpenia jednotlivých druhov v spoločenstve a
- druhovej heterogenite, ako charakteristike zahŕňajúcej v sebe druhovú bohatosť a vyrovnanosť v jednom.

Pre popis a kvantifikáciu jednotlivých znakov druhovej diverzity je vypracovaných viacero techník a metód. Napr. pri kvantifikácii druhovej bohatosti môžeme siahnuť po metódach založených od najjednoduchšieho vyjadrenia počtu druhov až po zložité matematické výrazy a procedúry ako Rarefraction metóda (Sanders 1968), Jackknife odhad (Heltsh a Forrester 1983), Bootstrap procedúra (Smith & Van Belle 1984), metóda logaritmického radu (Fischer et al. 1943), metóda lognormálneho rozdelenia (Preston 1984 in Krebs 1989).

Medzi ďalšie metódy vypracované k hodnoteniu druhovej diverzity môžeme zaradiť aj metódu diverzitných profilov (Gove et al. 1996), ktoré umožňujú grafické porovnanie diverzity medzi spoločenstvami.

Pravdepodobne najpopulárnejšia a najčastejšie používaná je metóda indexov. Počas ich historického vývoja sa postupne vyšpecifikovali tri samostatné skupiny, t.j. indexy druhovej bohatosti (napr. *N0* (Hill 1973), *R1* (Margalef 1958), *R2* (Menhinick 1964)), indexy druhovej heterogenity (napr. Simpsonov index $\lambda$ (Simpson 1949), Shannonov index *H‘* (Shannon a Weaver 1949) a Hillove čísla *N1* a *N2* (Hill 1973)) a indexy druhovej vyrovnanosti (napr. *E1* (Pielou 1975), *E3* (Heip 1974), *E5* (Hill 1973)). Indexy diverzity predstavujú presne definované a teda objektivizované parametre, ktoré umožňujú exaktne popísať nájdenú a pozorovanú diverzitu. Pri periodických zisťovaniach umožňujú spolu so súpisom zaznamenaných druhov efektívne sledovať a vyhodnocovať zmeny, ktoré nastali v sledovanom území. Možno ich využiť v riadiacom a rozhodovacom procese v lesnom hospodárstve, napr. na kontrolu výsledkov hospodárskych opatrení alebo jednoducho na zachytenie informácií pre vyhodnotenie budúcich zmien v druhovej diverzite.

Problematiku hodnotenia druhovej diverzity na báze výberových metód môžeme nájsť vo viacerých prácach habilitanta. V práci [P02]**Merganič – Šmelko (2004)** sa popisuje nová metóda komplexného hodnotenia a kvantifikácie druhovej diverzity v lesných ekosystémoch. Bežne používané indexy druhovej diverzity odrážajú len jednu zo zložiek druhovej diverzity a to buď druhovú bohatosť alebo heterogenitu či vyrovnanosť. Ich integráciou bol navrhnutý model BIODIVERSS, ktorým sa určuje stupeň druhovej diverzity na základe piatich indexov diverzity (*R1, R2, λ, H'* a *E1*). Pravdepodobnosť správnej klasifikácie stupňa druhovej diverzity pomocou modelu BIODIVERSS je pomerne vysoká. Už pri intenzite výberu cca 1.5% dosiahne približne 90% úspešnosť.

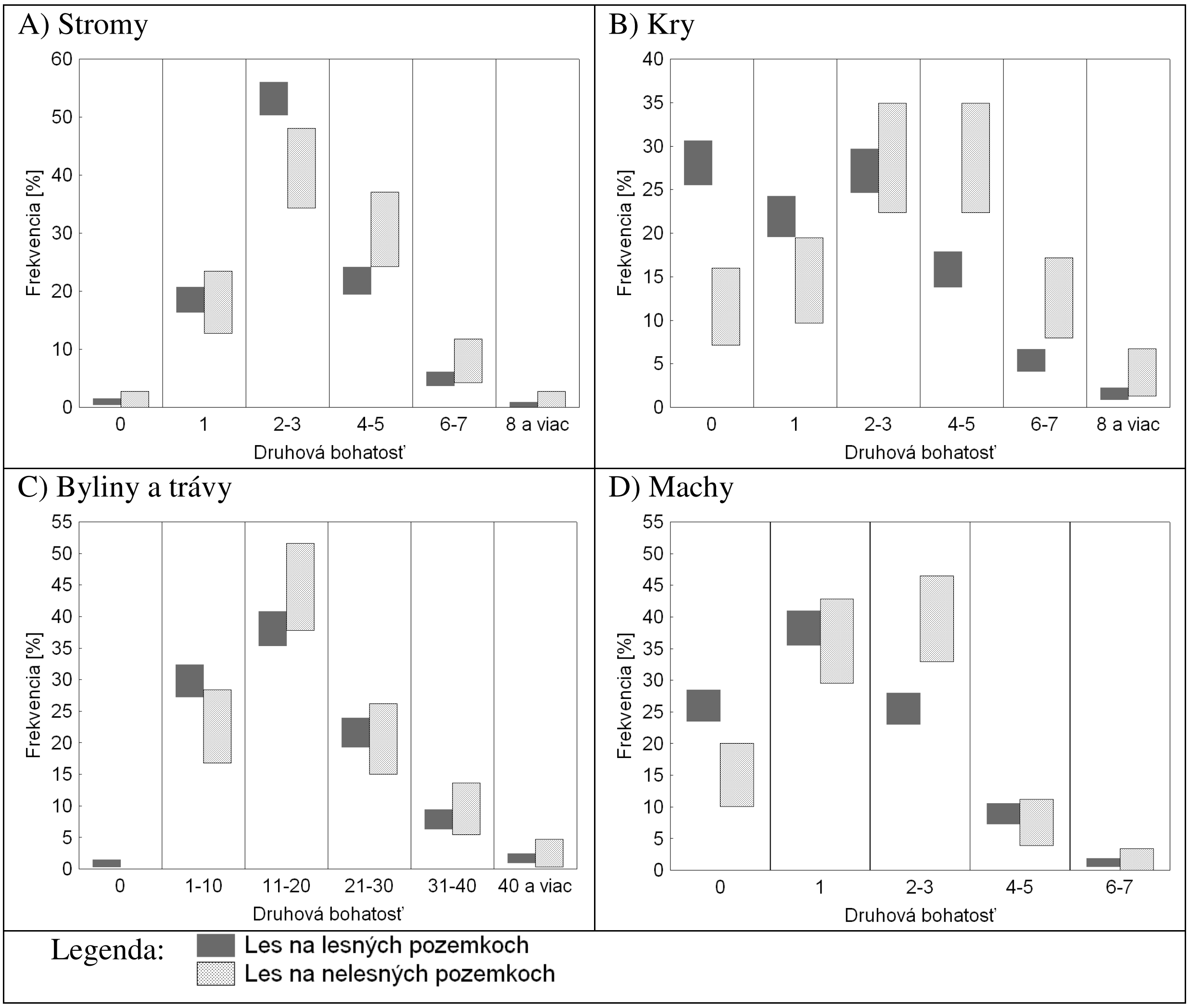
Navrhovaný model BIODIVERSS je konštruovaný so širokou valenciou a hoci je vytvorený iba na báze štyroch reprezentatívnych porastov, v skutočnosti je zostavený na podklade 865 optimálne veľkých skusných plôch. Využitie optimálne veľkých skusných plôch pre hodnotenie druhovej diverzity eliminuje dobre známy problém silnej závislosti druhovej diverzity od plochy hondoteného objektu, keďže v takomto prípade daný kvantifikátor vždy reprezentuje skupinu o rovnakom počte stromov (približne 20). Preto je správna aplikácia modelu BIODIVERSS podmienená využitím údajov z optimálne veľkých skusných plôch.

Model BIODIVERSS je jednoduchý a praktický. Môže sa použiť priamo pri hodnotení v teréne, pretože jeho aplikácia vyžaduje len vreckovú kalkulačku. Preto by jeho zavedenie do praxe v rámci taxácie lesov nemalo spôsobovať žiadny technický problém.

Model BIODIVERSS slúži na určovanie biologickej druhovej diverzity stromovej vrstvy najmä na maloplošnej (porastovej) úrovni.Tento model možno však s výhodou použiť aj pri regionálnych a veľkoplošných inventarizáciách, ak predpokladáme, že stupeň druhovej diverzity určený na skusnej ploche reprezentuje určitú časť hodnoteného územia. Ďalšie zhodnotenie už podlieha bežným matematicko-štatistickým postupom.

V širšom zmysle predstavuje navrhovaný model BIODIVERSS návrh metodiky tvorby modelu, ktorý by mohol platiť pre celoeurópske pomery. Na jeho odvodenie by bolo potrebné spracovať údaje z viacerých porastov z rôznych častí Európy, pričom parametrizačnú vzorku pre každý stupeň druhovej diverzity by tvorilo niekoľko porastov. Vytvoril by sa tak jednotný systém pre posudzovanie druhovej diverzity, kde na základe jednoznačne kvantitatívne určených znakoch (indexoch diverzity) dostaneme jednoduché kvalitatívne stanovisko ľahko interpretovateľné aj pre menej zainteresované osoby. V oblasti práce s verejnosťou by to zohralo veľmi dôležitú úlohu napr. pri implementácii Dohovoru o biologickej diverzite.

Aplikácia metodických princípov habilitanta k hodnoteniu druhovej diverzity našla upotrebenie aj pri realizácii Národnej inventarizácie a monitoringu lesov SR 2005-2006. V práci [P03]**Šebeň – Merganič – Bošeľa (2008)** sa analyzujú výsledky Národnej inventarizácie a monitoringu lesov (NIML) Slovenskej republiky z hľadiska bohatosti rastlinných druhov ako významnej zložky biologickej diverzity. Druhová bohatosť sa zhodnotila pre lesy na lesných a nelesných pozemkoch, v lesných vegetačných stupňoch a integrovaných rastových stupňoch. V triediacich kategóriách sa stanovil aj celkový počet druhov. Výsledky sa uvádzajú vrátane 95% intervalov spoľahlivosti pravdepodobnosti výskytu počtu druhov pre stromy s hrúbkou nad 7 cm, krov, bylín a machov (Obrázok 1). Výsledky poukazujú na pomerne hodnotné zistenia druhovej bohatosti slovenských lesov.



**Obrázok 1** **95%-ný interval spoľahlivosti podielu tried druhových početností a ich porovnanie podľa kategórií pozemkov (Šebeň et al. 2008)**

## *3.2. Hodnotenie štrukturálnej diverzity lesných porastov na báze výberových matematicko-štatistických metód*

Hodnotenia založené len na podklade druhovej diverzity môžu byť zavádzajúce, pretože niektoré ekosystémy, ktoré sú prirodzene rovnorodé alebo tvorené prevažne jedným druhom, môžu z hľadiska biodiverzity vykazovať rovnako vysokú vážnosť ako silne druhovo heterogénne spoločenstvá. Preto Franklin et al. (1981) rozoznávajú okrem druhového zloženia ďalšie dve primárne charakteristiky ekosystémov, a to štruktúru a funkciu, ktorým sa však v rámci štúdia diverzity venuje oveľa menej pozornosti (Franklin 1988 in Noss 1990). Kým „funkcia“ zahŕňa ekologické a evolučné procesy vrátane génových tokov, prírodných katastrof a kolobehu živín, pod štruktúrou sa rozumie hmotné usporiadanie systému (Noss 1990). Podobne definujú štruktúru aj iní autori, napr. Gadow (1999) ju charakterizuje ako špecifické usporiadanie prvkov v systéme, kým Heupler (1982 in Lübbers 1999) ako ich umiestnenie a vzájomné prepojenie. Vychádzajúc z tejto všeobecnej definície sa štruktúra lesa definuje ako priestorové rozmiestnenie biomasy, čiže stromov a ich znakov (hrúbka, výška a pod.) (Zenner 1999, Gadow 1999, Gleichmar a Gerold 1998), resp. Lexer et al. (2000) hovoria o zoskupení biotických a abiotických zložiek v lesnom ekosystéme.

Štruktúru lesa je možné vnímať na viacerých hierarchických úrovniach (Kint et al. 2000). Na úrovni krajiny je definovaná ako rozrôznenie porastových typov (O´Hara 1998) a stupeň fragmentácie biotopov (Andrén 1994). Pre výskum biodiverzity sa však za dôležitejšiu považuje porastová úroveň (Kuuluvainen et al. 1996), pretože plošné a vertikálne rozmiestnenie stromov v poraste definuje trojdimenzionálny priestor biotopu vtákov, hmyzu, cicavcov, epifytov, nedrevnej vegetácie a pôdnych mikroorganizmov (Ratcliffe et al. 1986, Kuuluvainen et al. 1996). Porastová štruktúra človekom nenarušených lesných ekosystémov v sebe navyše nesie informácie o dynamike vývoja pralesov (Hofgaard 1993). V hospodárskych lesoch je však táto veličina nemenej významná, pretože napomáha zhodnotiť ich súčasný stav, ich tzv. blízkosť k prírode a v konečnom dôsledku aj ich ekologickú stabilitu (Pretzsch 1998, Zenner 1999).

Štruktúra lesa sa popisuje rôznymi štruktúrnymi prvkami, ako je priestorové rozdelenie, hustota, diferencovanie, zmiešanie (Zenner 1999). Pri správnom hodnotení štruktúry porastov je však potrebné ju vnímať komplexne ako charakteristiku zloženú z:

- horizontálnej štruktúry, t.j. plošného, dvojdimenzionálneho rozmiestnenia stromov po ploche porastu
- a vertikálnej štruktúry vyjadrenej rozrôznením porastu vo vertikálnom smere (Jaehne a Dohrenbusch 1997).

Lübbers (1999) k týmto dvom zložkám pridáva ešte aj tzv. mikroštruktúry, pod ktorými rozumie napr. množstvo odumretého dreva v poraste, formy stromov a pod.

Gadow a Hui (1999) rozlišujú tri základné znaky štruktúry:

1. pozícia, teda priestorové rozdelenie stromov po ploche lesa,
2. zmiešanie, t.j. rozmiestnenie jednotlivých druhov drevín po ploche porastu,
3. diferenciácia, t.j. charakteristiku rozdielov v rozmeroch medzi susediacimi jedincami v horizontálnom i vertikálnom smere.

Na kvantifikáciu štruktúry, jej vyššie uvedených zložiek ako aj jej rozrôznenia existuje viacero metód. Najstarší, najjednoduchší a v praxi najviac využívaný je slovný kvalitatívny popis zmiešania, hustoty a iných charakteristík porastu. Existujú však aj rôzne grafické metódy, napr. grafy rozdelenia hrúbkových, výškových početností, plány rozmiestnenia stromov, či kvantitatívne metódy (indexy, štatistické metódy). Hoci sú prvé dve skupiny metód pomerne jednoduché, zvyčajne nie je pomocou nich možné popísať rozdiely v štruktúre do jemných detailov. Tento nedostatok sa snažia odbúrať kvantitatívne metódy. Čiastočné prehľady týchto metód uvádza Pielou (1977), Füldner (1995), Gleichmar a Gerold (1998), Kint et al. (1999), Lübbers (1999), Gadow a Hui (1999), Neumann a Starlinger (2001), Pommerening (2002).

Indexy, t.j. metódy kvantifikujúce diverzitu za pomoci rozličných štatistických a matematických prístupov, predstavujú najväčšiu skupinu kvantitatívnych metód. Popisujú buď jednu alebo viac zložiek diverzity súčasne. Pravdepodobne najznámejším štrukturálnym indexom je agregačný index *R* navrhnutý Clark a Evans-om (1954), ktorý popisuje horizontálne rozmiestnenie stromov. Iné známe indexy navrhli napr. Cox (1971), Pielou (1959, 1977), Gadow (1993), Pretzsch (1996, 1998), Ferris-Kaan et al. (1998), Neumann a Starlinger (2001), Holdridge (1967), Jaehne a Dohrenbusch (1997), Zenner (1999), Zenner a Hibbs (2000), atď. Posledné štyri spomínané sú komplexné štrukturálne indexy popisujúce viacero štrukturálnych zložiek spolu, kým predchádzajúce charakterizujú len jednu časť.

Pri popise porastovej štruktúry môžu byť nápomocné aj rôzne priestorové štatistické metódy (napr. geoštatistika) (Kint et al. 2000). Niektoré z týchto metód a ich použitie prezentujú Biondi et al. (1994), Kuuluvainen et al. (1996), Goovaerts (1997), Mateu a Ribeiro (1998) a Kint (1999).

Jeden z veľmi dôležitých prvkov charakteristiky štruktúry lesného porastu je tvar rozdelenia hrúbok resp. tvar hrúbkových početností porastu. Zároveň je tvar hrúbkových početností resp. ich model jedným z najdôležitejších komponentov stromovo orientovaných rastových simulátorov. Precíznosť odhadu parametrov modelu hrúbkových početností bola a je záujmom pozornosti odborníkov. V práci [P04]**Merganič – Sterba (2006)** sa prezentuje nový algoritmus pre odhad parametrov Weibulovej funkcie v prípadoch, keď sú hrúbky stromov merané od určitej minimálnej hranice, čo je bežný prípad pri taxácii lesných porastov. Algoritmus je postavený na metóde momentov. Základná rovnica kompletnej dvojparametrovej Weibulovej funkcie napísaná metódou momentov má nasledovný tvar (Weisstein 2003):

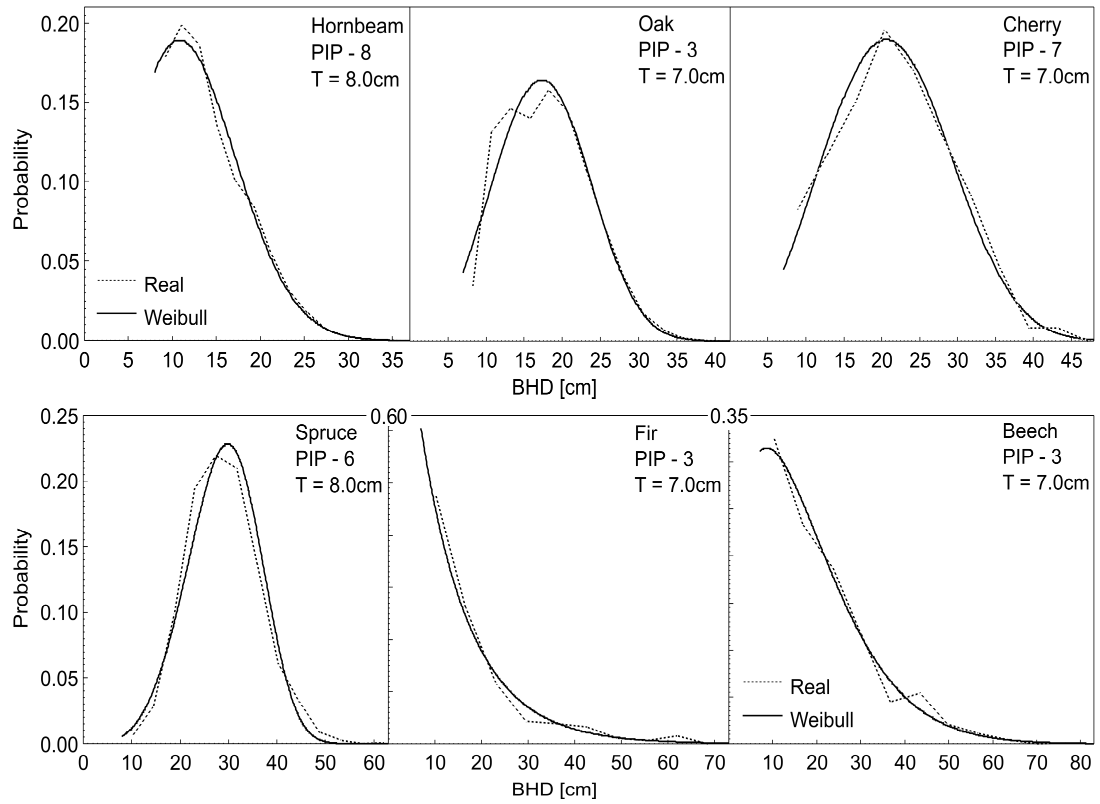
$$M(r) = b^r \cdot \Gamma\left(1 + \frac{r}{c}\right) \tag{4}$$

kde: *r* znamená poradie – rad momentu *M*, *b* a *c* sú parametere Weibulovej funkcie a $\Gamma()$ je Gamma funkcia.
Momentový zápis ľavostranne orezanej funkcie je nasledovný:

$$M(r) = \int_T^{\infty} x^r \cdot \left( \begin{cases} 0 & if \quad 0 \le x < T \\ \dfrac{f(x)}{\int_T^{\infty} f(x)dx} & if \quad T \le x \end{cases} \right) = \int_T^{\infty} x^r \cdot f_T(x)dx \tag{5}$$

Podrobná rešerš literatúry potvrdila, že metóda momentov je jedna z najpresnejších metód odhadu parametrov Weibullovej funkcie (Al-Fawzan 2000, Nanang 1998, Ueno a Ōsawa 1987, Shifley a Lentz 1985). Navyše, ako bolo ukázané v prácach Shifley a Lentz (1985) a Merganič – Sterba (2006), výhodou metódy momentov sú minimálne požiadavky na údaje. Na vytvorenie hrúbkového rozdelenia stačí poznať strednú hrúbku a variačný koeficient hrúbok. Naše analýzy zároveň ukázali, že sumárne štatistiky výsledného rozdelenia odvodeného použitím uvedenej metódy sa rovnajú nameraným hodnotám. Z toho dôvodu by sa nemalo stať, že skutočná stredná hrúbka, ktorá do modelu vstupuje, sa bude signifikantne odlišovať od strednej hrúbky vypočítanej z generovaných údajov. Iné metódy odhadu parametrov, napr. regresné metódy, ktoré sú založené na vzťahu medzi parametrami Weibullovej funkcie a vybranými porastovými charakteristikami (najčastešie stredná hrúbka, maximálna alebo minimálna hrúbka, stredná výška atď.), môžu viesť k vychýleným odhadom, keďže vzťahy týchto premenných nie sú najmä s parametrom *c* veľmi tesné (Nagel and Biging 1995, Biging et al. 1994, Little 1983, Van Laar and Mosandl 1989, Clutter and Belcher 1978).

Použitie metódy momentov je výhodné aj z toho dôvodu, že zjednodušuje parametrizáciu distribučnej funkcie. Ak zoberieme do úvahy štatistický charakter meraných premenných, musíme si uvedomiť, že skutočné hodnoty strednej hrúbky a variačný koeficient sú len odhadované, keďže sú vypočítané z výberov. Napriek tomu sú tieto odhady vždy lepšie ako odhady maximálnej hrúbky. Údaje reprezentujúce celú populáciu sú zriedka dostupné. Preto sú merané údaje zvyčajne ovplyvnené chybou z výberu. V niektorých prípadoch, keď údaje pochádzajú z viacvrcholového rozdelenia alebo keď výberový odhad nie je dostatočne konzistentný, sa Weibullová funkcia nespráva logicky. Napr. pri simulácii hrúbok môžeme dostať nelogické (veľmi vysoké) hodnoty. Ak používame veľmi malú pravdepodobnosť *F(x)* alebo $F_T(x)$, ktorá zodpovedá simulácii 10,000,000 stromov, by v rovniciach (4) alebo (7) uvedených v práci Merganič – Sterba (2006) modelovaná maximálna hrúbka nemala presiahnuť dvojnásobok nameranej maximálnej hrúbky. Ak je výsledok z rovníc (4) alebo (7) mimo preddefinovaného rozsahu, hodnota by sa mala vylúčiť a mala by sa generovať nová hodnota. Avšak, takáto situácia nastane iba zriedka a neovplyvní štatistické charakteristiky funkcie. Výhodou tejto metódy oproti iným metódam odhadu je jej univerzálnosť a fakt, že nevyžaduje špeciálne parametrizácie, napr. pre jednotlivé druhy drevín (Obrázok 2). Napr. Nagel a Biging (1995), ktorí použili regresnú metódu, stanovili parametre Weibullovej funkcie pre každú drevinu zvlášť. Práca tiež prezentuje nový algoritmus odhadu parametrov v prípadoch, keď sú hrúbky stromov merané od určitej hraničnej hrúbky. Ako poukázali Zutter et al. (1986), použitie kompletného Weibullového rozdelenia na popis orezaných údajov môže v parametroch spôsobiť veľké systematické chyby. Táto chyba rastie, ak je hodnota strednej hrúbky blízka hraničnej hodnote, od ktorej sa hrúbky stomov merali.



**Obrázok 2** **Príklad odhadu šiestich rôznych hrúbkových početností pre šesť druhov drevín vo vybraných porastoch ŠLP TU Zvolen (Merganič a Sterba 2006)**

Merganič – Sterba (2006) zistili, že Weibullová funkcia nie je vhodná na popis hrúbkového rozdelenia mladých lesných porastov. Podobné zistenie prezentoval Nanang (1998), ktorý analyzoval vhodnosť viacerých typov rozdelenia na popis hrúbkových rozdelení. Autor zistil, že pre popis hrúbkového rozdelenia v mladých porastoch je vhodnejšie log-normálne rozdelenie. Na základe našich výsledkov a výsledkov Nanang-a (1998) odporúčame použiť Weibullovú funkciu len vtedy, ak je stredná hrúbka väčšia ako 7 cm.

Získané informácie je možné veľmi výhodne využiť pri modelovaní porastovej štruktúry. Z existujúcich jednostromových rastových simulátorov používa na základe našich vedomostí metódu momentov len rastový simulátor TWIGS (Miner et al. 1988) vo svojom sub-modeli TREEGEN. Tento generátor však využíva troj-parametrovú Weibullovu funkciu a neumožňuje modelovanie hrúbkových rozdelení, ak sú hrúbky stromov merané od určitej preddefinovanej hraničnej hodnoty. Z nášho pohľadu je dvoj-parametrová Weibullova funkcia vhodnejšia ako troj-parametrová funkcia, pretože logicky by mala funkcia začínať od 0. Tento predpoklad by mal platiť aj vo veľmi starých porastoch. V takýchto prípadoch je napriek tomu, že funkcia začína od 0, veľmi nepravdepodobné, že pri simulácii hrúbok dostaneme hrúbky blízke 0.

Komplexnejšie hodnotenie štruktúry lesných porastov môžeme nájsť v prácach **[P05]Vorčák – Merganič – Saniga (2006)** a **[P06]Vorčák – Merganič – Merganičová (2006)**. V prácach sa rozoberá význam štrukturálnej diverzity, pretože diverzita je v silnom prepojení na procesy i funkčnosť ekosystému. Diverzita má blízky vzťah k naturálnej produkcii, ale v hlavnej miere sa spája so stabilitou ekosystému. V tejto súvislosti má dôležité postavenie oblasť zaoberajúca sa metódami kvantifikácie diverzity, pretože matematická kvantifikácia umožňuje nielen objektívnejšie ohodnotiť, ale v konečnom dôsledku aj lepšie pochopiť a popísať vzťahy, ktoré v ekosystéme fungujú. Medzi prvé práce zaoberajúce sa kvantifikáciou štrukturálnej diverzity horských lesov na Slovensku patrí práca „Monitoring diverzity horských lesov severnej Oravy". (Merganič et al. 2003), z ktorej vychádzajú aj uvedené príspevky.

V prácach sa analyzuje vplyv nadmorskej výšky a vývojového štádia na komplexný index štrukturálnej diverzity navrhnutý autormi Jaehne a Dohrenbusch (1997), na index diferenciácie podľa Füldnera (1995) a na agregačný index podľa Clarką a Evansa (1954). Skúmané horské lesy sa nachádzajú v supramontánnom stupni v drsných prírodných podmienkach, ktoré v rozhodujúcej miere formujú ich štruktúru. Toto konštatovanie sa v plnej miere potvrdilo, pretože z dosiahnutých výsledkov vyplýva, že nadmorská výška významnejšie ovplyvňuje štrukturálnu diverzitu a tým aj štruktúru vývojových štádií prírodného lesa. Súhrnne je možné konštatovať, že štrukturálna diverzita klesá so zvyšujúcou sa nadmorskou výškou až do výšky, kde dochádza k prirodzenému rozpojeniu porastov, odkiaľ opäť stúpa. V podmienkach Babej hory ide o výškovú hranicu okolo 1400 m n.m. Zároveň sa jednoznačne potvrdilo, že najvyššiu štrukturálnu diverzitu majú porasty v štádiu dorastania, nasleduje štádium rozpadu a optima.

Dlhodobo formovaná štruktúra porastov na Babej hore sa vyznačuje vlastnosťami, ktoré udržujú tieto ekosystémy v rovnováhe a stabilite. Štrukturálna diverzita lesných porastov je veľmi dôležitou charakteristikou stavu a vývoja porastov. Úzko súvisí so všetkými procesmi prebiehajúcimi v ekosystéme. Poznanie a rešpektovanie týchto zákonitostí môže významne napomôcť lesnému hospodárovi pri obhospodarovaní lesa v meniacich sa ekologických podmienkach a zabezpečiť jeho trvalosť. Vysoký stupeň stability porastov na Babej hore dokazuje aj skutočnosť, že ani vetrové smršte v novembri 2002 a 2004 výrazne nepoškodili tento ekosystém, hoci v ďalších horských oblastiach na Slovensku vznikli katastrofálne škody. Vysoký stupeň stability porastov na Babej hore je rozoberaný aj v ďalšej práci **[P07]Vorčák – Merganič – Merganičová (2007)**, v ktorej sa konštatuje, že horské lesy predstavujú jedinečný a polyfunkčný ekosystém. Sú mimoriadne dôležitým stabilizačným prvkom v krajine, ale zároveň aj rovnako dôležitým regulačným faktorom hydrických, eróznych, protilavínových a ďalších procesov. Stabilita tohto ekosystému má preto mnohonásobný význam. V práci sa analyzuje statická stabilita lesných porastov v NPR Babia hora v Oravských Beskydách na podklade piatich ukazovateľov: depigmentácie, defoliácie, korunovosti, štíhlostného koeficienta a poškodenia stromov. Z analýzy vyplýva, že s nárastom nadmorskej výšky rastie aj percento depigmentácie, čo je podľa nášho predpokladu spôsobené vplyvom ozónu ($O_3$) ako fotooxidanta. V priemere je však hodnota depigmentácie veľmi malá a porasty v NPR Babia hora je možné zaradiť do stupňa 0 - bez zmeny zafarbenia. Z hodnotenia defoliácie vyplýva, že defoliácia s nadmorskou výškou naopak klesá. Vzhľadom na defoliáciu je možné zaradiť porasty v NPR Babia hora do 1. stupňa. Depigmentácia a defoliácia sa výrazne neprejavila ani v minulosti v období 80. rokov, kedy boli nižšie položené smrekové porasty výrazne poškodené vplyvom vysokého imisného tlaku z miestnych i diaľkových zdrojov, pretože daný masív je vystavený prevládajúcemu prúdeniu vzduchu, ktorý prinášal množstvo polutantov.

Veľkosť koruny - korunovosť plní dôležitú úlohu vo fyziológii drevín a odolnosti voči komplexu škodlivých činiteľov. Typickým javom horských lesov v masíve Babej hory je, že jednotlivé smreky majú hlboké a úzke koruny formované okrem genetických vlastností aj ich vzájomným priestorovým usporiadaním. Hodnota korunovosti hornej porastovej vrstvy signifikantne rastie s nadmorskou výškou. Najvyššie hodnoty dosahuje korunovosť v štádiu dorastania. Na základe korunovosti hodnotíme porasty v NPR Babia hora ako veľmi stabilné porasty.

Hodnoty štíhlostného koeficienta ako hlavného ukazovateľa statickej stability lesných porastov potvrdili vysokú rezistenciu horských lesov v masíve Babej hory. Tento ukazovateľ signifikantne klesá s rastúcou nadmorskou výškou a vo výškovej kategórii nad 1460 m n. m. dosahuje veľmi nízke hodnoty (0.35). Ide o vysoko stabilné porasty, kde je poškodenie porastov veternou smršťou takmer nemožné, pokiaľ koruny nie sú zaťažené námrazou. Z hľadiska poškodenia stromu je hlavný faktor sneh a námraza, ktoré sa prejavili najvýraznejšie vo výškovej kategórii nad 1460 m n. m.

Na základe hodnotených ukazovateľov ekologickej stability sa v práci konštatuje, že horské lesy v masíve Babej hory predstavujú vysoko stabilný ekosystém dlhodobo formovaný v danej klimaticko – geografickej oblasti do stavu vnútornej homeostázy. Táto zabezpečuje trvalú stabilitu v rôznych extrémnych situáciách. Významnú úlohu pre tento smrekový ekosystém hrá aj surová klíma, hlavne teplota a zrážky. Tieto faktory priaznivo pôsobia proti šíreniu biotických škodlivých činiteľov (hmyzu) a výrazne vplývajú na štruktúru a rastové pomery porastov.

## 3.3. Vplyv vybraných faktorov na diverzitu lesných ekosystémov

Diverzita v určitom priestore a čase je určená kombináciou abiotických obmedzujúcich podmienok, biotických interakcií a rušivých vplyvov/kalamít (Frelich et al. 1998, Nagaraja et al. 2005, Misir et al. 2007, Ucler et al. 2007, Spies a Turnier 1999). Abiotické faktory, ako nadmorská výška, sklon, expozícia, textúra pôdy, klíma, atď. určujú podmienky fyzického prostredia a teda aj primárne rozšírenie druhov. Tieto vzťahy boli sledované a študované už v 19. storočí (Hansen a Rotella 1999). Parametre ovplyvňujúce rast rastliny a dostupnosť zdrojov, napr. klíma, sú považované za primárne faktory (Terradas et al. 2004), kým terénne charakteristiky, napr. nadmorská výška, sa považujú za nepriame faktory pretože nemajú priamy vplyv na rast rastliny, ale sú korelované k primárnym faktorom (Pausas et al. 2003; Bhattarai et al. 2004). Nepriame faktory sa často používajú pri analýzach, keď informácie o primárnych faktoroch nie sú dostupné (Pausas a Saez 2000). Najčastešie sa skúma vzťah diverzity k nadmorskej výške (Grytnes a Vetaas 2002, Bhattarai a Vetaas 2003, Bachman et al., 2004), kým vplyvy ostatných topografických charakteristík sa preverujú len zriedka (Johnson 1986, Palmer et al. 2000). Hoci sa moderní ekológovia zameriavajú predovšetkým na iné faktory, napr. prírodné rušivé vplyvy – kalamity, v poslednom čase začal vplyv abiotických faktorov na druhovú diverzitu v vedcov opätovne zaujímať (Burns 1995, Rosenzweig 1995, Austin et al. 1996, Ohmann a Spies 1998, Hansen a Rotella 1999). Väčšina prác však skúma len vzťah faktora prostredia k počtu druhov drevín, čo reprezentuje iba jednu zložku diverzity. V práci **[P08]Merganič – Quednau – Šmelko (2004)** sa analyzuje vplyv vybraných vlastností geomorfológie (expozícia, sklon, nadmorská výška, typ terénu) na diverzitu stromovej vrstvy, ktorá je kvantifikovaná 9 indexami druhovej diverzity (*N0, R1, R2, H', N1, N2, E1, E3, E5*). Výsledky práce ukázali, že iba nadmorská výška má signifikantný vzťah k druhovej diverzite. Hoci sú vzťahy veľmi slabé, pretože korelačné koeficienty nepresahujú hodnoty 0.3–0.4, závislosť druhovej diverzity na nadmorskej výške sa zistila aj v iných prácach a tento vzťah sa vo všeobecnosti v ekologickej literatúre akceptuje. Priestorová interpolácia stupňov druhovej diverzity, ktoré boli určené modelom

BIODIVERSS, pomocou geoštatistických metód nám dáva cenné informácie na regionálnej úrovni. Ako pri všetkých štatistických metódach, výstup z modelu je odhadom skutočného stavu a nemusí sa presne zhodovať s realitou. Napriek tomu nám umožňuje vykonať rýchle analýzy, ktoré v prepojení s GIS metódami umožňujú užívateľovi testovať vplyv rôznych faktorov na napr. druhovú diverzitu. Použitím uvedených nástrojov sme mohli zhodnotiť vzťah druhovej diverzity k typu hospodárenia a k fragmentácii porastu. Prezentovaná práca využíva údaje z regionálnej inventarizácie lesa. Takúto analýzu by bolo možné vykonať aj na údajoch z lesných hospodárskych plánov. Ich využitie by však mohlo pri kvantifikácii druhovej diverzity na regionálnej úrovni spôsobiť viacero problémov, pretože: a) v rámci taxácie lesov sa zaznamenávajú len dominantné druhy, t.j. informácie o vzácnych druhoch chýbajú, b) indexy druhovej diverzity sú silne korelované s veľkosťou (plochou) hodnoteného porastu, c) klasické indexy druhovej diverzity sú pozične nezávislé, t.j. neberú do úvahy priestorové zmiešanie druhov. Preto je v takých prípadoch výhodné používať údaje z inventarizácie lesa, čo umožní štandardizáciu hodnotiacich metód. Kvantifikácia stupňa druhovej diverzity modelom BIODIVERSS rieši problém rozličnej veľkosti hodnoteného objektu a berie do úvahy priestorové zmiešanie druhov. Mapa znázorňujúca priestorovú interpoláciu stupňa druhovej diverzity sa môže považovať za jednoduchý nástroj pre lesného hospodára, ktorý je užitočný pre hodnotenie napr. štrukturálnej diverzity, stability lesného ekosystému, pri práci s verejnosťou ako aj z hľadiska implementácie Dohovoru o biologickej diverzite. Regionalizácia sa vykonala geoštatistickým prístupom na báze „indicator“ krigingu. Aplikácia tohto druhu krigingu v našej práci s lesníckou tématikou bola jedna z prvých publikácií. Finálna krigingová mapa sa ukázala ako vhodný nástroj pri interepretovaní druhovej diverzity na záujmovom území.

V práci **$^{P09}$Ozcelik – Gul – Merganič – Merganičová (2008)** sme skúmali vplyv porastových charakteristík (porastový zápoj, kruhová základňa, zásoba, vek, stredná hrúbka, počet stromov, index heterogenity) a geomorfologických znakov (nadmorská výška, expozícia, sklon) na druhovú diverzitu na príklade neobhospodarovaných zmiešaných lesných porastov vo východnej časti oblasti Čierneho mora v Turecku. Drevinová diverzita a heterogenita kruhovej základne skúmaných lesných ekosystémov sa kvantifikovali Shannon-Weaver-ovým a Simpson-ovým indexom. Vzťahy medzi drevinovou diverzitou, heterogenitou kruhovej základne, porastovými charakteristikami a geomorfologickými znakmi sa skúmali pomocou regresnej analýzy. V práci sme zistili, že vzťahy medzi drevinovou diverzitou a porastovými charakteristikami sú rôzne tesné s korelačným koeficientom od 0.02 do 0.70. Korelácia heterogenity kruhovej základne s porastovými charakteristikami sa pohybovala od 0.004 do 0.77 ($R^2$). Na základe našich výsledkov sa porasty s vyššou drevinovou diverzitou vyznačujú väčšou strednou hrúbkou, vyšším počtom hrúbkových tried, vyššou kruhovou základňou a nižšou hodnotou indexu homogenity. Pri skúmaní vplyvu geomorfologických znakov na heterogenitu drevín a kruhovej základne sme zistili, že všetky analyzované vzťahy majú koeficient determinácie $R^2 < 0.24$. Štatisticky signifikantná korelácia sa zistila len medzi drevinovou diverzitou a expozíciou.

V roku 1993, t.j. rok po schválení Dohovoru o biologickej diverzite, bola na konferencii o ochrane lesov Európy v Helsinkách prijatá rezolúcia „Stratégia dlhodobej adaptácie lesov Európy na klimatickú zmenu“, ktorú podpísali takmer všetky európske krajiny. Významné aktivity vo vzťahu k efektom globálnej zmeny na lesy a lesné hospodárstvo sa rozvíjajú aj v rámci Medzivládneho panelu pre klimatické zmeny (IP CC), IGBP a IUFRO - programu „Global Change and Terrestrial Ecosystems“, z ktorého je pre lesníctvo najvýznamnejšia aktivita „Efekty globálnej zmeny na obhospodarované lesy“ (Minďáš a Škvarenina 2000).

Štúdium vplyvu klimatických zmien na druhovú diverzitu fytocenóz naberá v súčasnej dobe pomerne veľký význam, s čím súvisí aj množstvo kvantifikačných matematicko-štatistických metód. Medzi veľmi často používané metódy pri výskume dopadu klimaticko - antropogénnych vplyvov na lesné ekosystémy patria indikačné metódy založené na ekologických nárokoch jednotlivých rastlinných druhov. Takúto metodiku navrhol aj Ellenberg (1979, 1992), ktorý pre väčšinu rastlinných druhov definoval ich ekologické nároky vo vzťahu k šiestim najdôležitejším ekologickým faktorom: svetlo, teplo, kontinentalita, vlhkosť, reakcia na pH a dusík. Analýza spočíva vo fytocenologickej analýze spoločenstiev v určitej oblasti (fytocenologickom zápise) a následnom priradení Ellenbergových ekohodnôt každému rastlinnému druhu. Po spracovaní takéhoto fytozáznamu je možné indikovať vplyv prevládajúceho faktora v hodnotenej oblasti. V prípade, že sú k dispozícii opakované merania, teda časové rady, môže byť analýza rozšírená na kvantifikáciu predpokladaných klimaticko – antropogénnych vplyvov. Z výsledkov takýchto štúdií vyplýva, že naozaj dochádza k zmene druhového zloženia, resp. pokryvnosti (kvantitatívneho podielu) rastlinných druhov, čo indikuje zmenu ekologických podmienok (Markert et al. 2003). Prevažná väčšina autorov konštatuje tieto zmeny v súvislosti s depozíciou dusíka a zakysľovaním (Pitcairn et al. 2003, Ling 2003, Bohling 2003, Abolina et al. 2001, Sorensen a Tybirk 2001, Brunet et al. 2000, Oredsson 1999, Diekmann et al. 1999, Tyler a Olsson 1997, Diekmann et al. 1998, Roder et al. 1996). Problematike zmien bylinnej zložky v lesných ekosystémoch sa venovali aj Fallkengren a Grerup (1986, 1987, 1989), Tyler (1987), Thimonier et al. (1992). Bylinnej synúzii ako indikátoru významných a dlhodobejších zmien v lesnom ekosystéme sa v našich podmienkach venovali Ambros a Míchal (1992), Ambros et al. (1995), Križová (1994, 1996), Nič (1995, 1999), Vološčuk (2001) a Pavlenda a Ištoňa (2000).

V prácach **[P10]Merganič – Ištoňa (2004)** a **[P11]Merganič – Ištoňa – Merganičová (2005)** je analyzovaný vzťah medzi zmenou druhovej bohatosti lesných fytocenóz 6. lesného vegetačného stupňa a zmenou ekologických podmienok. Druhová bohatosť je kvantifikovaná troma indexami *N0* (Hill 1973), *R1* (Margalef 1958) a *R2* (Menhinick 1964). Zmena ekologických podmienok je stanovená fytoindikáciou druhov voči šiestim ekologickým faktorom (vlhkosť, teplota, pH reakcia, kontinentalita, svetlo a dusík) definovaných podľa Ellenberga et al. (1992) za 29-ročnú periódu. Empirický materiál predstavuje 14 výskumných plôch z oblasti Stredných Beskýd založených v roku 1972 za účelom celonárodného typologického prieskumu a opätovne meraných v roku 2001. Plochy reprezentujú tri skupiny lesných typov a to *Fagetum-abietino-piceosum, Abieto-Fagetum* a *Fageto-Abietum.*

Z dosiahnutých výsledkov vyplýva, že v uvedených skupinách lesných typov došlo za uvedenú periódu k poklesu druhovej bohatosti. Za najvážnejší faktor môžeme považovať zakysľovanie prostredia pravdepodobne spôsobené znečistením ovzdušia. Ďalšími významnými faktormi, ktoré ovplyvňujú druhovú bohatosť hodnotených lokalít, sú teplota, vlhkosť a kontinentalita. Uvedené ekofaktory štatisticky preukázateľne spôsobujú nárast v pokryvnosti druhov indiferentných k daným ekofaktorom. Rastúci trend v pokryvnosti indiferentných druhov môže indikovať to, že dochádza k pomerne častým výkyvom podmienok prostredia, čo úzko súvisí v súčasnosti s veľmi frekventovaným javom globálnych klimatických zmien. V takýchto podmienkach ubúdajú druhy špecificky viazané na konkrétne podmienky, resp. klesá ich pokryvnosť, čím sa zákonite uvoľňuje priestor druhom so širokou ekologickou valenciou. Výskyt a rozširovanie sa takýchto druhov môže v budúcnosti spôsobiť problémy v typológii pri klasifikácii podmienok prostredia.

V ďaľšej práci **[P12]Merganič – Merganičová – Vorčák – Ištoňa (2008)** analyzujeme vplyv vývojového štádia prírodného lesa na fytocenózu. Analýza bola zameraná na preukázanie existencie diferenciačných druhov viažucich sa na vývojové štádiá ako aj skupiny druhov s rovnakou indikačnou hodnotou podmienok prostredia v zmysle Ellenberga et al. (1992). Z výsledkov analýzy vyplýva, že v danej oblasti sa vo fytocenóze nenachádza druh, ktorého zvýšená alebo znížená pokryvnosť by poukazovala na príslušnosť k vývojovému štádiu prírodného smrekového lesa. Najsilnejší a signifikantný vzťah sme zistili medzi pokryvnosťou *Polytrichum formosum* Hedw. a štádiom optima a medzi *Dicranum scoparium* Hedw. a štádiom dorastania. Zmeny v pokryvnosti týchto druhov môžeme charakterizovať tzv. „veľkým cyklom“. Ich pokryvnosť je vysoká len v určitom vývojovom štádiu, kým v ďalších dvoch štádiách je výrazne nižšia, avšak v oboch štádiách zhruba na rovnakej úrovni. Určitý, aj keď štatisticky nesignifikantný náznak cyklických zmien („malý cyklus“) sme zistili aj v pokryvnosti taxónov *Avenella flexuosa* (L.) Parl., *Dryopteris dilatata* (Hoffm.) A. Gray, *Gentiana asclepiadea* L., *Hypnum cupressiforme* Hedw., *Oxalis acetosella* L., *Rubus idaeus* L., *Senecio germanicus* Wallr. a *Vaccinium myrtillus* L., pri ktorých sa pokryvnosť kontinuálne mení.

Z analýzy vplyvu vývojového štádia prírodného lesa na podiel druhov fytocenózy s rovnakou indikačnou hodnotou podmienok prostredia podľa Ellenberga et al. (1992) vyplýva, že v skúmanej oblasti sa ekologické podmienky indikované fytocenózou v závislosti od vývojového štádia prírodného lesa štatisticky signifikantne nemenia. Stav porastov v danej oblasti (trvalo rozpojený zápoj) a špecifické klimatické podmienky vytvárajú predpoklady priestorovo homogénnej fytocenózy, teda stavu, ktorý je podobný klimaxu.

Vplyv zveri na lesné ekosystémy je už dlhodobo predmetom skúmania vedcov (Putman 1986, Gill 1992a, b, Gill a Beardall 2001, Rooney 2001, Côté et al. 2004). Zver ovplyvňuje nielen vegetáciu, ale aj iné živočíchy, bezstavovce, pôdu, cyklus živín, atď., a to priamo i nepriamo (Putman 1986, Rooney 2001, Rooney a Waller 2003), pozitívne ako aj negatívne (Putman 1986, Reimoser et al. 1999, Gill a Beardall 2001, Crête et al. 2001, White et al. 2004). To, či a ako ovplyvňuje prítomnosť zveri určitú zložku biodiverzity, závisí od hustoty zveri, druhovo a stanovištne špecifických faktorov, a od vzťahu medzi zverou a skúmanou zložkou (e.g. konkurencia, predácia) (Putman 1986, Stewart 2001, White et al. 2004). Vo všeobecnosti však platí, že nadmerná hustota zveri má celkovo nepriaznivý vplyv na biodiverzitu (Putman 1986, Rooney 2001, Côté et al. 2004, White et al. 2004, Carson et al. 2005), hoci niektoré druhy rastlín, bezstavovcov a živočíchov môžu z takéhoto stavu profitovať (White et al. 2004).

Štruktúru lesa ovplyvňuje aj zver (Putman 1986, Gill a Beardall 2001, Rooney 2001, Rooney a Waller 2003, Côté et al. 2004). Dostatočne vysoká hustota zveri môže znížiť vertikálnu štruktúru lesa (Rooney 2001). Na druhej strane, pod určitou hranicou hustoty zveri nedochádza k žiadnemu poškodeniu a je badateľný len veľmi malý vplyv na drevinové zloženie alebo na štruktúru lesa (Gill 1992a).

Zmladenie, ktoré predstavuje budúcnosť lesného porastu, je zo stromovej vrstvy ekosystému na poškodenie zverou najcitlivejšie (Potvin et al. 2003). Zvyčajne sa rozlišujú tri hlavné typy priameho poškodenia: odhryz, obhryz a poškodzovanie stromov vytĺkaním parožia (Gill 1992a, b, Motta 1996). Hoci všetky tieto vplyvy spôsobujú poškodenie jednotlivého stromu, na úrovni porastu nemusia mať nevyhnutne negatívny vplyv na drevinovú diverzitu. Celkový vplyv závisí od mnohých faktorov, napr. od načasovania a intenzity poškodenia, drevinového zloženia podrastu, od toho, akú drevinu zver ovplyvňuje, od citlivosti dreviny na poškodenie a pod. (Putman 1986, Gill 1992a, Reimoser et al. 1999, Côté et al. 2004). Ak napr. zver zníži podiel najzastúpenejšej dreviny v podraste resp. zmladení, drevinová diverzita sa môže zvýšiť (Helle a Aspi 1983, Gill 1992b), kým poškodenie vzácnych a citlivých drevín môže spôsobiť stratu druhov z ekosystému (Martin a Daufresne 1999).

Rooney (2001) uvádza, že pred osídlením severnej Ameriky bola hustota zveri na danom území nízka, pričom bola regulovaná počasím, predátormi, štruktúrou lesa a druhovým zložením. Zmeny v prírodných lesoch spôsobené človekom vniesli do týchto vzťahov nerovnováhu. Preto je v súčasnosti ťažké stanoviť hustotu zveri, ktorú možno v určitých podmienkach očakávať v rovnovážnom lesnom ekosystéme. Z tohto dôvodu sa zakladali dlhodobé experimenty s oplôtkami, v ktorých je možné skúmať reakciu lesných ekosystémov na vylúčenie zveri z lesa (e.g. Eiberle 1967, Leibundgut 1974, Ertl 1989, König a Baumann 1990, Pollanschütz 1992, König 1997, Nomiya et al. 2003, Von Oheimb et al. 2003, Stone et al. 2004, etc.). Hoci nám takéto štúdie neposkytnú informácie o pôvodnom stave lesa, pretože zver sa v lese vyskytuje prirodzene, môžu výrazne napomôcť objektívnemu hodnoteniu vplyvu zveri na lesný ekosystém (Reimoser et al. 1999).

V práci **[P13]Merganič – Russ – Beranová – Merganičová (2009)** analyzujeme vplyv zveri na druhovú a výškovú diverzitu mladých rastových stupňov lesa v troch lokalitách Českej republiky. Skúmané objekty sú tvorené jednou oplotenou časťou, v ktorej bola prítomnosť zveri vylúčená, a jednou neoplotenou časťou s voľným prístupom zveri. Empirický materiál pre matematicko-štatistické analýzy sa získal výberovým zisťovaním (inventarizácia lesa) so systematickým rozmiestnením skusných plôch. Druhová a výšková diverzita sa kvantifikovala 10 indexami diverzity.

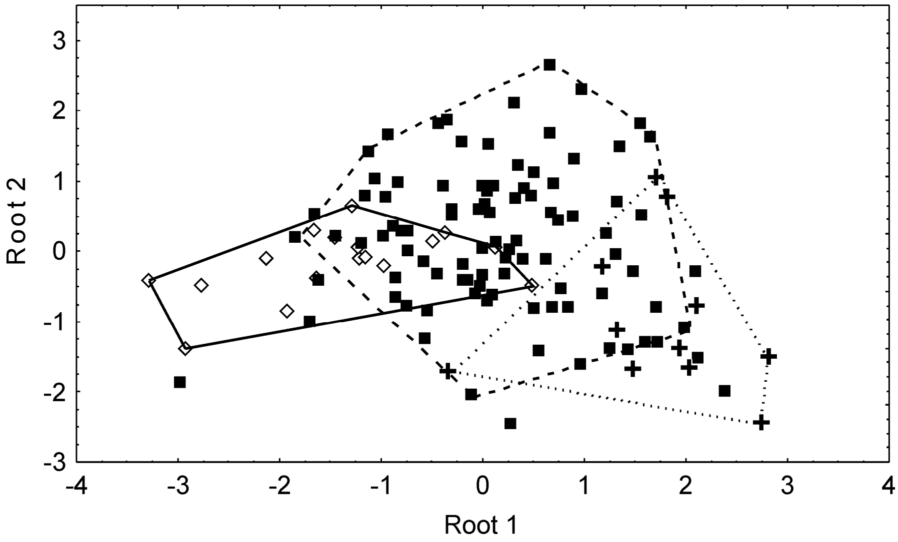
Analýzou sa zistilo, že nadmerná hustota zveri významným spôsobom ovplyvňuje druhovú a výškovú diverzitu mladých rastových stupňov lesa. Vyšší negatívny vplyv zveri na druhovú diverzitu je možné očakávať na kyslých stanovištiach, kým na živných stanovištiach je na vplyv zveri citlivejšia výšková diverzita. Ak je však hustota zveri dlhodobo pod normovaným stavom, výskyt zveri neovplyvňuje druhovú a výškovú diverzitu mladých rastových stupňov lesa.

# 4. Zložky štruktúry lesa – indikátor prirodzenosti lesných ekosystémov

Pojmy ako biodiverzita a prirodzenosť sa v ochrane prírody často používajú (Schnitzler et al. 2008), pričom prirodzenosť sa považuje za jedno z najdôležitejších kritérií hodnotenia stavu lesných ekosystémov (Bartha et al. 2006). Veľmi často predstavuje hlavný nástroj podporujúci plánovanie ochrany (Hoerr 1993, Schmidt 1997 in Bartha et al. 2006). Význam prirodzenosti potvrdzujú aj mnohé medzinárodné dohovory a schémy, v ktorých je zaradená medzi hlavné kritériá, napr. prirodzenosť je jedným z celoeurópskych indikátorov trvaloudržateľného hospodárenia (MCPFE 2002).

Hoci vo vedeckej literatúre sa vo veľkom diskutuje o definícii prirodzenosti (viď. napr. Götmark 1992, Franklin 1998, Williams 2002, Cole et al. 2008, etc.), zvyčajne sa prirodzenosť chápe ako stav, ktorý môže pretrvať bez zásahu človeka (Cole et al. 2008). Stupeň prirodzenosti ako kvantitatívne vyjadrenie teda indikuje intenzitu ľudských zásahov. V rámci Ministerských konferencií o ochrane lesov v Európe (MCPFE) sa rozlíšujú tri stupne prirodzenosti lesa: lesy bez vplyvu človeka, prirodzené lesy a umelo založené lesy. Lesy bez zásahu človeka sú lesy, v ktorých sú pôvodné prírodné procesy a drevinové zloženie do určitého stupňa zachované alebo boli obnovené. Prirodzené lesy majú určité charakteristiky prírodného lesa, ktoré zaručujú, že dynamika a biodiverzita sa približujú pôvodnému ekosystému. Umelo založené lesy predstavujú človekom vytvorené lesné spoločenstvá, ktoré sa úplne odlišujú od pôvodného ekosystému.

Stupeň prirodzenosti lesa sa hodnotí na základe vybraných znakov (Bartha et al. 2006), najmä: pôvodnosť druhov a genotypov, diferenciácia porastovej štruktúry (napr. rozdelenie hrúbkových početností, vertikálna a veková štruktúra, výskyt odumretého dreva a prirodzeného zmladenia lesa), ako aj výskyt a rozsah antropogénneho vplyvu v daných lesných ekosystémoch (napr. výskyt a metódy ťažby a obnovy lesa, príprava pôdy, výskyt lesných ciest, rekreačné aktivity, pastva, poškodenie lesa) (Moravčík et al. 2010). Celková prirodzenosť je výsledkom kombinácie čiastkových indikátorov prirodzenosti. Počet parametrov vybratých na hodnotenie prirodzenosti sa od publikácie k publikácii rôzni. Niektorí autori používajú len jeden indikátor, napr. Pasierbek et al. (2007) založili svoj index prirodzenosti len na hrúbkovom rozdelení. Podobne Glončák (2007) analyzoval prirodzenosť len na základe drevinového zloženia, kým Šmídt (2002) bral do úvahy drevinové zloženie a štruktúru lesa. Hepcan a Coskun (2004) navrhli pre výpočet celkovej prirodzenosti aditívny model, do ktorého vstupujú štyri indikátory: (1) vzdialenosť od ciest, (2) vzdialenosť od osád, (3) biofyzická prirodzenosť (do akej miery je prírodné prostredie bez biofyzikálneho narušenia spôsobeného ľudskou činnosťou), a (4) jedinečnosť (výskyt vzácnej prírodnej vegetácie). Bartha et al. (2006) navrhli komplexné hodnotenie prirodzenosti lesa na základe zhodnotenia prirodzenosti zloženia a štruktúry porastovej vrstvy, krovinovej vrstvy, zmladenia, a pôdnej vegetácie. [P14]**Moravčík – Sarvašová – Merganič – Schwarz (2010)** prezentovali klasifikačný model stupňa prirodzenosti lesa (pralesy, prirodzené a umelo vytvorené lesy) vytvorený pre smrekové lesy Slovenska. Model bol vytvorený na základe štatistických princípov za použitia rozsiahlej databázy skusných plôch zo smrekových lesov 7. vegetačného stupňa (na základe klasifikácie Zlatníka 1976). Článok prezentuje dva varianty tohto modelu, jeden založený na diskriminačnej analýze, kým druhý využíva aditívny prístup. Analýza údajov z horských smrekových lesov odhalila štatisticky signifikantné indikátory stupňa prirodzenosti lesa: aritmetický priemer korunovosti stromu (podiel dĺžky koruny k výške stromu), objem odumretého dreva, pokryvnosť tráv, pokryvnosť machov a lišajníkov, a agregačný index podľa Clark a Evansa (1954). Okrem toho sa do konečného modelu dodal variačný koeficient hrúbok, pretože jeho prítomnosť v modeli pozitívne ovplyvnila správnosť klasifikácie stupňa prirodzenosti lesa. Správnosť klasifikácie bola pomocou navrhnutého diskriminačného modelu 74.5%. Pri aditívnom modeli sa definovali intervaly hodnôt integrovaného indikátora pre každý stupeň prirodzenosti lesa s ohľadom na interval chyby aritmetického priemeru a percentily hodnôt v jednotlivých stupňoch prirodzenosti lesa. Celková správnosť klasifikácie pomocou aditívneho modelu bola 63.4%. V druhej časti sa navrhla schéma využitia klasifikačného modelu stupňa prirodzenosti lesa v rámci rozhodovacieho procesu vyhlasovania lesných chránených území. V rámci tejto schémy sa stupeň prirodzenosti lesa považuje za základné kritérium pri určovaní hodnoty ochrany lesných ekosystémov. Medzi ďalšie kritériá na podporu rozhodovania sme zaradili možnosti zlepšenia prirodzenosti menej prirodzených lesných ekosystémov, ktoré sú vyhlásené za chránené, výskyt ohrozených druhov a výskyt iných prírodných hodnôt.



**Obrázok 3** **Grafická interpretácia klasifikácie stupňa prirodzenosti lesa pomocou kanonickej analýzy (Moravčík et al. 2010)**
**Legenda: kosoštvorec – pralesy, štvorec – prírodné lesy, krížik – umelé lesy**

Ako vyplýva z predošlej práce, odumreté drevo je dôležitým štrukturálnym prvkom pre hodnotenie prirodzenosti lesného ekosystému. Pri konštrukcii vyššie uvedeného klasifikačného modelu stupňa prirodzenosti lesa bola zásoba odumretého dreva druhým najsignifikantnejším indikátorom prirodzenosti lesa (Merganič 2008). Podobne aj štúdia vzťahu komplexného ukazovateľa biodiverzity a ekonomickej hodnoty lesného porastu na podklade celoslovenských údajov z NIML SR preukázala v tomto vzťahu dôležitú pozíciu zásoby odumretého dreva (Merganič a Merganičová 2008). Uvedomenie si dôležitosti ekologickej funkcie odumretého dreva viedlo k jeho zaradeniu medzi indikátory biodiverzity a trvalej udržateľnosti na Európskej úrovni. Ministerská konferencia o ochrane lesov v Európe (MCPFE) zvolila odumreté drevo ako jeden z 9 Pan-európskych indikátorov trvalej udržateľnosti lesných ekosystémov. Európska environmentálna agentúra (EEA) zaradila odumreté drevo medzi 15 hlavných indikátorov biodiverzity (Humphrey et al. 2004). V mnohých schémach indikátorov sa odumreté drevo zaraďuje medzi štrukturálne indikátory. V USA sa v rámci FIA (Forest Inventory and Analysis program of the USDA Forest Service) berie odumreté drevo ako indikátor štrukturálnej diverzity lesa, uhlíkových zásob a zásob palivového dreva (Woodall a Williams 2005).

Pohľad lesníkov na odumreté drevo sa v súčasnosti mení. V minulosti sa chápal každý odumretý strom ako potencionálny zdroj nákazy pre les, napr. z dôvodu premnoženia podkôrneho hmyzu a iných „škodcov“ lesa. Prevádzkové opatrenia boli zamerané na spracovanie a speňaženie každého kusa dreva, ktorý sa v lese vyskytoval (Mössmer 1999). Dnes pri presadzovaní prírode blízkeho hospodárenia v lese význam odumretého dreva v lesnom hospodárstve narastá, keďže najnovšie vedecké poznatky dokumentujú jeho dôležitosť pre biodiverzitu lesných ekosystémov (Müller a Schnell 2003), cyklus živín (Lexer *et al.* 2000) ako aj prirodzenú obnovu lesa najmä v extrémnych horských a severských podmienkach (Mai 1999, Hofgaard 1993).

Odumreté drevo plní niekoľko významných funkcií v lesných ekosystémoch. Vplýva najmä na

- produktivitu lesných porastov
- biodiverzitu lesných porastov
- vývoj lesných porastov
- dlhodobý kolobeh živín v lesných ekosystémoch
- geomorfológiu lesných pôd a vodných tokov.

Existencia odumretého dreva v lesných porastoch zlepšuje ich produktivitu (Marra a Edmonds 1994, Mcminn a Crossley 1993). Tento pozitívny vplyv vyplýva z vlastností odumretého dreva:

- odumreté drevo je významnou zásobárňou a potencionálnym zdrojom živín
- priaznivo ovplyvňuje množstvo organickej hmoty v pôde
- vzťahom k organizmom rozkladajúcim odumretú hmotu ovplyvňuje kvalitu rozkladu organickej hmoty
- priaznivo ovplyvňuje pôdnu vlhkosť
- vyvára podmienky pre život symbiotických a dusík viažúcich baktérií
- priaznivo ovplyvňuje pôdne procesy (zakysľovanie a podzolizácia pôd)
- je dôležitým substrátom pre obnovu rastlinných a živočíšnych druhov.

V súčasnosti je pravdepodobne najznámejším a najviac skloňovaným význam odumretého dreva pre biodiverzitu lesných porastov (Svoboda 2007). Odumreté drevo predstavuje priestor pre život mnohých živočíchov, rastlín a húb (Montes et al. 2004), najmä pre malých cicavcov, vtákov hniezdiacich v dutinách stromov, machy, saproxylické huby a bezstavovce (Humphrey et al. 2004). V škandinávskych krajinách je na odumretom dreve závislých cca 6000—7000 druhov, čo predstavuje 25% všetkých lesných druhov v tomto regióne (Stokland et al. 2003). Podobne je to aj v iných častiach sveta. Preto sa odumreté drevo považuje za hlavný faktor biodiverzity najmä vzhľadom k druhovej bohatosti ako jednej zložky biodiverzity (Schuck et al. 2004, Ferris a Humphrey 1999).

Okrem druhovej diverzity zvyšuje odumreté drevo aj štrukturálnu diverzitu, keďže jeho rozmiestnenie po poraste silne varíruje. Táto priestorová variabilita je spôsobená vlastnosťami prostredia ako je topografia alebo pôdne podmienky (Stokland et al. 2003), ale závisí aj od vývoja porastu a faktorov spôsobujúcich mortalitu (sucho, oheň, vietor, hmyz, atď.) (Humphrey et al. 2004). Výsledkom hlúčkovitej akumulácie odumretej drevnej hmoty v lese je vytváranie mozaiky najrôznejších typov biotopov, čo zvyšuje nielen druhovú diverzitu ale aj diverzitu biotopov (Svoboda 2007).

Viaceré štúdie z ihličnatých lesov poukázali na to, že hrubé ležiace odumreté drevo je vhodným substrátom pre klíčenie semien a odrastanie semenáčikov (Harmon et al. 1986). Tento fenomén bol pozorovaný v lesoch mierneho pásma ako aj v boreálnych lesoch (Harmon a Franklin 1989; Szewczyk a Szwagrzyk 1991, Vorčák et al. 2005), čo indikuje, že hrubé odumreté ležiace drevo môže hrať v dynamike týchto lesov dôležitú úlohu (Svoboda 2007). Vhodnosť odumretého ležiaceho dreva pre uchytenie a rast mladých jedincov však závisí od jeho kvalitatívnych parametrov, najmä od stupňa rozkladu, s ktorým súvisia ďalšie charakteristiky (množstvo živín, vlhkosť). Napr. Merganič et al. (2003) zistili, že z celkovej

8-stupňovej škály navrhnutej Holeksom (2001) bol najvhodnejším podkladom pre vznik a prežívanie prirodzenej obnovy 5. a vyšší stupeň rozkladu, na ktorých zaznamenali takmer 99% inventarizovanej obnovy. V ekosystémoch s bohatou synúziou podrastu je zároveň obnova, ktorá vznikla na odumretom dreve, chránená pred konkurenciou s nadzemnou vegetáciou.

Mechanické a fyzikálne vlastnosti odumretého dreva veľkých dimenzií významne ovplyvňujú geomorfológiu lesných pôd a malých vodných tokov v lesných ekosystémoch (Stevens 1997). Na povrchu lesných pôd prispieva odumretá drevná hmota k:

- zvýšeniu stability svahov a stability pôdneho povrchu, čím zabraňuje pôdnej erózii a prispieva ku kontrole povrchového odtoku, ktorý unáša pôdne častice a organickú hmotu z pôdneho povrchu
- ovplyvneniu charakteru malých vodných tokov v lesných porastoch

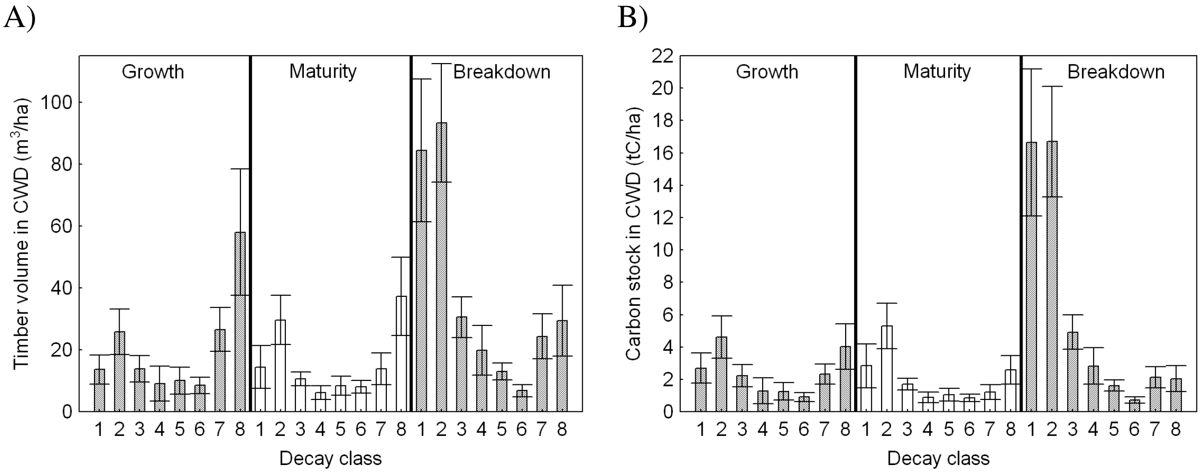
Veľký význam pri stabilizácii pôdy a kontrole povrchového odtoku má odumreté drevo najmä na prudkých svahoch, predovšetkým pri umiestnení kolmo na svah.

Odumreté drevo hrá centrálnu úlohu v materiálnom a energetickom toku ako aj v kolobehu živín. Hrubé odumreté drevo predstavuje zásobáreň uhlíka, dusíka, fosforu a ďalších živín, pričom niektorí autori (Turner et al. 1995, Janisch a Harmon 2001, Holub et al. 2001, Mackensen a Bauhus 2003) ho považujú za hnojivo s postupným uvoľňovaním živín. Stevens (1997) uvádza, že odumreté drevo hrubých dimenzií zásobí ekosystém živinami viac ako 100 rokov. Z tohto dôvodu predstavuje odumreté drevo významný článok ochrany ekosystému pred stratami živín, ku ktorým môže dôjsť pri kalamitách, a teda pozitívne vplýva na stabilitu a rovnováhu ekosystému (Zimmerman et al. 1995, Wei et al. 1997, Arthur a Fahey 1990, Harmon et al. 1990).

Tématiku odumretého dreva môžeme nájsť vo viacerých prácach habilitanta. V práci **P15 Merganičová – Merganič – Vorčák (2004)** hodnotíme zásobu moderového dreva v lesných porastoch národnej prírodnej rezervácie Babia hora. Výsledky poukazujú, že v skúmanej oblasti je možné so 68% pravdepodobnosťou očakávať priemernú zásobu moderového dreva v rozsahu 144.62 $m^3$/ha ± 19.81 $m^3$/ha, resp. 56.95% ± 8.95% zo zásoby živých stromov. Analýza vplyvu skúmaných faktorov potvrdila predpoklady, že vývojové štádium štatisticky signifikantne ovplyvňuje zásobu moderového dreva. Jeho vplyv je však významný v nižších polohách rezervácie, kým na hornej hranici lesa je štruktúra porastov natoľko rozpojená, že vývoj lesa prebieha v nepretržitom cykle a štádiá je už pomerne ťažko odlíšiť. Zaujímavým poznatkom je aj zistenie, že v štádiu optima predstavuje zásoba moderového dreva približne 20% zo zásoby živých stromov a to bez ohľadu na výškovú kategóriu. Ďalším skúmaným faktorom bola nadmorská výška, ktorej vplyv sa štatisticky preukázal len v rámci hodnotenia absolútneho množstva moderového dreva. Po relativizácii k zásobe živých stomov sa jej vplyv nepotvrdil, pretože množstvo moderového dreva koreluje so zásobou živých stromov a tá úzko koreluje s rastúcou nadmorskou výškou.

Uvedené poznatky sú významným príspevkom k obohateniu doterajších znalostí o zásobe odumretého dreva, ktorú možno v človekom nenarušených lesných ekosystémoch očakávať. Hodnoty jeho množstva dokumentujú, že odumreté drevo je trvalou súčasťou prirodzených lesných ekosystémov na hornej hranici lesa a preto je žiadúce venovať mu zvýšenú pozornosť.

V práci **P16 Merganičová – Merganič (2010)** sme stanovili uhlíkové zásoby v hrubom odumretom dreve v smrekových pralesoch v prírodnej rezervácii Babia Hora na Slovensku. Analýzy zistili, že uhlíkové zásoby silne závisia od vývojového štádia a nadmorskej výšky. Najväčšie uhlíkové zásoby v hrubom odumretom dreve sú v štádiu rozpadu, ktoré je charakteristické najväčším množstvom najmenej rozloženého dreva (Obrázok 4). S nárastom nadmorskej výšky uhlíkové zásoby v hrubom odumretom dreve klesajú v dôsledku nižšej produktivity lesa, ktorá sa prejavuje menšími rozmermi stromov na hornej hranici lesa, kde sa rozdiely medzi vývojovými štádiami strácajú.



**Obrázok 4** **Zásoba (A) a množstvo uhlíka (B) v odumretom dreve v jednotlivých vývojových štádiách a 8 stupňoch rozkladu definovaných podľa Holeksu (2001) v porastoch NPR Babia Hora (Merganičová a Merganič 2010)**

# 5. Záver

Zisťovanie stavu a vývoja lesov je významnou súčasťou hospodárskej úpravy lesov. Matematická kvantifikácia znakov lesa je nástroj, ktorý umožňuje nielen objektívnejšie ohodnotiť, ale v konečnom dôsledku aj lepšie pochopiť a popísať vzťahy v ekosystéme.

Predkladaná práca predstavuje súhrn šestnástich pôvodných vedeckých prác. Vo všetkých prácach sa využívajú matematicko-štatistické výberové dizajny ako aj široká paleta matematicko-štatistických metód zhodnotenia údajov vrátane moderných viacrozmerných metód a geoštatistických prístupov. Z dosiahnutých výsledkov, teoretických a praktických skúseností odporúčame pri inventarizáciách a monitoringu lesa navrhovať komplexné informačné spektrum. Pri výbere atribútov dávať silný dôraz na štruktúru lesa, pretože štrukturálna výstavba lesných porastov je veľmi dôležitou charakteristikou stavu a vývoja porastov, ktorá úzko súvisí so všetkými procesmi prebiehajúcimi v ekosystéme. Pri návrhu výberových dizajnov odporúčame vzhľadom na stav informačných technológií a stav a rozsah informačných databáz v širšom meradle uplatňovať princípy stratifikovaných výberov. Aplikácia uvedených princípov môže významným spôsobom spresniť hodnotenie stavu a vývoja lesných porastov.

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# 7. Príloha

## 7.1. Článok v kapitole 2. Výberové metódy – progresívny nástroj zisťovania a monitorovania lesných ekosystémov

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JOURNAL OF FOREST SCIENCE, *54*, 2008 (10): 476–483

# Some methodological aspects of the National Forest Inventory and Monitoring in Slovakia

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**ABSTRACT**: The work presents the conceptual information about the National Forest Inventory and Monitoring in Slovakia. It introduces some methodological approaches to the field data collection (determination of tree heights by two-phase method, regression formulas for tree volumes and assortments of forest tree species, quantification of deadwood volume in sample plots) and biometrical models prepared for data processing and generalisation of the results. The design and conception of Slovak National Forest Inventory and Monitoring were set with the aim to enable providing complex and integrated information about the state and changes of production and ecological characteristics of the forest ecosystems.

**Keywords**: tree heights; tree volume; deadwood volume; biometrical models; Slovak forestry

## Basic conception of the National Forest Inventory and Monitoring in Slovakia

In Europe, Slovakia belongs to the countries with a relatively high proportion of forestland (40%), rich in tree species composition, with variable natural conditions, and with intensive forest management. It has a long tradition in detecting the forest conditions. At present, three different systems exist for assessing the forest state – survey of natural conditions and forest ecology, detection of forest stands condition for forest management needs, and the national monitoring of forest health conditions executed yearly in a grid of 16 × 16 km. Lately, the fourth system was established, namely National Forest Inventory and Monitoring (NFIM) in Slovakia, which was first executed in 2005 and 2006. Its aim is to detect the conditions of all components of forest ecosystems periodically and to observe the changes on national and regional levels, as by other NFIs. In the presented paper, we provide information on the basic conception of the Slovak NFIM and on some methodical aspects, which can be interesting for a wider expert society in this field.

National Forest Inventory and Monitoring in Slovakia 2005–2006 was executed upon the decision of the Ministry of Agriculture from July 1, 2004. It was performed on all lands covered by forest tree species, i.e. on forest lands and on other forested lands including the protected areas. Slovak NFIM was drawn up as a combined aerial-terrestrial sampling method with a systematic distribution of sample units over the whole country. In the aerial images, sampling units are circular plots of the size of 2,500 $m^2$ distributed in a grid of 2 × 2 km, which serve for the distinction between the land categories Forest/Non-forest and for the determination of the forest area. The terrestrial inventory plots are established in a grid of 4 × 4 km. In these plots, information covering the whole information spectrum is collected. The information spectrum is broad, as it consists of more than 100 variables, while four different types and sizes of sample plots (Fig. 1) are optimised to their attributes. In the terrain, the plots are permanently invisibly fixed, which enables periodical observations of all attributes and variables by the same method and at the same place over a longer time period. Data is collected using the computer-

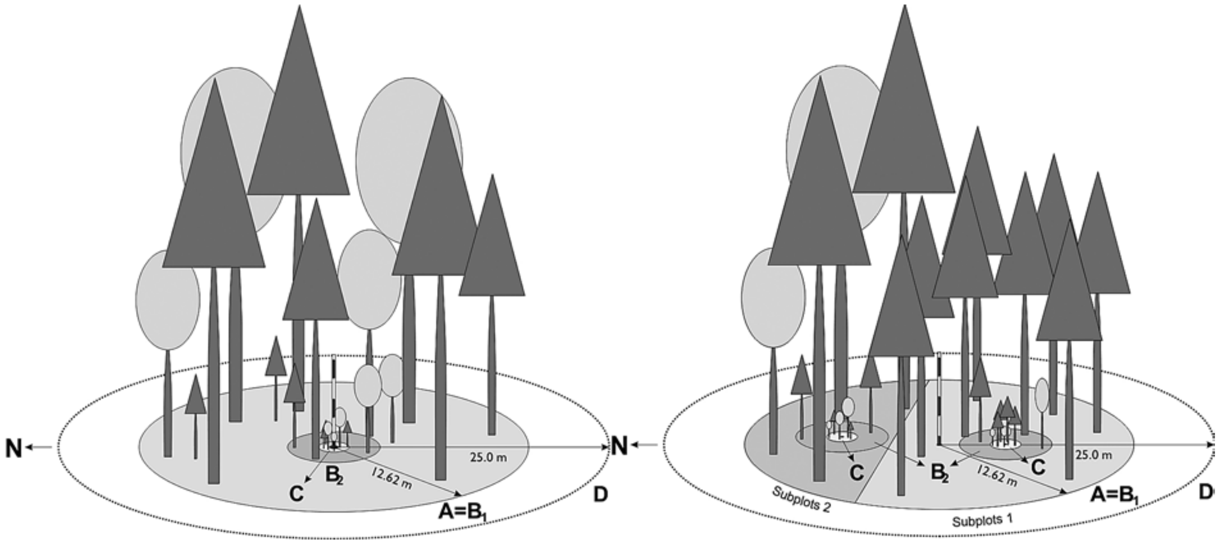

Fig. 1. Scheme of the sample plot (on the left-hand side without subplots, on the right-hand side divided into 2 subplots): A – a constant circle with radius $r$ = 12.62 m, on which terrain, site, stand and ecological characteristics, food sources for animals are detected and lying deadwood and stumps are inventoried, B – two concentric circles ($r$ = 12.62 m and 3 m) for detecting tree characteristics on trees with diameter at breast height $d_{1.3} \geq 12$ cm (B$_1$) and $d_{1.3} = 7–12$ cm (B$_2$), C – a variable circle for thin trees with diameter $d_{1.3} < 7$ cm, its radius $r$ = 1.0 m, 1.41 m or 2.0 m is chosen according to tree density, D – an enlarged constant circle with radius of 25 m established for the inventory of forest edges, forest roads and water sources

based Field-Map Technology (IFER 1999–2006). The whole implementation of NFIM is ensured by the National Forest Centre in Zvolen in accordance with detailed methodological instructions (Šmelko et al. 2005, 2006).

Slovak NFIM in its form fulfils the latest scientific and practical requirements for the complex detection and periodical comparison of the forest condition. Its precision level is restricted to a large extent by the lack of financial resources, and thus the grid of the sample plots (4 × 4 km) is relatively sparse. This will ensure sufficient precision of the final data only on the national level (by forest area 1%, by timber volume 1.5%), while on the regional level the precision will be 2–4 times lower. The next Slovak NFIM is presumed to be carried out in years 2014–2016 in a denser grid (terrestrial 2 × 2 km, and in low forested areas 1.41 × 1.41 km, and an aerial grid of 1 × 1 km, or 500 × 500 m) to obtain more exact data.

### Determination of tree heights by two-phase method – a combination of estimation and measurement

The determination of tree heights in the sample plots belongs to serious methodological problems. On one hand, "one tree principle" is in general pushed forward, i.e. the requirement to know the heights of all trees in a sample plot, which enables to record the forest height structure in its whole variation range and is also optimal for the derivation of other variables (tree volume and its increment, assortments etc.). On the other hand, from the economical point of view, one is forced to consider the measurement of tree heights on a smaller number of trees (sample trees), and to assign to the rest of the trees the average height value from the local height curve derived from the sample plot or from the general height tariff. This method has several disadvantages – it reduces the real variability of heights and can cause deviations in the height of individual trees by several metres.

Based on our previous research (Šmelko 1994), a two-phase method, i.e. the combination of estimation ($E$) and measurement ($M$), was chosen for the Slovak NFIM. First, the heights of all $n_1$ trees in the sample plot are estimated (qualified ocular estimation is ensured by previous training). Next, a subsample of $n_2$ trees is determined, and the heights of these trees are measured. For example, each second or third tree is selected preferably from higher trees (according to the principle of unequal probabilities). It is specified that a minimum of 10 trees have to be measured. If there are less than 20 trees in the sample plot, all trees are measured. During the subsequent data processing, the estimated heights $h_E$ are rectified using the PPP-sampling theory with a multiple quotient $\bar{q}$ as follows:

$$h_{i(korig)} = h_{i(E)} \times \bar{q} \Rightarrow \bar{q} = \frac{\sum_{i=1}^{n_2} q_i}{n_2},$$

$$q_i = \frac{h_{i(M)}}{h_{i(E)}}, \quad i = 1, 2 \ldots n_2 \qquad (1)$$

The necessity for rectification is examined by a statistical test. In the case that the quotient $\bar{q}$ does not differ from 1.00 significantly, the rectification is not needed, and the height estimated is considered to be equal to that measured (i.e. deviations are not systematic, but have a random character and are in tolerance with natural measurement variability).

The presented method meets both above-mentioned requirements – it provides tree heights in the whole variation range, while its work and time demands are acceptable and the results are sufficiently precise. The experience obtained from the database showed that only in 21% cases the height estimations deviated from the measurements systematically (i.e. they were biased), in general they very closely correlated with the measurements, the variability of $q_i$ ($s_q$%) being only 9.8%.

### Set of regression formulas for tree volumes and assortments of forest tree species

Timber volume determination and its assortment structure is a key task of every NFI, while several specific conditions must be met. For automated data processing, appropriate mensurational relations expressed in a mathematical form are required. The results obtained are to be stated in the volume units used in national conditions, and at the same time comparable on a wider international scale. The results should also provide effective background information for more comprehensive utilisation.

Considering these demands, the following solution was taken for the NFIM of the SR. The suitability of the existing volume and assortment tables and of their mathematical models was verified with regard to the purpose of the NFIM. It was shown that the volume regression formulas $v = f(d_{1.3}, h)$ for 12 forest tree species (Petráš, Pajtík 1991) satisfactorily describe the tree volumes ($v$) over the whole range of diameters ($d_{1.3}$) 0.1–100 cm and heights ($h$) 1.3–45 m (see Fig. 2). Only small corrections or substitute solutions were necessary. In the case of less frequent tree species, the volume formulas for related tree species (in accordance with morphological stem similarity) were used. It was decided, that the tree volumes would be determined in three volume units as follows:

(1) commercial timber (i.e. wood with minimum diameter at the top end 7 cm) inside bark, which is usually used in home practice,
(2) commercial timber outside bark used in most European NFIs,
(3) total tree volume outside bark, which will be used for determining the carbon content in woody biomass and in its basic components (tree, stem, branches, bark).

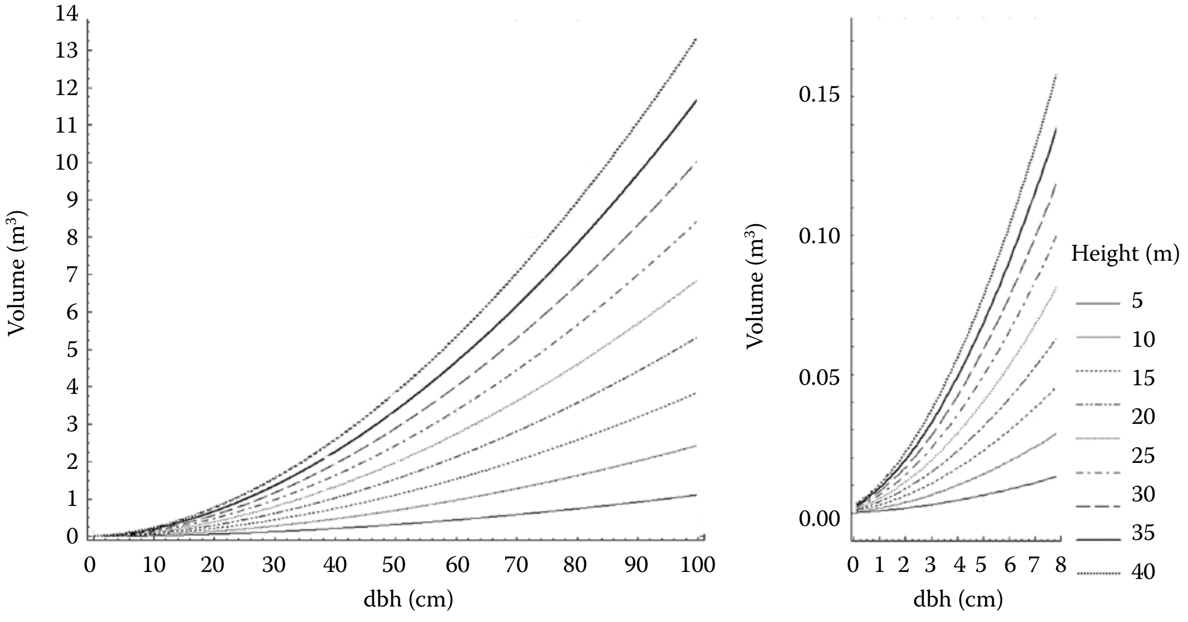

Fig. 2. Course of the volume formula $v = f(d_{1.3}, h)$ for spruce and its stem volume outside bark

The differences in those three volume units are actually rather high – e.g. standing volume per 1 hectare for all tree species from the NIML database resulted in 266–298–334 m$^3$, i.e. their relative indeces, being 1.00–1.12–1.25, respectively. Mathematical models for the partition of tree volume into 6 different assortment types (Petráš, Nociar 1991) were also shown as usable. They are derived for all main tree species, the input data being the tree diameter and height, quality of the bottom third of the stem (A, B, C), stem damage (yes, no), and in the case of the beech also the age and growth area (flysch). In the outputs of the Slovak NFIM, aggregated assortment types will be used, and for monitoring the changes in the quality structure of the forest stands relative proportions of trees in individual quality classes will be determined. Information about tree volumes from the models of volume and assortment tables is interconnected. By now, the use of another diameter $d_k$ from a higher part of the stem in the model $v = f(d_{1.3}, h)$ has not been considered. Although also in Slovak conditions Ďurský and Šmelko (2002) found that adding another diameter $d_{7m}$ or $d_{0.3h}$ improves the description of an actual stem shape of the tree and increases the precision of the tree volume determination (standard error will decrease by 0.62), the necessary three-argument volume modules $v = f(d_{1.3}, h, d_k)$ are not available at the moment.

## Quantification of deadwood volume in sample plots

Lately, standing and lying deadwood in forest ecosystems has become more and more significant and hence, its detection was included within almost all NFIs in Europe. The assessment methods for obtaining necessary information vary between the countries; they differ in the definitions of individual parts of this wood, in the lower limit from which it is recorded, and in detection details. While in the case of small-sized wood only the estimation of its coverage in the sample plot is usually carried out, for larger deadwood its volume is also determined.

In the Slovak NFIM, the chosen methodology allows to quantify the volume of all deadwood, both large and small-sized. Standing dead trees are inventoried by the same method as living trees. In the case of the lying large deadwood (with minimum top diameter outside bark 7 cm), its length and diameter at both ends of the piece situated within the sample plot are measured, and its volume is calculated by Smalian's method. The stumps from felled or dead trees are recorded if their diameter is 7 cm or more (at the standard height of 0.2 m above ground), their height and diameter on the cut section are measured, their volume is determined by stereometry, while the shape of the bottom stem part is considered in a simpler form (using the models of morphological curves for all main tree species).

For the lying small-sized wood, two-phase detection was tested:

(1) The first phase is carried out on each sample plot, or a subplot. The following characteristics are estimated: relative coverage of small-sized lying deadwood, prevailing group of tree species (coniferous, broadleaved), its average diameter (with precision of 1 cm), and average decomposition grade. Relative coverage stands for the percentual proportion of the total area of the sample plot, which would be covered by small-sized lying deadwood if all pieces were placed side by side. In the case that deadwood is huddled together, or placed into a pile, it is estimated what area this wood would cover after its dismantling.

(2) The second phase of detection is carried out only on each fourth sample plot (with a random start, e.g. on sample plots No. 2, 6, 10, etc.). Its aim is to determine the volume of small-sized wood in m$^3$, which corresponds to the area, or to the relative coverage of small-sized wood estimated during the first detection phase. This is achieved on the basis of sample piles taken as follows:

- From the occurring small-sized wood with the diameter of 1–7 cm, a sample pile with dimensions $W$ (width) and $L$ (length) is created in the selected sample plot. Individual pieces of small-sized wood are placed side by side as densely as possible, while the width $W$ of the sample pile should be approximately 1 m and its length $L$ should correspond to the average length of pieces with the diameter of up to 1 cm at the top end. The pieces can be placed once from the bottom end and once from the top end.
- For each sample pile, which is delimited by the range poles, the following characteristics are assessed. Its width $W$ and length $L$ are measured with precision of 0.05 m, the prevailing tree species and prevailing decomposition grade are estimated, and the diameters of all small-sized wood pieces are measured in the half of their average length $L/2$ with a simple measuring tool (Fig. 3).
- Using the data obtained, a biometric model is derived, which expresses the real wood volume of densely placed small-sized wood in an area of 1 m$^2$ as a function of the tree species and average small-sized wood diameter, and if necessary,

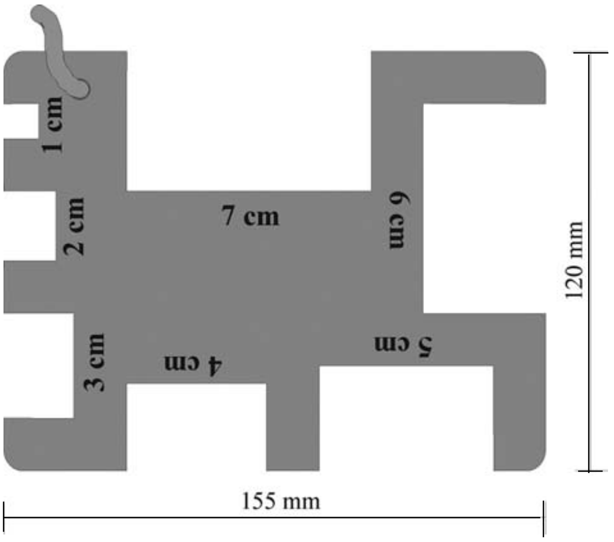

Fig. 3. Simple measuring tool used for measuring diameter of small-sized wood

also of other attributes influencing the given relation. Using this model, the volume of small-sized lying deadwood will be estimated also on other NFIM sample plots. Fig. 4 presents such a model derived from the 2005 and 2006 database. As can be seen, the relation between the volume and average diameter is tight and hence, well applicable.

Apart from the described second phase, another alternative was also tested, namely the line intersect sampling (Shiver, Borders 1996). In each fourth sample plot, two perpendicular lines were established, one in the direction North-South, the other in the direction West-East. With all pieces of small-sized lying deadwood, diameter $d_i$ was measured at the point of intersection with the line with a simple measuring tool, with precision of 1 cm. The volume of small-sized deadwood $T$ in m$^3$ per 1 hectare was directly derived from the measured diameters $m$ of small-sized wood pieces using the formula

$$T = \frac{\pi^2}{8L} \sum_{i=1}^{m} d_i^2 \qquad i = 1, 2 \,..... \, m \qquad (2)$$

(valid regardless of wood pieces length). This variant showed to be less suitable for the volume estimation than the first one, probably because of the insufficient length $L$ of lines set for this purpose (quadruple the radius of the sample plot = 50.48 m).

### Biometrical models used to generalise the results from sample plots for the whole inventoried territory

The data obtained from the sample plots have to be numerically processed and generalised for the whole inventoried territory using specific mathematical-statistical (biometrical) models, which cannot be universal but have to correspond to the used sampling design of the NFI and the properties of detected variables. First, it is necessary to consider if the sample plots are equal or variable in size, if they are distributed at random or systematically over the inventoried territory, and if the variables are quantitative or qualitative (categorical). The aim is to derive parameters applicable to the entire country or its parts on the basis of a relatively small sample size (from $n$ sample plots), and to determine the precision frames of their determination.

In this contribution, we discuss only two of such parameters – total and mean values of the stand quantitative variable, and the relative proportion of the tree qualitative variable. The models are derived

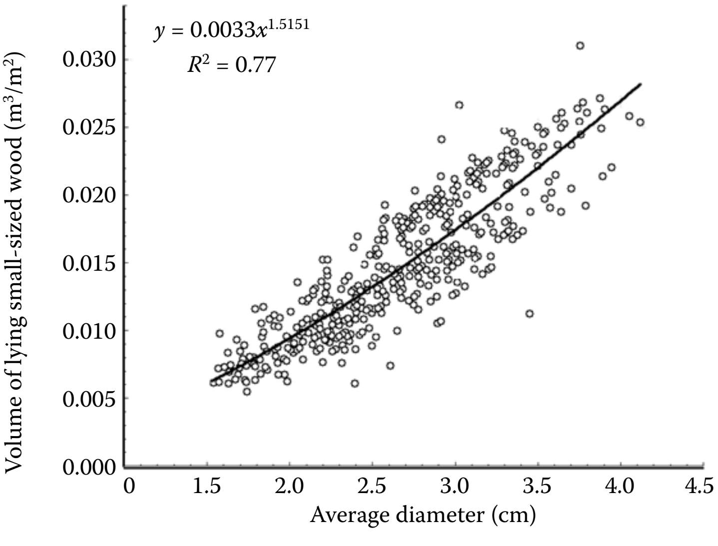

Fig. 4. Volume of small-sized lying deadwood (m$^3$) placed at 1 m$^2$ as a function of its average diameter

with regard to the fact that the Slovak NFIM has a systematic sampling design and that the stand and tree variables of trees with diameter $d_{1.3}$ equal to or above 12 cm were determined in the sample plot $B_1$, which is of a constant size of 500 $m^2$. However, in the cases where the sample plots were situated on the boundary Forest/Non-forest or encompassed different forest categories (different age, ownership category etc.), the sample plots were divided into smaller parts – subplots, resulting in a variable area of the sample units.

### Estimation of parameters of the stand quantitative variable

Let us assume that we evaluate the stand quantitative variable $Y$, e.g. timber stock, the number of trees etc. The target parameter, which has to be determined, is $\tau_Y$ – actual total value of variable $Y$ in the whole forest area $A$. This is equal to the sum of values $y_i$ of all $i = 1, 2 \ldots N$ individuals (trees) in the population

$$\tau_Y = \sum_{i=1}^{N} y_i \qquad (3)$$

or to the forest area $A$ multiplied by average $\mu_Y$ related to 1 ha

$$\tau_Y = A\mu_Y \qquad (4)$$

In the Slovak NFIM, model (4) is used. Area $A$ is determined from the sample results of aerial and terrestrial inventory. Average $\mu_Y$ is estimated by sample mean $\bar{Y}$ obtained by measuring the variable $Y$ on $m_i$ trees situated in $n$ sample plots, each of an area $X_i$. Standard error of estimation $\bar{\tau}_Y$ from the sample results is:

$$S_Y = \sqrt{A^2 S_{\bar{Y}}^2 + \bar{Y}^2 S_A^2} \qquad (5)$$

The estimation of $\mu_Y$ by $\bar{Y}$ can be executed by either of the two methods described below.

**A) The method "Ratio of Means".** This model is generally valid for random sampling (Loetsch, Haller 1973; Cochran 1977; Schaeffer et al. 1990, etc.). It is based on the averages, or on the sums of values of the quantitative variable $Y_i$ and the area of the sample plot $X_i$, where the average $\mu_Y$ is estimated from the proportion $\bar{R}$ with standard error $S_{\bar{R}}$

$$\bar{R} = \frac{\bar{Y}}{\bar{X}} = \frac{\sum_{i=1}^{n} Y_i}{\sum_{i=1}^{n} X_i} \qquad (6)$$

$$S_{\bar{R}} = \sqrt{\frac{\sum_{i=1}^{n}(Y_i - \bar{R}X_i)^2}{n(n-1)\bar{X}^2}} = \sqrt{\frac{\sum_{i=1}^{n}(Y_i^2 + \bar{R}^2\sum_{i=1}^{n}X_i^2 - 2\bar{R}\sum_{i=1}^{n}X_iY_i}{n(n-1)\bar{X}^2}} \qquad (7)$$

The magnitude of standard error (7) is influenced by the variability of $Y_i$ and $X_i$ values, as well as by their correlation ($r_{YX}$). Similarly, the relative standard error is derived from the relative standard errors of these components according to the relation

$$S_{\bar{R}}\% = \sqrt{(S_{\bar{Y}}\%)^2 + (S_{\bar{X}}\%)^2 - 2r_{YX}S_{\bar{Y}}\%S_{\bar{X}}\%} \qquad (8)$$

**B) The method "Mean of Ratios".** This model was recommended by Saborowski and Šmelko (1998), and Šmelko and Saborowski (1999) for systematic sampling of unequally sized sample plots. On the basis of the theoretical analysis and computer simulations, the authors found that in the case of systematic design, the probability to be selected into the sample is higher for larger sample plots than for smaller plots, what causes a systematic deviation (bias) in the estimations. Therefore, in each sample plot ($i$) the sample data $Y_i$ need to be recalculated to equal area (1 ha) using the following formula:

$$Y_{\text{ha}}(i) = \frac{Y_i}{X_i} \qquad (9)$$

These hectare values $Y_{\text{ha}}(i)$ are used for the estimation of $\mu_Y$ and $S_{\bar{Y}}$ as follows:

$$\bar{Y}_{\text{ha}} = \frac{\sum_{i=1}^{n} Y_{\text{ha}(i)}}{n} \qquad (10)$$

$$S_{\bar{Y}\,\text{ha}} = \sqrt{\frac{\sum_{i=1}^{n}(Y_{\text{ha}(i)} - \bar{Y}_{\text{ha}})^2}{n(n-1)}} =$$

$$= \sqrt{\frac{\sum_{i=1}^{n}(Y_{\text{ha}(i)}^2 - \frac{\left(\sum_{i=1}^{n} Y_{\text{ha}(i)}\right)^2}{n}}{n(n-1)}} \qquad (11)$$

A preliminary assessment of the data from the Slovak NFIM by both methods provided e.g. for commercial timber (i.e. wood with minimum diameter at the top end 7 cm) inside bark with all tree species these results:

$\bar{R}$ = 266.2 m$^3$, $S_{\bar{R}}$ = ± 5.15 m$^3$ and $\bar{Y}_{\text{ha}}$ = 263.9 m$^3$, $S_{\bar{Y}\text{ha}}$ = ± 5.16 m$^3$.

The differences in the average value 2.3 m$^3$ (0.9%) and in standard error 0.01 m$^3$ (0.2%) are not very high. Further spatial analyses, e.g. geostatistics and correlation analyses, did not reveal any significant systematic trends in the distribution of the values of basic variables over the whole country. For example, the correlation coefficients calculated for basal area per hectare within the distance of 50 km fluctuate between –0.12 and 0.34. This shows that the spatial autocorrelation between the values is low and practically negligible. Due to these facts, the first model A, was applied in the evaluation of NFIM data.

## Estimation of parameters of tree qualitative variable

Let us assume that we evaluate a tree qualitative variable, for example the relative proportion of trees π in quality classes A, B, C. We estimate the proportion $\pi_A$ by, the sample proportion $p_A$ and its standard error $S_{pA}$ based on the number of trees $a_i$ belonging to class $A$ and on the total number of trees $m_i$ on individual sample plots $i = 1, 2 \dots n$. This is a typical cluster sampling with unequal numbers of individuals because the number of trees $m_i$ will always vary, even if the area of the sample plots is constant. Thus, the proportion $p_A$ and its standard error $S_{pA}$ has to be determined using the model "Ratio of Means":

$$p_A = \frac{\sum_{i=1}^{n} a_i}{\sum_{i=1}^{n} m_i} \tag{12}$$

$$S_{pA} = \sqrt{\frac{\sum_{i=1}^{n} a_i^2 + p_A{}^2 \sum_{i=1}^{n} m_i^2 - 2p_A \sum_{i=1}^{n} a_i m_i}{n(n-1)\bar{m}^2}} \tag{13}$$

It can be proved, that in this case the estimate derived from binomial distribution cannot be used

$$p_A = \frac{\sum_{i=1}^{n} a_i}{\sum_{i=1}^{n} m_i} \tag{14}$$

$$S_{pA} = \sqrt{\frac{p_A(1-p_A)}{\sum_{i=1}^{n} m_i - 1}} \tag{15}$$

as this is applicable only to one-tree sampling (sampling of individual trees over the whole area). Although this method gives a good estimate of $p_A$, the calculated standard error is incorrect, having a much lower value. Likewise, the approach based on the proportions $p_{A(i)}$ assigned to each sample plot individually is not applicable

$$p_{A(i)} = \frac{a_{A(i)}}{m_i} \quad \Rightarrow \quad p_A = \frac{\sum_{i=1}^{n} p_{A(i)}}{n}$$

$$S_{pA} = \sqrt{\frac{\sum_{i=1}^{n} (p_{A(i)} - p_A)^2}{n(n-1)}} \tag{16}$$

This method could be used only if the total number of trees $m_i$ in the sample plots were the same in all sample plots. The discrepancy in the results obtained from these three methods is documented in the following example. The results document the proportion of spruce trees in quality class A calculated from the Slovak NFIM database:

| | Ratio of means | Binomial distribution | Method ad (16) |
|---|---|---|---|
| $p_A$ = | 0.1235 | 0.1235 | 0.1411 |
| $S_{pA}$ = | ± 0.0162 | ± 0.0056 | ± 0.0158 |

The presented considerations demonstrate that the data processing of the Slovak NFIM 2005–2006 varies with regard to the model characteristics and features of the evaluated variable.

## CONCLUSION

The presented article gives information on the basic characteristics of the Slovak NFIM, which was first executed in years 2005–2006 as a pilot project and its implementation at the same time. We also present some methodological approaches to the field data collection and biometrical models prepared for data processing and generalisation of the results. The NFIM methodology makes use of the existing international experience and knowledge from our own research at a maximum rate. It is characterised by a high variation in the selection of the design of sample plots, in the assessment of variables, and in their biometric evaluation. The aim was to optimise the methods in such a way that they may best suit the features of the detected variables, applied sampling design, and economical requirements.

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Received for publication March 28, 2008

Accepted after corrections July 17, 2008

## Niektoré metodické aspekty národnej inventarizácie a monitoringu lesov na Slovensku

**ABSTRAKT**: Príspevok prezentuje základnú koncepciu Národnej inventarizácie a monitoringu lesov (NIML) Slovenska, ktorá sa po prvýkrát uskutočnila v rokoch 2005–2006. Opisuje niektoré metodické princípy terénneho zberu údajov (určovanie výšok stromov dvojfázovou metódou, regresné rovnice uplatnené pri stanovení objemu a sortimentácii stromov lesných drevín, spôsob kvantifikácie objemu mŕtveho dreva na skusných plochách) a biometrické modely pripravené pre spracovanie údajov a zovšeobecnenie výsledkov. Výberový dizajn a celá koncepcia NIML boli navrhnuté tak, aby umožňovali vo zvolených časových intervaloch poskytovať komplexné a integrované informácie o stave a zmenách produkčných a ekologických charakteristík lesných ekosystémov na celoštátnej i regionálnej úrovni.

**Kľúčové slová**: výška stromov; objem stromov; objem odumretého dreva; biometrické modely; slovenské lesníctvo

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Eur J Forest Res (2004) 123: 157–165
DOI 10.1007/s10342-004-0032-8

ORIGINAL PAPER

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# Quantification of tree species diversity in forest stands—model BIODIVERSS

Received: 28 August 2003 / Accepted: 24 February 2004 / Published online: 20 August 2004

**Abstract** This paper presents a method for complex evaluation and quantification of tree-layer species diversity in forest ecosystems—model BIODIVERSS. The model was constructed for the assessment of species diversity on a small-scale (forest-stand) level, but may be applied with advantage to regional and large-scale inventories. The probability of correct classification of species diversity degree is relatively high; e.g. with a sampling intensity of only 1.5%, the correctness of the classification by the model is already 90%. Model BIODIVERSS is a simple and practical tool. It can be used directly during fieldwork, since for its application only a pocket calculator is needed.

**Keywords** Biodiversity · Tree species diversity · Forest inventory · Model · Discriminant analysis

## Introduction

In 1992 the United Nations Conference on Environment and Development was held in Rio de Janeiro, where, in addition to other documents, the Convention on Biological Diversity was ratified. The government of the Slovak Republic, recognising the importance of preserving biological diversity, agreed with and joined the Convention in May 1993. On 1 April 1997, the "National Strategy for Biodiversity Protection in Slovakia" was approved by the Slovak government in response to the regulations listed in Article 6 of the Convention. The strategy became a fundamental programming act for the implementation of the Convention on Biological Diversity in the Slovak Republic, and within the specified time horizons, it is being developed and realised according to the action plans that were approved by the government of the Slovak Republic (Straka and Guziová 1998).

The term "biological diversity" is still not uniformly defined. In the Convention on Biological Diversity (Article 2), this term refers to "the variability among living organisms from all sources including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part". The term biological diversity as it is described in the Convention thus covers not only the diversity within a species and between species, but also the diversity of ecosystems (Sibl et al. 1996). The trilogy of the standard components included in the definition, i.e. diversity at the genetic, species and ecosystem levels, became a conventional definition of biodiversity.

As the above-stated biodiversity definition implies, species diversity represents one of several important biodiversity components. Its evaluation in a forest ecosystem is mainly focused on higher plants. A great number of different methods can be used for the evaluation of species diversity (e.g. see Ludwig and Reynolds 1988; Krebs 1989). All of the proposed methods are usually based on at least one of the following three characteristics (Bruciamacchie et al. 1995):

1. Species abundance—the oldest and the most simple understanding of species diversity expressed as a number of species in an ecosystem
2. Species evenness—a measure of the equality in species composition in a community
3. Species heterogeneity—a characteristic encompassing both species abundance and evenness

The most popular methods for measurement and quantification of species diversity are certainly species diversity indices. During their historical development,

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the indices have been split into three categories: indices of species abundance, species evenness and species heterogeneity (Ludwig and Reynolds 1988; Krebs 1989). The indices of each group explain only one of the above-mentioned components of species diversity. In order to evaluate and quantify the species diversity in a complex way, we considered it necessary to develop a method encompassing all the individual parts. The model BIODIVERSS presented in this work is a proposal of such a method.

## Methods

Data used in this study come from permanent inventory plots (PIPs) established in 1977 with the aim of examining the representativeness of forest inventory sample plots (Šmelko 1979). The PIPs are located in forest stands with the structure typical of the forests of the Forest School Enterprise of the Technical University Zvolen. The areas of the PIPs range from 3–7 ha. All trees on each PIP are numbered, and their position within the forest stand is described by the *x*, *y* coordinates in a rectangular system. For every tree the following basic biometrical characteristics were determined: tree species, diameter at breast height, tree height and volume.

For the purposes of this work, eight permanent inventory plots were used. Each permanent inventory plot represents a specific stand structure. Together they cover the whole range of species composition from very diverse stands (PIP 8 and 1) to pure homogenous stands (PIP 3). Since none of the real PIPs could have been characterised as a stand with a very high species diversity, abundance and evenness, such a plot was artificially simulated using the growth simulator SILVA 2.2 (Pretzsch et al. 1998). The modelled PIP (plot number 8) represents the maximum species diversity that can occur within the context of modern forestry. The basic descriptive statistics of the PIPs are presented in Table 1.

Every PIP was divided into sample subplots of optimum size. The optimum size was determined in an optimisation study of the precision and costs of the sample inventory (Šmelko 1968) and is defined as an area encompassing approximately 15–25 (on average 20) trees. From this definition it is clear that the optimum size of the sample inventory plot depends on the forest stand density.

On the optimum-size subplots, forest stand variables [basal area (*BA*), stand volume (*V*), number of trees (*N*)] were determined for each tree species separately. From these stand variables, 12 different species diversity indices were calculated for every sample plot: *N*0 (Hill 1973), *R1* (Margalef 1958), *R2* (Menhinick 1964), $\lambda$ (Simpson 1949), *H′* (Shannon and Weaver 1949), *N*1, *N*2 (Hill 1973), *E1* (Pielou 1975, 1977), *E2* (Sheldon 1969), *E3* (Heip 1974), *E4* and *E5* (Hill 1973). The use of different variables (*BA, V, N*) in the calculation of indices is well

**Table 1** Basic characteristics of permanent inventory plots

| Permanent inventory plot | Area (ha) | Age (years) | *BA* ($m^2$ $ha^{-1}$) | *V* ($m^3$ $ha^{-1}$) | *N* ($ha^{-1}$) | Number of species | Species composition from *BA* (%) | Size of sample plot (ha) | Number of sample plots | Species diversity degree[a] |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | 3.12 | 90 | 27.9 | 307.8 | 542 | 10 | Beech 51.8, fir 21.8, hornbeam 20.6, maple 1.6, cherry 1.4, lime 1.0, elm 0.7, birch 0.5, oak 0.3, ash 0.2 | 0.04 | 78 | 3 |
| 2 | 2.98 | 75 | 31.4 | 275.6 | 710 | 7 | Beech 45.9, oak 44.9, fir 6.7, hornbeam 2.1, maple 0.1, cherry 0.04, lime 0.02 | 0.03 | 99 | 2 |
| 3 | 5.16 | 50 | 26.2 | 208.5 | 950 | 6 | Oak 99.6, hornbeam 0.4, spruce 0.02, fir 0.02, beech 0.02, lime 0.02 | 0.02 | 258 | 1 |
| 4 | 6.18 | 70 | 41.9 | 554.4 | 766 | 5 | Spruce 58.2, fir 27.0, oak 14.7, maple 0.04, beech 0.004 | 0.03 | 206 | 2 |
| 5 | 6.48 | 70 | 28.4 | 254.3 | 920 | 7 | Oak 52.4, hornbeam 39.1, lime 5.4, beech 2.4, maple 0.4, cherry 0.1, fir 0.1 | 0.02 | 324 | 2 |
| 6 | 6.48 | 70 | 23.5 | 209.1 | 803 | 7 | Oak 52.4, hornbeam 39.8, lime 4.9, beech 2.2, maple 0.5, fir 0.1, cherry 0.1 | 0.02 | 324 | 2 |
| 7 | 6.48 | 90 | 29.6 | 300.1 | 672 | 6 | Oak 50.5, hornbeam 39.5, lime 6.0, beech 3.4, maple 0.5, cherry 0.1 | 0.03 | 216 | 2 |
| 8[b] | 6.25 | 72 | 38.6 | 434.1 | 681 | 10 | Spruce 21.6, fir 20.5, larch 18.3, beech 15.8, pine 8.1, oak 6.7, maple 2.7, elm 2.7, lime 2.7, ash 1.1 | 0.03 | 208 | 4 |

[a] *1* Low, *2* medium, *3* high, *4* very high

[b] Artificially simulated permanent inventory plot

supported. If the number of trees $N$ is used for the calculation, the size of the tree is neglected, whereas when using the basal area $BA$, the tree size is taken into account through its diameter, or more precisely by the square of its diameter. The variable stand volume, $V$, includes, in addition to the diameter, the second main tree characteristic, i.e. its height. Before the final calculations of the indices were performed, the stand characteristics determined on the sample subplots had to be standardised on a per hectare basis, since the subplots differed in size.

As already mentioned, each PIP in our data set represents a specific degree of diversification. In order to enable a simple but objective comparison of biodiversity between the stands, a scale of diversity classification was created. Using the data, four degrees of species diversity were defined: low (1), medium (2), high (3) and very high (4) species diversity. The species diversity degree of a particular PIP was assessed from complex evaluation of its structure with regard to:

1. The number of tree species in a stand
2. Species distribution over the forest area
3. Species composition and evennessThe final degree of species diversity of each PIP is shown in Table 1.

## Model BIODIVERSS

The fundamental method of the model BIODIVERSS is predictive discriminant analysis (Cooley and Lohnes 1971; Huberty 1994; StatSoft 1996). Using the parameterisation set consisting of four permanent inventory plots (1, 3, 5 and 8), four discriminant equations were derived, one for each species diversity degree. These equations are the core of the model, since they serve to classify an evaluated forest stand into one of the four species diversity degrees. In the proposed model BIODIVERSS, five indices of species diversity were taken as independent variables: $R1$, $R2$, $\lambda$, $H'$ and $E1$. These indices represent all three groups of diversity and are defined as follows:

$$R1 = (S-1)/\ln(N) \quad \text{(Margalef 1985)} \tag{1}$$

$$R2 = S/\sqrt{N} \quad \text{(Menhinick 1964)} \tag{2}$$

$$\lambda = \sum_{i=1}^{S} w_i^2 \quad \text{(Simpson 1949)} \tag{3}$$

$$H' = -\sum_{i=1}^{S} [w_i \ln(w_i)] \quad \text{(Shannon and Weaver 1949)} \tag{4}$$

$$E1 = H'/\ln(S) \quad \text{(Pielou 1975, 1977)} \tag{5}$$

where $S$ is the number of species, $N_i$ is the number of trees per ha on a sample plot, $w_i$ is the proportion of species $i$ on a sample plot calculated from $BA$ per hectar.

The indices of diversity and evenness are calculated from the species proportion assigned from basal area per hectare $BA$ (Merganič 2001) to account for the size of individuals in the forest stand. The given combination of the indices gave us the best classification results from the variety of tested combinations.

The general formula of the discriminant model is:

$$\text{Discriminant score}_j = R1 \cdot b_{j1} + R2 \cdot b_{j2} + \lambda \cdot b_{j3} + H' \cdot b_{j4} + E1 \cdot b_{j5} + b_{j6} \tag{6}$$

where $b_{j1-6}$ are regression coefficients and $j$ is the degree of species diversity (1–4). The examined forest stand is assigned the species diversity degree for which the calculated discriminant score is maximum.

In the second step, the quality of the proposed discriminant model BIODIVERSS was analysed along with the influence of the sampling intensity on the probability of correct classification. This part of the work was performed as follows. On each PIP, i.e. also on those PIPs that were not included in the parameterisation data set, we performed repeated sampling while changing the number of optimum-size sample subplots. The number of subplots in one sampling ranged from 1–30, which corresponds to the sampling intensity ($f$) of 0.31–38.46%. The maximum possible number of repetitions was set to 70 to ensure the independence of samples, since the smallest PIP 1 can be divided into 78 optimum-size subplots. The actual number of sampling repetitions varied among the PIPs and depended on the number of subplots per sampling and the area of the plot. Thus, in cases where we simulated samples consisting of one subplot, the sampling was repeated 70 times on each PIP. However, if the sampling comprised two subplots, the number of independent repetitions on each PIP was different, e.g. on PIP 1 we could repeat this sampling 39 times under the constraint that the samples were independent.

The results of this experiment are the probability models of correct classification that describe the relation between the sampling intensity and the proportion of correct estimations in the particular degree of species diversity. The probability model has the exponential form:

$$p_{M_j} = \left[1 - e^{\left(-f/a1_j\right)}\right]^{a2_j} \tag{7}$$

where $p_M$ is the modelled probability of correct classification, $j$ is the degree of species diversity, $f$ is the sampling intensity in percent defined as the ratio of the sampling area to the total area of PIP, and $a1$ and $a2$ are the regression coefficients.

For correct interpretation of the evaluation results, it is necessary to define the reliability of the final value $p_M$, i.e. to estimate the confidence interval in which the real probability of correct classification, $p_{PCC}$, will lie with a certain probability, since the $p_M$ value is calculated from samplings and can therefore be affected by the error. This error consists of two elements: (1) an error of the probability model of correct classification (due to its fitting to data), $SE_M$, and (2) an error, $SE_{PM}$, resulting from the distribution theory of sampling ratios, since the probability is in fact the ratio of correct estimations at a particular degree of species diversity. The distribution of sampling ratios is usually binomial and can be determined in two different ways:

1. If the sampling size $n$, i.e. the number of repeated simulations with the same design, is large and the real ratio $p$ is within the interval (0.3, 0.7) or if $n \times p \times (1-p) > 9$, the binomial distribution of sampling ratios, $p_M$, can be approximated to normal distribution. The confidence interval $CI$ is then defined as:

$$CI \rightarrow p = p_M \pm z_{\alpha/2} \cdot SE_{PM} = p_M \pm z_{\alpha/2} \cdot \sqrt{\frac{p_M \cdot (1-p_M)}{n}} \tag{8}$$

The overall error of probability of correct classification, $SE_{PCC}$, is calculated as:

$$SE_{PCC} = \sqrt{SE_M^2 + SE_{PM}^2} \tag{9}$$

$SE_M$ is the error of the probability model and $SE_{PM}$ is the error resulting from the distribution theory of sampling ratios.The confidence interval, in which the real probability of correct classification $P_{PCC}$ will occur with a certain probability, will be determined as follows:

$$CI \rightarrow p_{PCC} = p_M \pm z_{\alpha/2} \cdot SE_{PCC} \tag{10}$$

$p_M$ is the modelled probability of correct classification and $SE_{PCC}$ is the error of the probability of correct classification.

2. If the sampling size $n$ is small, or if the real ratio $p$ is smaller than 0.3 or larger than 0.7, the distribution of sampling ratios $p_M$ is distinctly left- or right-skewed. In such cases, the lower and upper limits of the confidence interval for the sampling ratio $p_M$ can be derived after the transformation of this variable, e.g. using the Fischer transformation. The limits determined in this way are then used for the calculation of the range of the overall error. As has already been stated, the overall error, $SE_{PCC}$, consists of two elements, whereby each element can have a different shape of distribution. Due to the lack of data it is not possible to analyse this problem in detail. Thus, the presented estimation is simplified, based on the theory that the minimum probability is 0, and the maximum probability is 1. If the lower limit of the confidence interval, i.e. $p_M - z_{\alpha/2} \times SE_{PCC}$, is less than zero or the upper limit exceeds 1, those values will be forced to 0 or 1 respectively. The opposite limit of the confidence interval is then calculated by adding or subtracting double the overall error $SE_{PCC}$, i.e. $0 + 2 \times SE_{PCC}$ for the upper limit, or $1 - 2 \times SE_{PCC}$ for the lower limit. This asymmetric confidence interval, in which the real probability of correct classification, $P_{PCC}$, will occur with a certain probability*, is therefore, in the case* $p_M + z_{\alpha/2} \times SE_{PCC} > 1$, defined with the following inequality:

$$CI \rightarrow p_{PCC} = \left[\left(1 - 2 \cdot z_{\alpha/2} \cdot SE_{PCC}\right) \leqslant p_M \leqslant 1\right] \tag{11}$$

## Results and discussion

The quantification of species diversity degree is performed in two steps. First, the discriminant score is calculated for each degree of species diversity (1–4) using Eq. (6) from the real values or means of diversity indices that enter the model as independent variables. In the second step, the estimated scores are compared. The stand of interest is assigned the degree of species diversity for which the calculated discriminant score is maximum. The values of the regression coefficients in the discriminant equations are presented in Table 2. Table 3 contains the results of the classification matrix for the parameterisation data set. These results show that 88.65% of all the forest stands were assigned to the correct species diversity degree, whereby the best classification was achieved for the outer degrees 1 and 4, for which the correct classification exceeded 95%. For the transition degrees 2 and 3, we obtained lower values of correct classification (86.34 and 73.08% respectively).

In Table 4, the statistical characteristics of the model are presented. According to the values of Fischer $F$ statistics and Wilks' lambda, we can say that the proposed discriminant model is statistically highly signifi-

**Table 2** Regression coefficients of the discriminant model BIODIVERSS

| Diversity index | Regression coefficient | Species diversity degree | | | |
|---|---|---|---|---|---|
| | | 1 | 2 | 3 | 4 |
| $R1$ | $b_1$ | −625.54 | −599.78 | −623.08 | −653.02 |
| $R2$ | $b_2$ | 623.89 | 616.99 | 737.29 | 779.46 |
| $\lambda$ | $b_3$ | 2,376.13 | 2,383.77 | 2,362.28 | 2,486.22 |
| $H'$ | $b_4$ | 1,208.42 | 1,194.35 | 1,196.97 | 1,282.84 |
| $E1$ | $b_5$ | 404.61 | 436.94 | 422.71 | 426.60 |
| | $b_6$ | −1,201.47 | −1,216.76 | −1,205.33 | −1,352.92 |

**Table 3** Classification matrix of the parameterisation set of the model BIODIVERSS

| Species diversity degree | Correct classification (%) | Number of sample plots per species diversity degree | | | |
|---|---|---|---|---|---|
| | | 1 | 2 | 3 | 4 |
| 1 | 95.74 | 247 | 11 | 0 | 0 |
| 2 | 86.34 | 25 | 278 | 18 | 1 |
| 3 | 73.08 | 2 | 15 | 57 | 1 |
| 4 | 95.17 | 0 | 0 | 10 | 197 |
| Total | 88.65 | 274 | 304 | 85 | 202 |

**Table 4** Statistic characteristics of the discriminant model BIODIVERSS

| Number of variables: 5 | | Number of groups: 4 | |
|---|---|---|---|
| Wilks' lambda: 0.02591 | | $^{1}F = 474.25^{***}$ | |
| Input variables | | | |
| Diversity index | Wilks' lambda | Partial lambda | $^{2}F$ |
| *R1* | 0.03036 | 0.85 | 53.43*** |
| *R2* | 0.03469 | 0.75 | 105.51*** |
| $\lambda$ | 0.03279 | 0.79 | 82.66*** |
| *H′* | 0.03852 | 0.67 | 151.66*** |
| *E1* | 0.03715 | 0.70 | 135.10*** |

$^{***1}F_{0.001(15,\ 2{,}581)} = 2.53$, $^{***2}F_{0.001(3,\ 935)} = 5.45$

cant (with significance ≤ 0.001). The Wilks' lambda can be interpreted in the following manner: if it is close to 0, the model is appropriate, if, on the other hand, the value is around 1, the model is not suitable. The partial lambda given in the third column of Table 4 provides us with the information about the contribution of each independent variable to the discrimination of the dependent variable. These values confirm that all indices implemented in the model are statistically significant, i.e. they all contribute to the discrimination of species diversity degree. The indices *H′*, *E1* and *R2* have the largest impact (Table 4).

The values of Wilks' lambda given in the second column of Table 4 represent another possibility for examining the individual contributions of indices in the classification. These values show the changes of Wilks' lambda after omitting a particular index from the model. For example, if the index *H′* is omitted, the value of Wilks' lambda is increased from 0.02591 to 0.03852, i.e. the classification will be worsened.

In order to explain the classification graphically, the canonical analysis was applied to the data set. Figure 1 shows the position of the groups of sample plots with the same species diversity index and their approximate borders. From this figure it is obvious that the outer categories of species diversity degrees have the highest probability of correct classification because their overlap with the neighbouring classes is the smallest. In addition, the great variance of the points assigned to one species diversity degree (Fig. 1) demonstrates that the model was derived from quite heterogeneous data. These results suggest that the usage of the model is not restricted to spatially homogeneous objects.

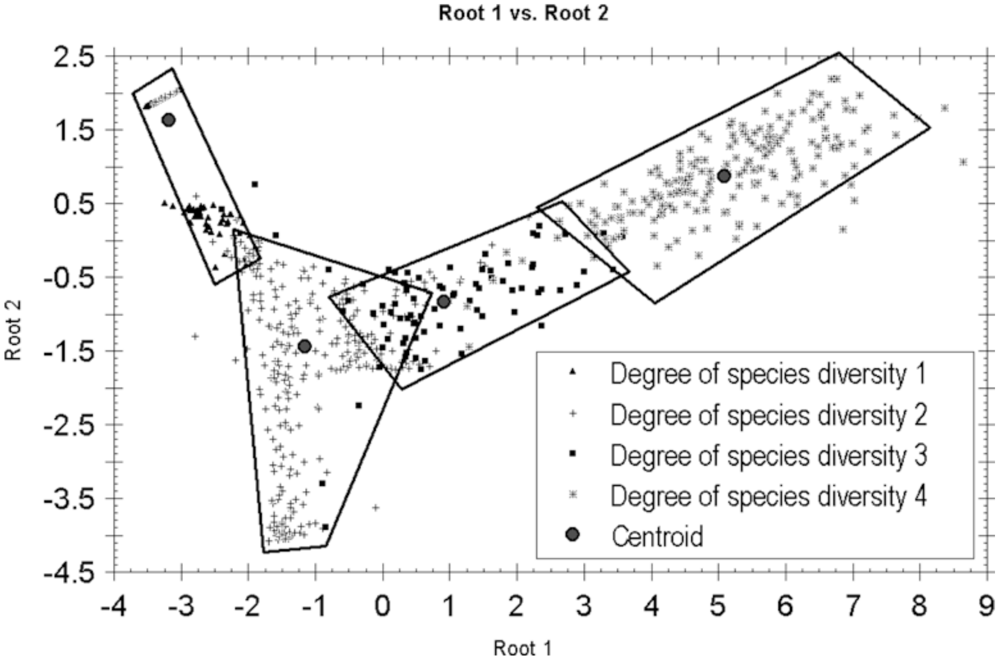

**Fig. 1** Graphical interpretation of the classification of species diversity degrees using canonical analysis

Table 5 encompasses the mathematic-statistical characteristics of the models derived for the estimation of the probability of correct classification by the model BIODIVERSS. Two probability models of correct classification were calculated, one for the species diversity degrees 1 and 4 with better classification (Fig. 2) and the other for the degrees 2 and 3 (Fig. 3). The degrees were combined due to the lack of data. For a simple and

**Table 5** Mathematic-statistical description of the models used for the estimation of probability of correct classification of species diversity degree

| Diversity degree | Regression coefficients | | Sum of squared differences | Correlation coefficient ($R$) | $100 \times R^2$ | $n$ | Standard error ($SE_M$) |
|---|---|---|---|---|---|---|---|
| | *a1* | *a2* | | | | | |
| 1 and 4 | 0.376655 | 0.100957 | 0.001069348 | 0.84695 | 71.732 | 20 | 0.007 |
| 2 and 3 | 7.198133 | 0.063952 | 0.384644015 | 0.60593 | 36.716 | 60 | 0.080 |

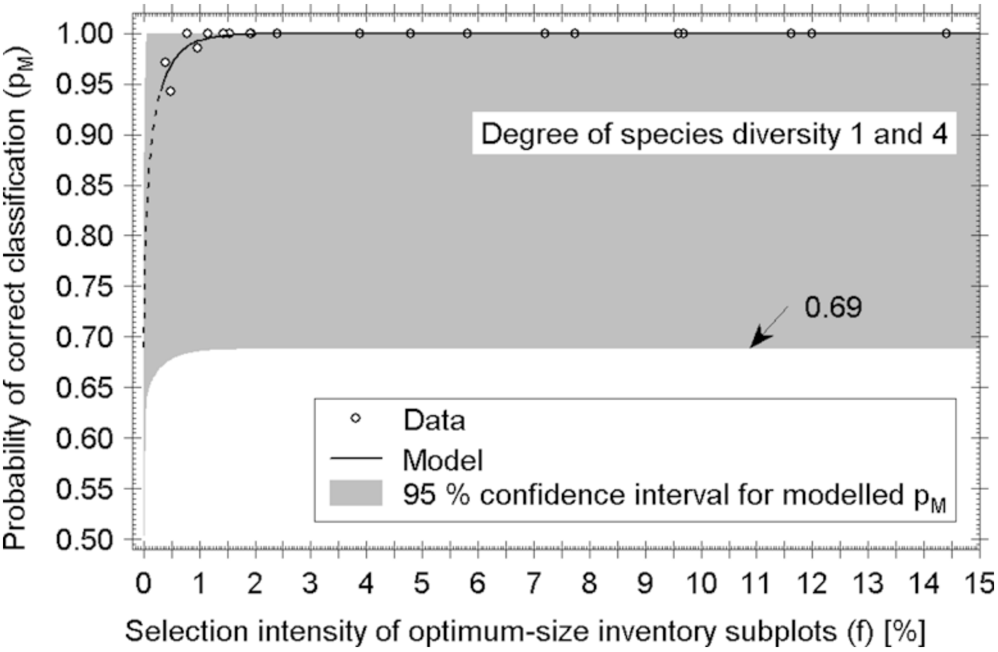

**Fig. 2** Probability model of correct classification for species diversity degrees 1 and 4 with 95% confidence interval

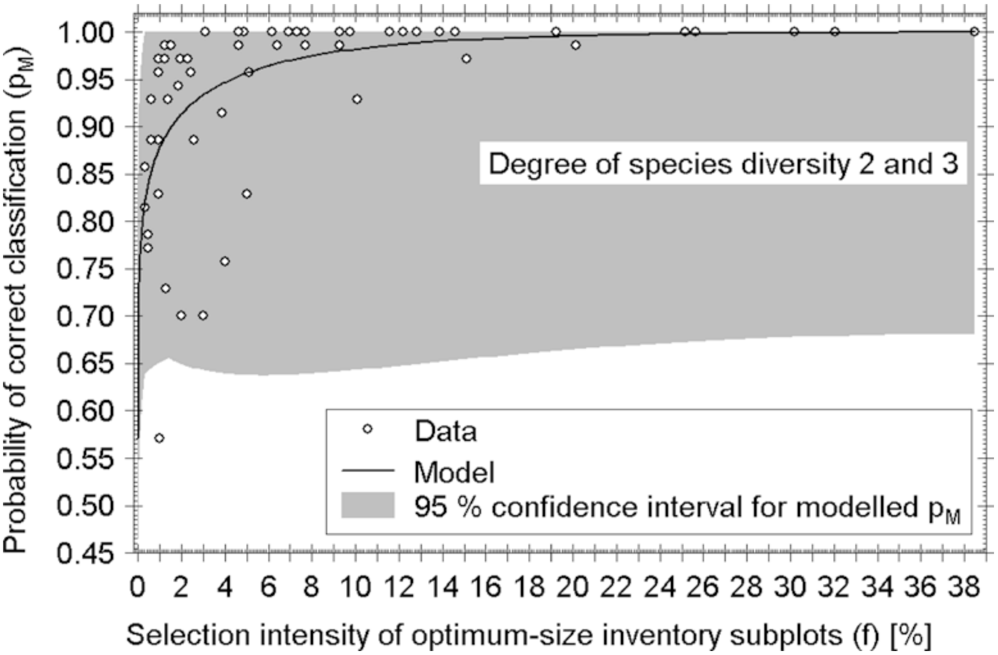

**Fig. 3** Probability model of correct classification for species diversity degrees 2 and 3 with 95% confidence interval

**Table 6** Characteristics of the forest stand no. 12 (3.2 ha)

| Optimum size of the sample plot: 0.03 ha | | | | | | | | | Number of sample plots: 4 | | | | Sampling intensity: 3.75% | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Plot no. | Tree species/species composition | | | | | | | | Sum | | Stand characteristics | | Diversity indices | | | | |
| | Beech | | Hornbeam | | Oak | | Maple | | | | | | | | | | |
| | *BA* ($m^2$ $ha^{-1}$) | $w_i$ | *BA* ($m^2$ $ha^{-1}$) | $w_i$ | *BA* ($m^2$ $ha^{-1}$) | $w_i$ | *BA* ($m^2$ $ha^{-1}$) | $w_i$ | $\Sigma BA$ ($m^2$ $ha^{-1}$) | $\Sigma w_i$ | *V* ($m^3$ $ha^{-1}$) | *N* ($ha^{-1}$) | *R1* | *R2* | $\lambda$ | *H′* | *E1* |
| 1 | 16.83 | 0.45 | 4.49 | 0.12 | 7.85 | 0.21 | 8.23 | 0.22 | 37.40 | 1.00 | 379.4 | 700.0 | 0.46 | 0.15 | 0.31 | 1.27 | 0.92 |
| 2 | 9.63 | 0.30 | 8.35 | 0.26 | 11.24 | 0.35 | 2.89 | 0.09 | 32.10 | 1.00 | 318.4 | 650.0 | 0.46 | 0.16 | 0.29 | 1.30 | 0.93 |
| 3 | 15.54 | 0.60 | 6.22 | 0.24 | 4.14 | 0.16 | 0.00 | 0.00 | 25.90 | 1.00 | 253.7 | 500.0 | 0.32 | 0.13 | 0.44 | 0.94 | 0.86 |
| 4 | 11.97 | 0.38 | 6.62 | 0.21 | 9.14 | 0.29 | 3.78 | 0.12 | 31.50 | 1.00 | 309.3 | 550.0 | 0.48 | 0.17 | 0.29 | 1.31 | 0.94 |
| Arithmetic mean of 1–4 | | | | | | | | | 31.73 | - | 315.2 | 600.0 | 0.43 | 0.15 | 0.33 | 1.21 | 0.91 |

practical applicability of the models, the confidence intervals of the average modelled values were also assessed. These intervals define the range within which the real probability value should occur with the probability of 95%.

## An example of the determination of species diversity degree using the model BIODIVERSS at the stand level

In the following, we present an example of how the model BIODIVERSS can be used for the assessment of species diversity within a single forest stand. For this purpose, a fictitious forest stand (no. 12) was used.

In this forest stand, the inventory of production and biodiversity characteristics was performed. During the inventory, four sample plots were established within the stand. Using their basic characteristics shown in Table 6, the degree of species diversity for the forest stand no. 12 can be assessed by the discriminant model BIODIVERSS. First, the diversity indices *R1–E1* were calculated. Next, the discriminant scores were determined from Eq. (6) with the regression coefficients listed in Table 2.

The values of the calculated discriminant scores are given in Table 7. The highest discriminant score was 1,253.6 for the third degree of species diversity. Thus, according to the calculations, the forest stand no. 12 belongs to the third degree of species diversity. The correctness of this classification was checked by the 95% confidence interval of the probability of correct classification calculated from Eqs. (10) or (11). From Fig. 4 we obtained the average probability of correct classification. In our example, for the sampling intensity $f = 3.75\%$, the average probability of correct classification is $p_M = 0.944$, i.e. 94.4% with the 95% confidence interval from 64–100%.

## Conclusion

This paper proposes a new method for the complex evaluation and quantification of species diversity in forest ecosystems. In contrast to species diversity indices that describe only one of the biodiversity components, i.e. either species abundance, evenness or heterogeneity, the suggested model BIODIVERSS estimates the species diversity degree of a stand using five diversity indices (*R1, R2,* $\lambda$, H′ and *E1*) and thus integrates all the partial biodiversity components. The method is based on the assumption that if high species diversity is observed in a small area within the forest stand, we can presume that the species diversity of the whole examined forest stand will also be high. The probability of correct classification of the species diversity degree using the model BIODIVERSS is relatively high. With only 1.5% sampling intensity, the success of classification already reaches approximately 90%.

**Table 7** Calculation of discriminant scores [see Eq. (6)] for forest stand no. 12 and all species diversity degrees. Forest stand no. 12 is categorised under diversity degree 3, because the highest score was achieved for that degree

| Diversity degree | $b_1$ | *R1* | $b_2$ | *R2* | $b_3$ | $\lambda$ | $b_4$ | $H'$ | $b_5$ | *E1* | $b_6$ | Discriminant score |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 1 | −625.54 | 0.43 | + 623.89 | 0.15 | + 2,376.13 | 0.33 | + 1,208.42 | 1.21 | + 404.61 | 0.91 | −1,201.47 | 1,240.44 |
| 2 | −599.78 | 0.43 | + 616.99 | 0.15 | + 2,383.77 | 0.33 | + 1,194.35 | 1.21 | + 436.94 | 0.91 | −1,216.76 | 1,250.30 |
| 3 | −623.08 | 0.43 | + 737.29 | 0.15 | + 2,362.28 | 0.33 | + 1,196.97 | 1.21 | + 422.71 | 0.91 | −1,205.33 | 1,253.16 |
| 4 | −653.02 | 0.43 | + 779.46 | 0.15 | + 2,486.22 | 0.33 | + 1,282.84 | 1.21 | + 426.60 | 0.91 | −1,352.92 | 1,247.36 |

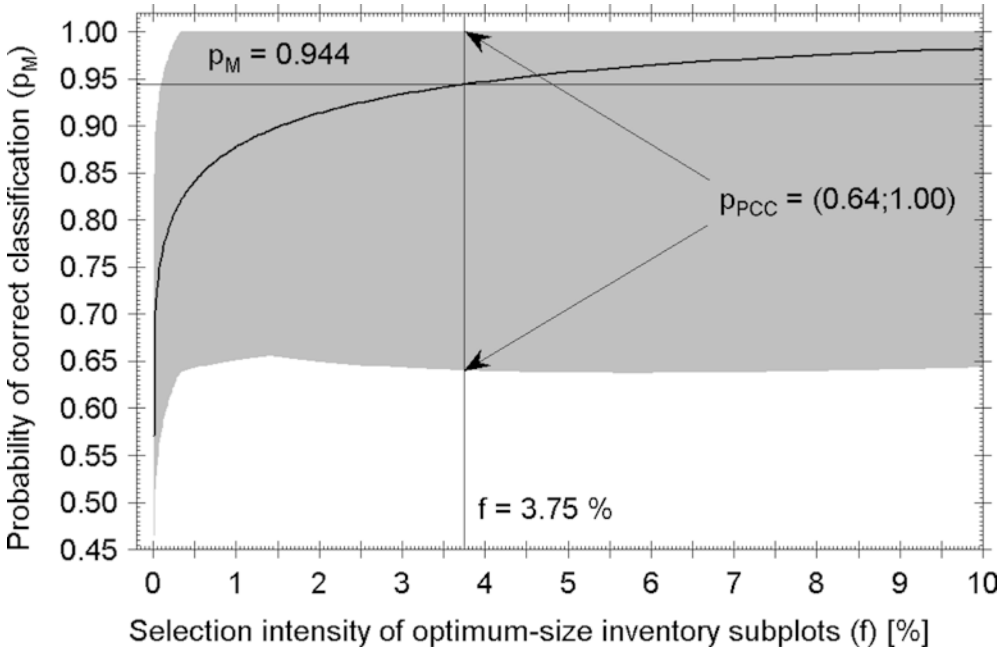

**Fig. 4** Probability of correct classification of the species diversity degree 3 with 95% confidence interval indicated by the *dark grey area*

Although the suggested model was constructed using data from only four stands, it has a wide valence, since it is, in fact, based on 865 optimum-size sample plots. The utilisation of optimum-size sample plots for the species diversity assessment of the tree layer eliminates the well-known problem arising from the strong dependency of species diversity on the size of the evaluated object, because here the particular diversity quantifier always represents a group comprising a similar number (approximately 20) of trees. Therefore, the correct application of the model BIODIVERSS requires the utilisation of optimum-size sample plots.

The presented model is a simple and practical tool that is easy to use. It can be used immediately during fieldwork because its application requires only a pocket calculator. Therefore, the implementation of the model into practice within enumeration surveys should cause no problems.

The model BIODIVERSS was designed for the determination of biological species diversity of the tree layer on a forest-stand scale. Nevertheless, the method can also be applied to regional or large-scale inventories if we assume that species diversity index determined on a sample plot represents a certain part of the evaluated area.

Moreover, the model can be considered as a methodological proposal for a model applicable to the whole of Europe. To derive such a European model, it would be necessary to gather data from a large number of forest stands throughout Europe representing all degrees of species diversity and all growing stages. Of course, in this case a uniform species diversity quantification system should be applied throughout all of Europe. Such a model would be advantageous for a consistent evaluation of tree species diversity based on unambiguous quantitative characteristics, i.e. diversity indices, which would be valuable for the implementation of the Convention on Biological Diversity.

**Acknowledgements** This work was carried out as a part of the grant project VEGA SR 1/7053/20 "Integrated Scheme of Sampling Methods for Permanent Survey of Forest Ecosystems Status" and the project ITM EU 1902 "Implementing Tree Growth Models as Forest Management Tools". We gratefully acknowledge Robert Williams for providing helpful review comments.

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**Lesnícky časopis – Forestry Journal**
ročník 54, Supplement 1, 2008, s. 81–89

# DRUHOVÁ BOHATOSŤ VEGETÁCIE LESOV SLOVENSKA ZISTENÁ V RÁMCI NÁRODNEJ INVENTARIZÁCIE A MONITORINGU LESOV SR

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ŠEBEŇ, V., MERAGNIČ, J., BOŠEĽA, M.: Species richness of vegetation in Slovak forests determined by National forest inventory and monitoring in Slovakia. Lesn. Čas.–Forestry Journal, 54, Supplement, 1, 2008, p. 81–89.

The paper analyses the results of National Forest Inventory and Monitoring of Slovak Republic (NFIM SR) from the point of species richness of vegetation, which is considered to be an important element of biological diversity. Analysed data come from 1,419 inventory plots distributed over the whole area of Slovakia in 4×4 km square grid. Species richness and total number of species were assessed separately for forests located on forest and non-forest land, in individual altitudinal zones, and in integrated growth stages. The work presents the results with 95% confidence interval of the occurrence probability of species number in the groups of trees with diameter above 7 cm, shrubs, plants, and mosses. The results revealed valuable information about species richness of Slovak forests.

**Key words:** *species richness, National Forest Inventory and Monitoring in Slovakia, frequency analysis, vegetation*

Príspevok analyzuje výsledky Národnej inventarizácie a monitoringu lesov (NIML) Slovenskej republiky z hľadiska bohatosti rastlinných druhov ako významnej zložky biologickej diverzity. Empirický materiál pre analýzy pochádza z 1 419 inventarizačných plôch (IP) rozmiestených po celom území Slovenska v pravidelnej sieti 4×4 km. Druhová bohatosť sa zhodnotila pre lesy na lesných a nelesných pozemkoch, lesných vegetačných stupňoch a integrovaných rastových stupňoch. V triediacich kategóriách sa stanovil aj celkový počet druhov. Výsledky sa uvádzajú vrátane 95 % intervalov spoľahlivosti pravdepodobnosti výskytu počtu druhov pre stromy s hrúbkou nad 7 cm, krov, bylín a machov. Výsledky poukazujú na pomerne hodnotné zistenia druhovej bohatosti slovenských lesov.

**Kľúčové slová:** *druhová bohatosť, národná inventarizácia a monitoring lesov SR, frekvenčná analýza, vegetácia*

## 1. Úvod a problematika

Biologická diverzita je prirodzená vlastnosť lesných ekosystémov. Termín je definovaný ako „rôznorodosť" všetkých živých organizmov vrátane ich suchozemských, morských a ostatných vodných ekosystémov a ekologických komplexov, ktorých sú súčasťou (Dohovor o biologickej diverzite). Označuje teda nielen rôznorodosť v rámci druhov a medzi druhmi, ale aj rozmanitosť (diverzitu) ekosystémov (Sibl *et al.* 1996 *ex* Merganič 2001). Hodnotenie diverzity lesného ekosystému sa v prevažnej miere zameriava na vyššie rastliny. Existuje veľké množstvo spôsobov, ktorými je možné diverzitu hodnotiť, ale v podstate každý z nich je založený minimálne na jednom z troch nasledujúcich znakov (Bruciamacchie 1996):

1. druhová bohatosť (najstaršie a najjednoduchšie poňatie diverzity vyjadrenej iba na základe počtu druhov);
2. druhová vyrovnanosť (miera rovnomernosti zastúpenia jednotlivých druhov v spoločenstve);
3. druhová heterogenita (charakteristika zahŕňajúca druhovú bohatosť a vyrovnanosť v jednom).

Cieľom príspevku je analyzovať druhovú bohatosť vegetácie (stromy s hrúbkou nad 7 cm, kry, byliny a machy) v rámci rôznych kategórií pozemkov, lesných vegetačných stupňov a integrovaných rastových stupňov.

## 2. Materiál a metodika

Hodnotenie počtu druhov sa vykonalo z databázy údajov NIML SR. Zber údajov sa uskutočnil v rokoch 2005 – 2006, pri výberovej reprezentatívnej metóde s komplexným zisťovaním informácií o lesoch (Šmelko *et al.* 2006). V rámci NIML bola založená sieť trvalých inventarizačných plôch (IP) v spone 4×4 km na celom území SR. Celkový počet IP bol 3 071 a z toho na 1 419 IP sa nachádzal les. Na uvedených IP bolo vykonané komplexné zisťovanie.

Druhovú bohatosť stromov s hrúbkou nad 7 cm je možné z databázy NIML SR analyzovať dvojakým spôsobom. Prvý spôsob je posudzovanie počtu druhov drevín na približne rovnakom počte jedincov nachádzajúcich sa najbližšie k stredu IP, ktorý bol v rámci NIML SR na podklade doterajších vedeckých poznatkov stanovený na 20 jedincov – stromov (Merganič 2001, Merganič *et al.* 2004). Druhý spôsob je posudzovanie druhovej bohatosti na rovnakej plošnej výmere. V predkladanom príspevku analyzujeme druhovú bohatosť stromov zisťovanú prvým spôsobom. Hodnotenie druhovej bohatosti krov, bylín a tráv, a machov je založené na podklade redukovaných fytocenologických zápisov (zisťovala sa len prítomnosť druhu a dominancia pri diferenciálnych druhoch). Plocha, na ktorej sa fytozápis robil, mala výmeru 500 m$^2$ a iba v prípadoch, že IP padla na rozhranie nelesného a lesného pozemku, mala menšiu výmeru (7,9 % prípadov).

Analýza druhovej bohatosti vegetácie sa vykonala v nadväznosti na základné výstupy z NIML osobitne pre nasledovné kategórie: *kategória pozemku*, kde rozlišujeme les na lesných pozemkoch a les na nelesných pozemkoch, 8 *lesných vegetačných stupňov* (lvs) a 6 *integrovaných rastových stupňov* (1 – holina, nálet, nárast, kultúra, 2 – Mladina, žŕdkovina, žrďovina, tenká kmeňovina, 3 – Stredná, hrubá a veľmi hrubá kmeňovina, 4 – Zmiešaný rastový stupeň (RS) nižší, 5 – Zmiešaný RS vyšší a 6 – Obnovované porasty). Pre každý triediaci znak sa stanovil celkový počet druhov.

Počty druhov na jednotlivých IP boli zatriedené do tried druhovej bohatosti, kde pre stromy s hrúbkou nad 7 cm, kry a machy rozlišujeme 6 tried (0, 1, 2–3, 4–5, 6–7 a 8 a viac druhov) a pre byliny 6 tried so širším intervalom t. j. 0, 1–10, 11–20, 21–30, 31–40 a 41 a viac druhov.

Analýza údajov je založená na frekvenčnej analýze. Pre každú hodnotu relatívneho podielu triedy druhovej bohatosti je kvantifikovaná jej výberová chyba a stanovený 95 % interval spoľahlivosti, v ktorom by sa mala nachádzať skutočná hodnota pravdepodobnosti výskytu konkrétneho počtu druhov. 95 % interval spoľahlivosti je vypočítaný zovšeobecnením teórie o normálnom a Studentovom rozdelení, pri ktorej sa ako kritická hodnota rozdelenia použila hodnota 1,96. Pri asymetrickom intervale (hodnoty spodného intervalu klesli pod 0 a hodnoty horného intervalu prekročili 1) sa aplikoval postup Merganič – Šmelko 2004. Použili sa nasledovné vzťahy:

$$p_{trieda} = \frac{n_{trieda}}{n} \qquad [1]$$

$$SE_{p\_trieda} = \sqrt{\frac{p_{trieda}.(1-p_{trieda})}{n-1}} \qquad [2]$$

$$95\%IS = p_{trieda} \pm 1{,}96\ .\ SE_{p\_trieda} \qquad [3]$$

$p_{trieda}$ – podiel, frekvencia triedy druhovej bohatosti,
$n_{trieda}$ – počet IP v jednotlivých triedach druhovej bohatosti v rámci kategórie,
$n$ – celkový počet IP v kategórii,
$SE_{p\_trieda}$ – stredná chyba podielu triedy druhovej bohatosti.

Pred vlastnou frekvenčnou analýzou sa vzhľadom na nerovnakú výmeru IP pri hodnotení druhovej bohatosti krov, bylín a machov analyzoval vzťah medzi počtom druhov a veľkosťou zostávajúcej časti IP. Vzťah medzi počtom druhov a výmerou plochy, na ktorej bol zisťovaný je teoretický známy a ovplyvňuje interpretáciu výsledkov (Merganič 2001, Križová 1995). Pomocou regresnej analýzy a základného logaritmického modelu ($f(x) = a+b\ .\ ln(x)$) sa štatisticky posudzovala tesnosť uvedeného vzťahu. V prípade, že tesnosť tohto vzťahu bola štatisticky signifikantná (t-test, Šmelko 1998), bol počet druhov na IP s výmerou menšou ako 500 m² upravený podľa modelu. Uvedená analýza sa vykonala pre všetky analyzované veličiny a všetky kombinácie s triediacimi znakmi. Z hodnotených veličín sa štatisticky signifikantný vzťah medzi počtom druhov a výmerou zistil len pri druhovej bohatosti machov v kategórii pozemku les na nelesných pozemkoch ($R_{xy}$ = 0,19, t-test = 2,8, v 5. lvs ($R_{xy}$ = 0,13, t-test = 2,2) a integrovanom rastovom stupni – Mladina, žŕdkovina, žrďovina, tenká kmeňovina ($R_{xy}$ = 0,16, t-test = 3,4). V ostatných prípadoch išlo o nesignifikantný vzťah.

## 3. Výsledky

Uvádzajú sa podiely jednotlivých tried počtu druhov podľa zvolených kategórií.

### *A) Druhová bohatosť stromov, krov, bylín, tráv a machov na lesných a nelesných pozemkoch*

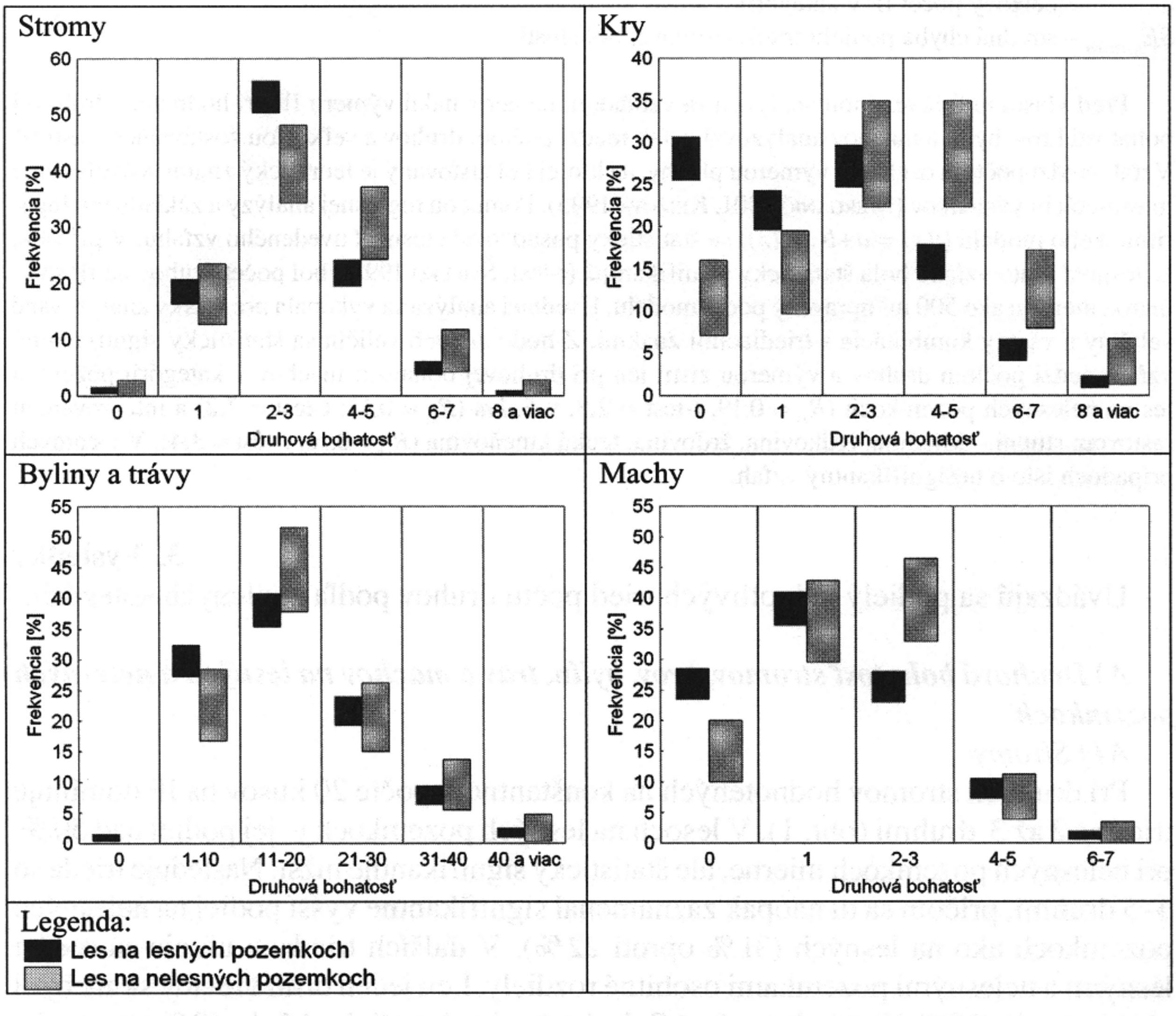
#### *A1) Stromy*

Pri druhoch stromov hodnotených na konštantnom počte 20 kusov na IP dominuje trieda s 2 až 3 druhmi (obr. 1). V lesoch na lesných pozemkoch je jej podiel nad 50 %, pri nelesných pozemkoch mierne, ale štatisticky signifikantne nižší. Nasleduje trieda so 4–5 druhmi, pričom sa tu naopak zaznamenal signifikantne vyšší podiel na nelesných pozemkoch ako na lesných (31 % oproti 22 %). V ďalších triedach už nie sú medzi lesnými a nelesnými pozemkami osobitné rozdiely. Len jeden druh dreviny sa zachytil na menej ako 20 % IP, trieda so 6 až 7 druhmi zaberá podiel od 5 do 10 % a viac ako

8 druhov netvorí vyšší podiel ako 1 %. Žiadne druhy sa nezaznamenali na holinách, resp. mladších rastových stupňoch, ktoré tvoria menej ako 1 %-ný podiel. Celkovo sa potvrdila druhová rozmanitosť a bohatosť drevín v lesoch. Spolu sa zaznamenalo 55 druhov stromov, z toho na lesných pozemkoch 54 a na nelesných 44. Informácie o súhrnom počte druhov sú však len informatívne, nedovoľujú porovnanie, pretože výmera lesov na nelesných pozemkoch je takmer 7-násobne menšia.

### *A2) Kry*

Triedenie podľa kategórie pozemku pri kroch ukázalo výrazné rozdiely v počte druhov. V lesoch na nelesných pozemkoch s približne rovnakými hodnotami dominujú triedy s 2–3 a 4–5 druhmi (obr. 1), spolu zaberajú asi 2/3 všetkých IP. S odstupom nasleduje trieda s jedným druhom (okolo 15 %), porovnateľný podiel zaberá trieda bez výskytu krov a s výskytom 6–7 druhov krov. Najnižší podiel (4 %) sa zistil pri druhovo



Obr. 1 95 %-ný interval spoľahlivosti podielu tried druhových početností a ich porovnanie podľa kategórií pozemkov najbohatšej triede s 8 a viac druhmi krov. Kry tvoria prirodzenú súčasť štruktúry lesov na nelesných pozemkoch a uplatňujú sa nielen ako ochranná zložka sukcesných vývojových štádií pri klíčení a odrastaní stromovitých drevín, ale zotrvávajú aj v ďalších vývojových štádiách.

V lesoch na lesných pozemkoch je naopak najzastúpenejšia trieda bez krov (takmer 30 %), nasledujú s mierne nižším podielom 2–3 druhy a 1 druh (viac ako 20 %). Od triedy so 4 a viac druhmi ich podiel výrazne klesá až po asi 2 % s 8 a viac druhmi krov. Okrem odstraňovania krov hospodárskymi opatreniami od obnovy cez výchovu porastov má na nízke zastúpenie krov vplyv aj vertikálna výstavba a spravidla hustý zápoj korún stromov zamedzujúci optimálny vývoj krov.

Priemerný počet krov na IP na lesných pozemkoch bol 2, na nelesných takmer 4. Celkovo sa zaznamenalo na lesných pozemkoch 50 druhov krov, na nelesných 30.

### *A3) Byliny a trávy*

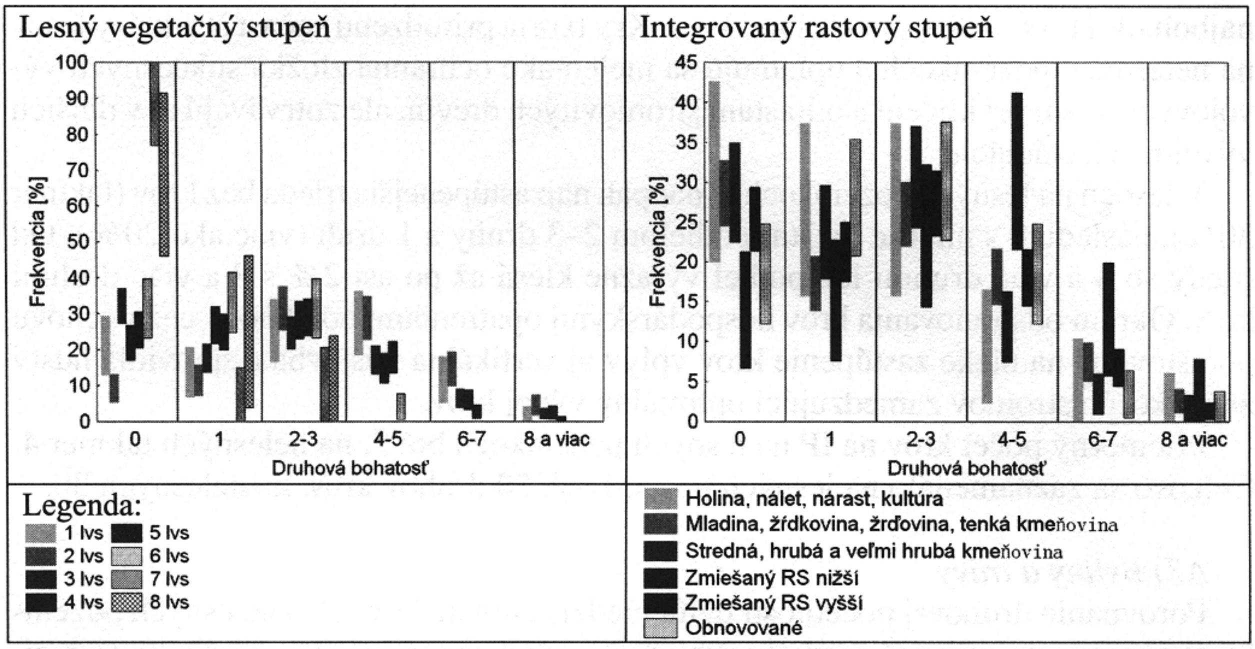
Porovnanie druhovej početnosti bylín medzi lesmi na lesných a nelesných pozemkoch nepriniesIo výrazné rozdiely. Najväčší podiel zaberá trieda s 11 až 20 druhmi, pri lesoch na lesných pozemkoch okolo 40 %, pri nelesných sa blíži k 45 %. Mierne vyšší podiel má trieda 1–10 druhov na lesných ako na nelesných pozemkoch. Triedy 31–40 druhov tvoria podiel okolo 10 %, nad 40 druhov menej ako 5 %. Priemerný počet druhov bol na lesných pozemkoch takmer 16, na nelesných 18. Podiel plôch bez vegetácie dosahuje menej ako 1 %, na nelesných pozemkoch sa takéto prípady vôbec nenašli. Spolu sa zaznamenalo 449 druhov bylín a 95 druhov tráv, z toho na lesných pozemkoch 426 druhov bylín a 90 druhov tráv, na nelesných pozemkoch 327 druhov bylín a 64 druhov tráv.

### *A4) Machy*

Početnosť druhov machov je spomedzi ostatnej vegetácie výrazne najnižšia. Zrejmý je rozdiel medzi lesnými a nelesnými pozemkami, nelesné sú o čosi bohatšie (podiel triedy 2–3 druhov je okolo 40 % oproti asi 26 % a naopak podiel triedy bez výskytu machov je okolo 15 % voči asi 26 % na lesných pozemkoch. Spolu sa na lesných pozemkoch zaevidovalo 38 druhov machov a na nelesných pozemkoch 24 druhov.

## ***B) Druhová bohatosť krov v lesných vegetačných stupňoch a integrovaných rastových stupňoch***

Porovnaním početnosti druhov krov podľa **lesných vegetačných stupňov** (obr. 2) sme zachytili výrazné rozdiely medzi 1. až 6. a 7. a 8. lvs. V posledných dvoch lvs dominuje trieda bez výskytu krov, nasleduje trieda s výskytom 1 druhu. Triedy s viac ako 4 druhmi sa v týchto lvs vôbec nevyskytli. Vo všetkých nižších lvs prevažuje trieda s 2–3 druhmi krov. Po nej nasleduje v prvom a druhom lvs početnejšia trieda 4–5 druhov, kým v treťom až šiestom lvs je to trieda s 1 druhom. Trieda bez výskytu krov však zaberá tiež výrazný, 10 až 30 % podiel. Trieda nad 8 druhov krov bez rozdielu lvs nezaberá viac ako 5 %.



Obr. 2 95 %-ný interval spoľahlivosti podielu tried druhových početností krov a ich porovnanie podľa lvs a integrovaného RS

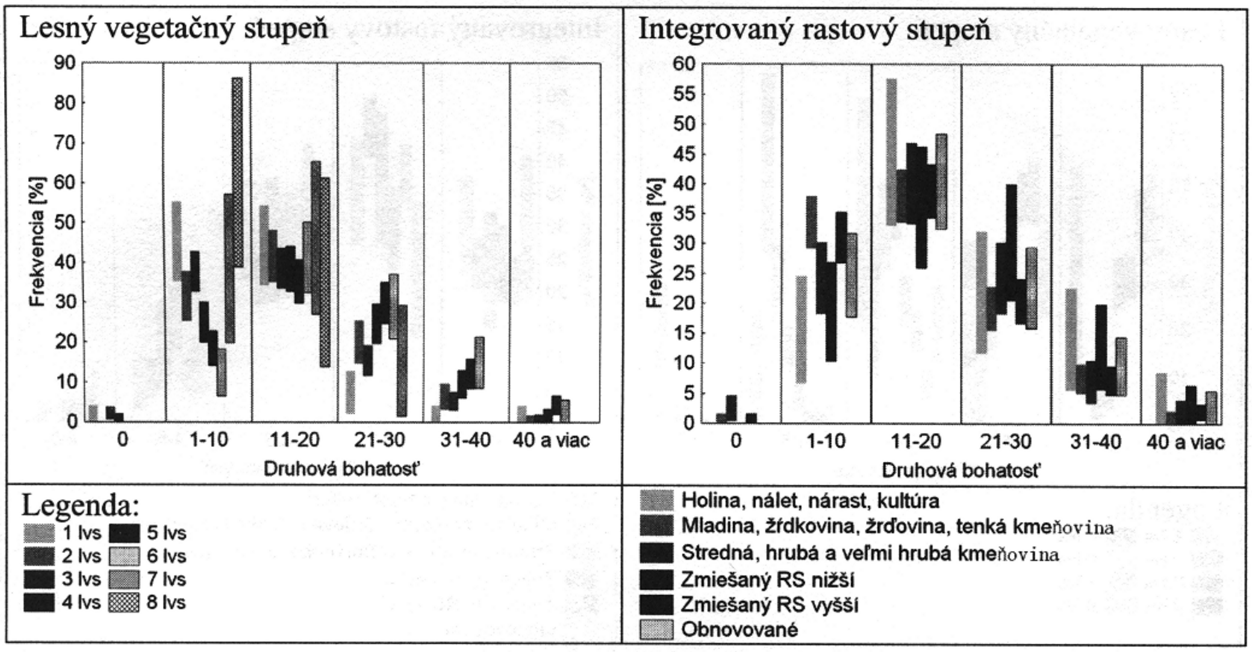
Zreteľne sa tu črtá tendencia klesania druhovej bohatosti krov so stúpajúcim lvs (na kry najbohatší je prvý a druhý lvs, nasledujú tretí, štvrtý, piaty a šiesty, najmenej druhov krov sa zaznamenalo v siedmom a ôsmom lvs).

Pri analýze početnosti druhov krov podľa **integrovaných rastových stupňov** (obr. 2) zisťujeme, že najnižšie počty početnosti druhov krov sa nachádzajú v mladých jednoetážových porastoch (holina, nálet, nárast), pri ktorých najväčší podiel zaberá trieda bez výskytu krov. So zvyšujúcim sa rastovým stupňom pozorujeme stúpajúci trend počtu druhov (maximum je dosiahnuté v zmiešanom RS vyššom). Pomerne veľkú druhovú bohatosť zaznamenávame aj v zmiešanom RS nižšom a v obnovovaných porastoch.

### *C) Druhová bohatosť bylín a tráv v lesných vegetačných stupňoch a integrovaných rastových stupňoch*

Pri **vegetačnej stupňovitosti** a početnosti bylinných a trávovitých druhov sa tiež prejavili zreteľné trendy (obr. 3). Druhovo najchudobnejší je 8., 7. a 1. lvs (takmer polovicu výskytu v nich tvorí trieda 1–10 druhov a druhú polovicu kategória 11–20 druhov). Trieda 1–10 druhov s rastúcim lvs vykazuje klesanie podielu a naopak viditeľne narastajú triedy s 21–30 druhmi, 31–40 druhmi a mierne aj 40 a viac druhmi. Tento trend končí pri 6. lvs a zvyšné dva lvs majú podobnú, resp. chudobnejšiu druhovú štruktúru bylín ako 1. lvs. Na druhy najbohatšie sú teda šiesty, piaty a štvrtý lvs.

Spolu sa najmenej druhov zachytilo v 8. lvs (48 bylín + 4 trávy), v 7. lvs (73 bylín + 6 tráv) a 1. lvs (194 bylín a 25 tráv). Ostatné lvs sú pomerne vyrovnané s celkový-



Obr. 3 95 %-ný interval spoľahlivosti podielu tried druhových početností bylín a ich porovnanie podľa lvs a integrovaného RS

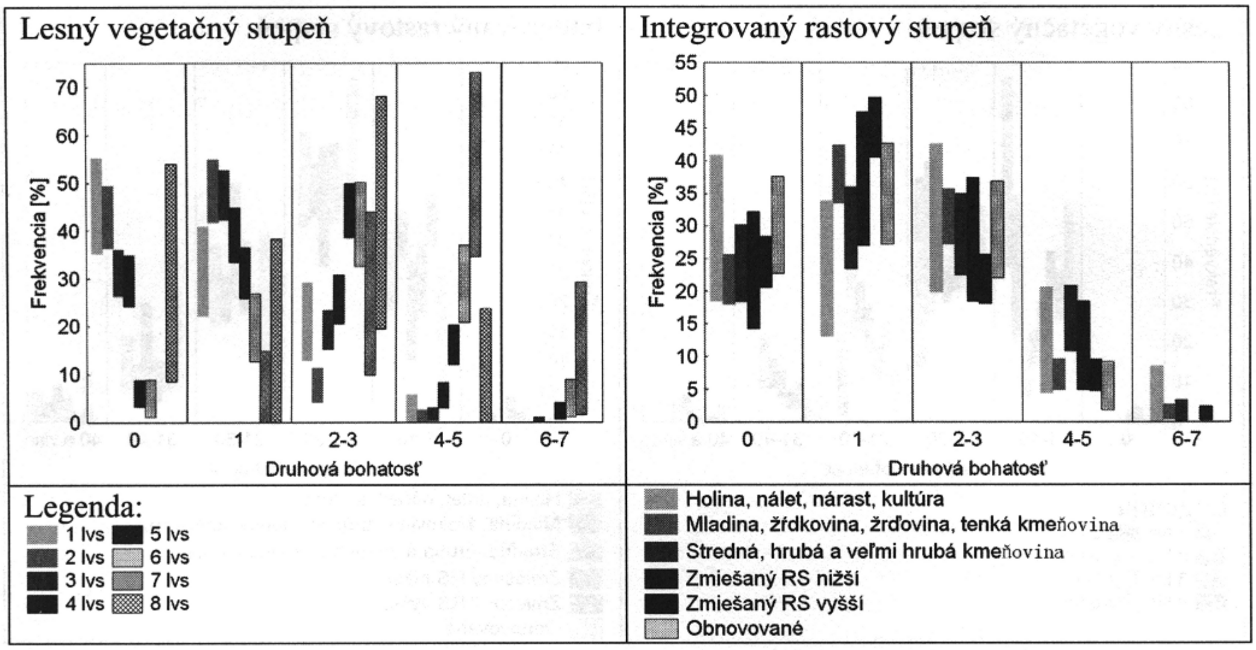
mi počtami bylín od 215 do 298 druhov a tráv od 19 po 36 druhov (pri porovnávaní sumárnych údajov je potrebné vziať v úvahu, že nie sú viazané na rovnakú plošnú výmeru).

Z hľadiska vývoja **(integrovaného rastového stupňa)** porastu sa v celku neprejavili rozdiely (obr. 3). Najzastúpenejšia trieda 11–20 druhov dosahuje asi 40 % podiel bez ohľadu na rastový stupeň (rozdielna je len dĺžka intervalu spôsobená menším či väčším počtom prípadov). Mierne vyššiu bohatosť bylinných druhov pozorujeme pri nižšom zmiešanom RS a mladých porastoch (holina, nálet, nárast, kultúra), triedy 21–30, 31–40 a 40 a viac). Naopak, mierne nižšia bohatosť sa zistila pri mladinách, žŕdkovinách a tenkých kmeňovinách (vysoký podiel triedy 1–10), ale okrem tejto kategórie rozdiel nie je štatisticky signifikantný.

Najvyšší počet druhov sme zaznamenali v 2. agregovanom rastovom stupni (mladina až tenká kmeňovina) s 380 druhmi bylín a 84 druhmi tráv, najmenej v prvom agregovanom rastovom stupni (holina, nálet, nárast, kultúra) s 219 druhmi bylín a 51 druhmi tráv.

### *D) Druhová bohatosť machov v lesných vegetačných stupňoch a integrovaných rastových stupňoch*

Pri **vegetačnej stupňovitosti** možno poukázať na postupný stúpajúci trend v početnosti druhov machov so stúpajúcou nadmorskou výškou (obr. 4). V nižších lvs prevažujú triedy s 0, resp. len jedným druhom machu, kým v siedmom a ôsmom sú to naopak bohatšie triedy so 4–5, resp. 2–3 druhmi. V stredných polohách okolo tretieho až štvrtého lvs je podiel vyrovnanejší, s miernym maximom v triede 2–3



Obr. 4 95 %-ný interval spoľahlivosti podielu tried druhových početností machov a ich porovnanie podľa lvs a integrovaného RS

druhov. Z obrázku 4 je zrejmá vyššia početnosť druhov machov so stúpajúcim lesným vegetačným stupňom s maximom v smrekovom lvs.

V rámci porovnania **početnosti druhov machov podľa rastového stupňa** porastu sa výraznejšie rozdiely neukázali (obr. 4). Najzastúpenejšia trieda všeobecne je 1 druh machu, v náletoch, nárastoch a kultúrach mierne vyšší podiel dosiahla trieda s 2–3 druhmi. Po nich nasleduje opäť vyrovnane pri všetkých rastových stupňoch podiel bez výskytu machov. Pri počte 4–5 druhov sa štatisticky významne líšia žŕďoviny s malým podielom od hrubších kmeňovín.

## 4. Záver

Výsledky inventarizácie potvrdili druhovú rozmanitosť a bohatosť našich lesov. Na druhy najbohatšia skupina vegetácie, trávy a byliny, dosiahla priemerný počet 16 druhov na jednu IP (500 m$^2$). Pri tejto skupine sa prejavil stúpajúci trend druhovej bohatosti vzhľadom na lvs s maximom v 6. lvs. Naopak pri rastovom stupni ani kategórii pozemku sme nezistili výrazné spojitosti k počtu druhov.

Pri stromoch s hrúbkou nad 7 cm sa zistila väčšia druhová bohatosť na nelesných pozemkoch.

Analýza druhovej bohatosti krov priniesla obdobné výsledky ako pri stromoch t. j. zistili sme výraznejšie vyššiu druhovú bohatosť na nelesných pozemkoch.

Druhovo najchudobnejšou skupinou sú machy, pri ktorých sa mierne prejavil vyšší počet druhov na nelesných pozemkoch ako na lesných. So stúpajúcim lesným vegetačným stupňov sa zaznamenal zrejmý trend rastu počtu druhov machov. Počet druhov machov v rôznych integrovaných RS nepreukázal väčšie spojitosti.

Priemerne sa na jednu IP (500 m$^2$) vyskytlo 3,8 druhov stromov, 2,4 druhov krov, 2,7 druhov tráv a 14 druhov bylín a 1,5 druhu machov alebo lišajníkov.

Druhová bohatosť je dôležitou zložkou biodiverzity a výsledky analýz poukazujú na jej priaznivé hodnoty v lesoch Slovenska.

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Eur J Forest Res (2006) 125: 427–439
DOI 10.1007/s10342-006-0138-2

ORIGINAL PAPER

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# Characterisation of diameter distribution using the Weibull function: method of moments

Received: 2 April 2006 / Accepted: 11 May 2006 / Published online: 28 June 2006

**Abstract** The shape of the diameter distribution is one of the important elements characterising forest stand structure. In this work we present the application of the method of moments for the estimation of the parameters of a 2-parameter Weibull function. Due to its properties, this function is often used for the description of the diameter distribution in forestry. The work analyses the properties of the Weibull function and its application to a data set representing natural (virgin forest Babia hora) and managed forests (Litschau and Forest School Enterprise of Technical University Zvolen). The parameters of the Weibull function are simply and reliably estimated from the basic stand variables, namely the mean diameter and the coefficient of variation of the diameters. The method is general and does not require specific parameterisation, e.g. for the individual tree species. The work also presents a new algorithm for the estimation of parameters in cases, where tree diameters are measured from a certain minimum recording limit. Based on this study, we suggest using the Weibull function in forest stands only for uni-modal diameter distribution with a mean diameter above 7 cm.

**Keywords** Weibull function · Diameter distribution · Moments · Complete data · Truncated data

Communicated by Hans Pretzsch

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## Introduction

Forest growth and yield models, such as Silva (Pretzsch 1992), Moses (Hasenauer 1994), Prognaus (Sterba et al. 1995) or Bwin (Nagel 1995), are becoming more and more important as they may form the core of decision support systems. They provide the user with forecasts of forest stand developments that may facilitate management decisions. These individual tree growth models need tree lists as input data, i.e. a list of the trees in a stand or a sample plot with at least the diameter at breast height, dbh, given. The distance dependent simulators Silva and Moses additionally need the spatial coordinates of the trees.

Very frequently, tree lists are not available, but rather some general description of the stand including its site index, density and mean diameter. In such cases, tree list needs to be generated from these measures. Such generators have been developed e.g. by Pretzsch (1993), Pommerening (2000) and several others. The value of these generators depends much on how reliably they can reproduce the dbh-distribution of differently structured stands. Commonly a certain kind of mathematical frequency distribution of breast height diameters is hypothesised, and the reliability of the generator depends on (1) how exactly this mathematical distribution fits the real dbh-distribution, (2) how accurate its parameters can be estimated, and (3) how accurately the frequency of trees within a given diameter interval can be predicted on the basis of summary stand characteristics. Within a stand other tree parameters, such as tree height, tree volume, tree assortment value etc. (Bailey and Dell 1973) and even the spatial distribution, are well correlated with dbh; therefore, an accurate generation of diameter distribution is crucial.

Many attempts have been made to describe diameter distributions mathematically: de Liocourt (1898) proposed a reversed J-shaped model for uneven-aged stands, which has been widely used in forestry (Meyer 1952, Leak 1965). For even-aged stands Schiffel (1904) proposed to use the Fekete-distribution, which depicts the dbh as a third degree polynomial of the cumulative frequency. Other distributions that have been successfully applied to the description of the dbh-distribution are, e.g., the Gram–Charlier series, the Pearl–Reed growth curve, Pearson's curve, the gamma-, beta- and log-normal distributions (Bailey and Dell 1973).

The Weibull function has become very popular for describing diameter distributions (Nagel and Biging 1995; Biging et al. 1994; Gerold 1988; Shiver 1988; Zutter et al. 1986; Shifley and Lentz 1985; Gadow 1984; Little 1983; Bailey and Dell 1973) because of its flexibility and ability to fit various distributions from the reversed J-shaped, through left-skewed and symmetrical distributions up to right-skewed distributions (Nagel and Biging 1995; Bailey and Dell 1973). That is also the reason why the Weibull function has earlier frequently been used to generate diameter distributions in forest growth simulators, e.g. Cactos (Wensel et al. 1986), Twigs (Miner et al. 1988), Silva (Pretzsch 1992), and Bwin (Nagel 1995).

In these applications, the parameters of the Weibull distribution are usually derived from stand characteristics using regression analysis. This technique can, however, cause bias in parameter estimates. Therefore, in this paper we present a more accurate, analytical method of parameter estimation. We will thus describe the estimation of the Weibull parameters using the method of moments, and, based on data collected in virgin and managed forests, show the suitability of this estimation method. In addition, we will consider the specific cases, where the measurement of diameters does not start from zero, but from a pre-defined recording limit (minimum diameter).

## Data

### Virgin forest Babia Hora

Babia Hora, with an elevation up to 1,725 m above sea level, is an isolated mountain massif belonging to the outer Western Carpathian mountain range lying in the northern part of Slovakia at the border to Poland. In 1926, a 118 ha national nature reserve was established to preserve the original forest ecosystems situated far from human settlements. In 1974, the reserve was enlarged and currently it covers an area of 504 ha (Korpel' 1989), where according to the Nature Protection Act no forest management is performed. The reserve encompasses the forest stands of Babia hora at an elevation from 1,100 m above sea level composed almost entirely of Norway spruce (*Picea abies* L. Karst.) with a small admixture of silver fir (*Abies alba* Mill.) and rowan (*Sorbus aucuparia* L.). The significance of these forest ecosystems was confirmed by a study of the genetic structure of spruce, which revealed that the spruce population of Babia hora can be regarded as a gene pool of the original populations of Norway spruce of the Western Carpathians (Gömöry 1988).

In the region of the Babia Hora national nature reserve, 57 permanent circular sample plots were established in 2002 (Merganič et al. 2003), each with an area of 0.05 ha (i.e. radius = 12.62 m). The plots are located at an elevation from 1,100 m above sea level to the timber line (approximately 1,500 m). They are equally distributed to four pre-defined elevation categories (below 1,260, 1,260–1,360, 1,360–1,460 m and above 1,460 m above sea level) and three development stages of virgin forests: stage of growth, maturity and breakdown as defined by Korpel' (1989). On each sample plot, every tree with a diameter at breast height above 7 cm was measured. Trees higher than 1.3 m but with dbh $\leq$ 7 cm were measured on the second concentric circle. Its radius was estimated directly in the field according to Šmelko (1968), who defined the optimum plot size as one that contains 15 to 25 trees. However, the size of this second concentric plot never exceeded the area of the first circle, i.e. its maximum radius was 12.62 m.

### Managed forests in Litschau

In 1977, 22 permanent plots were established in mixed Norway spruce (*P. abies* L. Karst.)—Scots pine (*Pinus sylvestris* L.) stands in Seilern-Aspang's Forest enterprise, in the Austrian part of the Bohemian Massif. The elevation of these stands varies between 400 and 600 m. The predominant soil type is a podsol, sometimes with pseudogley dynamics. The management of this enterprise is presently shifting from a clear-cutting system towards a continuous-cover system with single-tree selection harvests. The plots were established and measured in 5-year intervals in order to parameterise the first distance dependent forest growth simulator for mixed-species stands (Sterba 1982). The stand age ranged from 15 to 110 years, and their dominant height from 5 to 31 m. The proportion of Norway spruce (by basal area) ranged from 27 to 100%. The plot size was between 200 and 1,600 m$^2$ depending on the dominant height at the time of establishment. Most of the plots were heavily damaged by snow breakage in the winter 1979/80. Thinning was performed in 1983, releasing individual trees that belonged to crown class "dominant" and "co-dominant", using Johann's (1982) way to define the neighbours being removed. All the older plots now comprise at least one additional layer of regeneration. At each of the remeasurements of the plots, individual tree co-ordinates, tree species, breast height diameter, tree height and height to the crown base were measured for every tree exceeding a height of 1.3 m; thus there was no diameter recording limit.

### Managed forests of the Technical University Zvolen

The third set of data used in this study originates from the permanent inventory plots (PIP) established in 1977 with the aim of examining the representativity of forest inventory sample plots (Šmelko 1979). The PIP plots are located in forest stands with a structure typical for the forests of the Forest School Enterprise (FSE) of the Technical University Zvolen. This enterprise has a total area of 8,044 ha, out of which 7,744 ha are covered by forests. Within the forested area, 32% is identified as management forest, 7% as protection forest, and the rest as the forests for special purposes (e.g. for recreation, research, hunting). All the PIP plots are situated in the forest stands that are subjected to forest management according to the approved forest management plans, i.e. management operations on these plots are not influenced by research activities. The area of the PIP plots ranges from 3 to 7 ha. For the purpose of this work, 9 permanent inventory plots were used, each of them representing a specific stand structure. Together they cover the whole range of species composition from very diverse stands (PIP 8 and 1) to pure homogenous stands (PIP 3).

All trees on each PIP were numbered and their position within the plot is described by their co-ordinates. For every tree, besides tree species, the following basic biometrical characteristics were determined: diameter at breast height (dbh), tree height, and tree volume. Minimum recording dbh was 2.0 cm (PIP No. 5), 7.0 cm (PIP No. 1, 3, 4 and 7), and 8.0 cm in stands No. 2, 6, 8 and 9.

## Methods

### Weibull function

The Weibull distribution is named after G.W. Weibull, who in 1939 developed a statistical theory of the strength of materials based on the probability distribution proposed by Fischer and Tippett in 1928. The name Weibull distribution was first used by Mann (1967, 1968 cif. Bailey and Dell 1973). The complete three-parameter probability density function (pdf) is defined as follows:

$$f(x) = \frac{c}{b} \cdot \left(\frac{x - \alpha}{b}\right)^{c-1} \cdot \mathrm{e}^{-\left(\frac{x-\alpha}{b}\right)^{c}} \qquad (1)$$

where: $f(x)$ is density of trees with the size $x$ ($0 \leq x$), $\alpha$ is a location parameter, $b$ is a scale parameter, and $c$ is a shape (slope) parameter.

The shape parameter $c$ is a dimensionless number. If $c < 1$, the function has a reversed J shape. In the case when $c = 1$, the function becomes the exponential distribution. For $1 < c < 3.6$, the Weibull density function is mound shaped and left-skewed, whereby when $c$ equals 2, the curve results in the Rayleigh-distribution, which is a special case of the $\chi^2$-distribution. If the $c$ parameter reaches the value 3.6, the coefficient of skewness of the pdf function approaches 0 meaning that the shape of the Weibull function approximates the shape of the normal distribution. For $c > 3.6$ the distribution is right-skewed.

The scale parameter $b$ has the same dimension as the variable $x$, i.e. in the case of the dbh-distribution as the breast height diameter. Its value determines the location of the peak of the pdf curve. Thus, if $c$ is constant an increase in this parameter stretches out the pdf curve and decreases its height. However, the parameter $b$ does not influence the overall shape or behaviour of the distribution.

The location parameter $\alpha$ indicates the minimum value of $x$ and thus, the position of the distribution along the $x$-axis. The value of $\alpha$ does not affect the shape of the pdf function. In diameter distribution functions, this parameter is equal to the minimum tree diameter, i.e. if $\alpha = 0$, the minimum diameter is 0 cm and the distribution starts at $x = 0$.

Applying the condition $\alpha = 0$ to Eq. 1 we obtain the two-parameter Weibull probability density function, which will be used subsequently. It is given by:

$$f(x) = \frac{c}{b} \cdot \left(\frac{x}{b}\right)^{c-1} \cdot \mathrm{e}^{-\left(\frac{x}{b}\right)^{c}} \qquad (2)$$

Consequently, the cumulative distribution function (cdf) is defined as follows:

$$F(x) = 1 - \mathrm{e}^{-\left(\frac{x}{b}\right)^{c}} \qquad (3)$$

Setting $x = b$ leads to the cumulative probability corresponding to the scale parameter $b$, $F(b) \approx 0.63$. Thus, the parameter $b$ can be interpreted as the 63rd percentile of the diameter distribution, i.e. about 63% of trees have the diameter smaller than $b$ (Bailey and Dell 1973).

The inverse function of the Weibull-cdf is given by:

$$x = b \cdot [-\ln(1 - F(x))]^{\frac{1}{c}} \qquad (4)$$

This function is used for generating a tree diameter, whereby $F(x)$ is replaced by a random number from a uniform distribution on the interval (0;1).

In forestry, inventory data are frequently incomplete, i.e. they usually do not include observations below a certain minimum diameter. In such cases we talk about the truncated or censored data on the left end. For these cases, an extra parameter ($T$) needs to be included in the Weibull-function and hence, Eqs. 2, 3, and 4 look as follows (Nagel and Biging 1995):

$$f_T(x) = \frac{c}{b} \cdot \left(\frac{x}{b}\right)^{c-1} \cdot \mathrm{e}^{\left(\frac{T^c - x^c}{b^c}\right)} \qquad (5)$$

$$F_T(x) = 1 - \mathrm{e}^{\left(\frac{T}{b}\right)^{c}} \mathrm{e}^{-\left(\frac{x}{b}\right)^{c}} \qquad (6)$$

$$x_T = b \cdot \left[\left(\frac{T}{b}\right)^{c} - \ln(1 - F_T(x))\right]^{\frac{1}{c}} \qquad (7)$$

### Estimation of Weibull parameters using the method of moments

Analytical methods for estimating the parameters of the Weibull distribution include the least square and maximum likelihood methods, or the estimation from particular percentiles, or—only scarcely—from the moments (Al-Fawzan 2000).

In forestry applications, which use the Weibull function for the description of diameter (or other) distributions, most authors estimate the parameters using the maximum likelihood (Zutter et al. 1986; Gadow 1984; Little 1983; Bailey and Dell 1973) or percentile methods (Gerold 1988). The method of moments is used infrequently and often only when the intention is to compare the applicability of selected analytical methods for the estimation of Weibull-parameters (Kangas and Maltamo 2000; Maltamo et al. 2000; Shiver 1988; Nanang 1998; Ueno and Ôsawa 1987; Zarnoch and Dell 1985; Shifley and Lentz 1985). Few examples which deal with the algorithm used to calculate the Weibull parameters by the method of moments can be found in Burk and Newberry (1984) and Ek et al. (1975).

The method of moments is, due to its simplicity and accuracy of parameter estimation, described as the best method by several authors, e.g. Al-Fawzan (2000), Nanang (1998), Ueno and Ôsawa (1987), Shifley and Lentz (1985). Moreover, this method is convenient as the parameter calculation can be done on the basis of the mean and variance of the dataset. Thus, the needed moments can be estimated from the samples in an unbiased way, whereas the estimation of maximum dbh, which is in some cases used for the estimation of the Weibull parameters (Nagel and Biging 1995), is increasingly biased with decreasing sample size.

$$\text{Kurtosis} = \frac{M(4) - 4\cdot M(1)\cdot M(3) + 6\cdot (M(1))^2\cdot M(2) - 3\cdot (M(1))^4}{\left(\sqrt{M(2) - (M(1))^2}\right)^4} \tag{16}$$

The basic moment equation of the complete two-parameter Weibull function is (Weisstein 2003):

$$M(r) = b^r \cdot \Gamma\left(1 + \frac{r}{c}\right) \tag{8}$$

with: $r$ being the order of the moment $M$, $b$ and $c$ are the parameters of the Weibull function, and $\Gamma$ () is the Gamma function.

The moment equation of the left-truncated function is given by:

$$M(r) = \int_T^\infty x^r \cdot \left(\left\{ 0 \quad \text{if} \quad 0 \leqslant x.\right) = \int_T^\infty x^r \cdot f_T(x)\mathrm{d}x \right. \tag{9}$$

As given in Weisstein (2003), the statistical characteristics of the function, i.e. arithmetic and quadratic mean (AM, QM), standard deviation of arithmetic and quadratic mean (SD_AM, SD_QM), and coefficient of variation of arithmetic and quadratic mean (CV_AM, CV_QM), can be calculated from the moments using the following equations:

$$\text{AM} = M(1) \tag{10}$$

$$\text{QM} = \sqrt{M(2)} \tag{11}$$

$$\text{SD\_AM} = \sqrt{M(2) - (M(1))^2} \tag{12}$$

$$\text{SD\_QM} = \sqrt{M(2) - 2\cdot\sqrt{M(2)}\cdot M(1) + M(2)} \tag{13}$$

$$\text{CV\_AM} = \frac{\text{SD\_AM}}{\text{AM}}\cdot 100\text{or}\text{CV\_QM} = \frac{\text{SD\_QM}}{\text{QM}}\cdot 100 \tag{14}$$

Using the same mathematical principles we derived the moment equations for the other two statistical characteristics, skewness and kurtosis:

$$\text{Skewness} = \frac{M(3) - M(1)\cdot M(2) + 2\cdot (M(1))^3}{\left(\sqrt{M(2) - (M(1))^2}\right)^3} \tag{15}$$

### *Complete Weibull function*

Substituting SD_AM and AM, or SD_QM and QM in Eq. 14 for Eqs. 10 and 11 or 12 and 13, respectively, and the respective moments for Eq. 8 shows, that the coefficient of variation depends only on the parameter $c$:

$$\text{CV\_AM} = \left[\frac{\sqrt{\Gamma\left(\frac{c+2}{c}\right) - \left[\Gamma\left(\frac{c+1}{c}\right)\right]^2}}{\Gamma\left(\frac{c+1}{c}\right)}\right]\cdot 100\text{ or}$$

$$\text{CV\_QM} = \left[\frac{\sqrt{2\cdot\Gamma\left(\frac{c+2}{c}\right) - 2\cdot\sqrt{\Gamma\left(\frac{c+2}{c}\right)}\cdot\Gamma\left(\frac{c+1}{c}\right)}}{\sqrt{\Gamma\left(\frac{c+2}{c}\right)}}\right]\cdot 100 \tag{17}$$

The relationship between the parameter $c$ and the coefficient of variation is depicted in Fig. 1a.

Similarly, simplifying Eqs. 15 and (16) reveals that skewness and kurtosis of the Weibull function are also independent of the parameter $b$.

By solving Eq. 17 iteratively for $c$ until the observed coefficient of variation is achieved, the Weibull parameter $c$ can be found. Once $c$ is estimated, the parameter $b$ can be calculated by rearranging the moment Eq. 8 with $r = 1$ in the case when AM is used, and $r = 2$ if QM is applied:

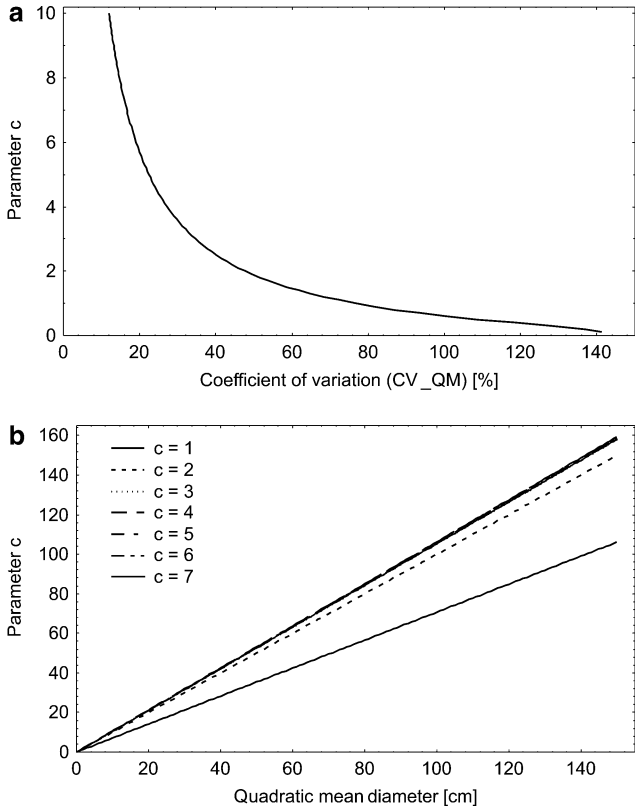

**Fig. 1** Relationships between the parameter *c* and coefficient of variation CV_QM of the complete Weibull function (**a**), and between the parameter *b* and quadratic mean for different values of the parameter *c*(**b**)

$$b = \frac{\mathrm{AM}}{\Gamma\left(\frac{c+1}{c}\right)} \text{ or } b = \frac{\mathrm{QM}}{\sqrt{\Gamma\left(\frac{c+2}{c}\right)}} \qquad (16)$$

Figure 1b shows, how *b* depends on *c* and quadratic mean.

### *Truncated Weibull function*

When data are truncated, it is incorrect to use the complete Weibull function for their description (Zutter et al. 1986). Instead, it is required to use the truncated form of the Weibull function (Eq. 5). The estimation of the parameters of the truncated function is, however, more complicated than for the complete function, as the values of coefficient of variation, skewness and kurtosis depend also on the values of the parameter *b*. An example of the relation between the parameter *b* and the coefficient of variation CV_QM is given in Fig. 2.

The algorithm for the estimation of the parameters of the truncated Weibull function consists of the following steps:

1. Determine the point of truncation *T*. In many cases, *T* is not equal to the lowest measured diameter, but the diameter, from which the measurements were performed. This value should be given either in the manual for the field measurements or in the lists from the field measurements.
2. Calculate the mean and the coefficient of variation from the measured data.
3. Estimate the initial value of the parameter *c* using Eq. 17.
4. Estimate the initial value of the parameter *b* using Eq. 18.
5. Iteratively change *b* and *c* until we obtain the modelled mean and the coefficient of variance calculated using the moment Eq. 9 that approximate the observed mean and coefficient of variation most (Fig. 3). The change of the parameters *b* and *c*, i.e. the iteration steps, can be set by a user with respect to his demands on the accuracy of the estimation. In this study the steps were set to 0.1 and 0.01 for the parameters *b* and *c*, respectively. These values were proved sufficient for practical use.

Although the algorithm for the parameter estimation of the truncated function seems to be rather cumbersome, it is very useful because (1) most forestry data are truncated, and (2) if the estimated values of the parameters *c* and *b* for the truncated Weibull distribution are used in the complete function, the distribution

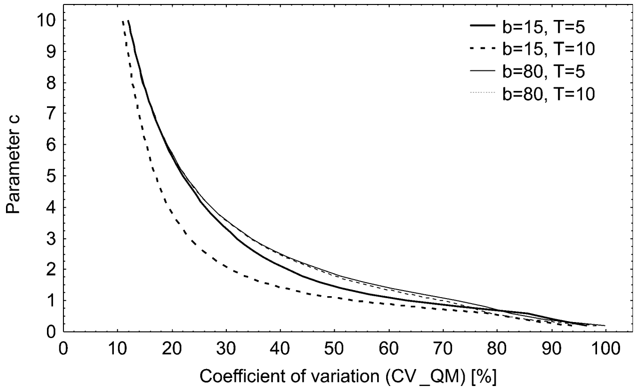

**Fig. 2** Influence of the parameter *b* and the minimum recording limit *T* on the relationship between the parameter *c* and the coefficient of variation CV_QM for the truncated Weibull function

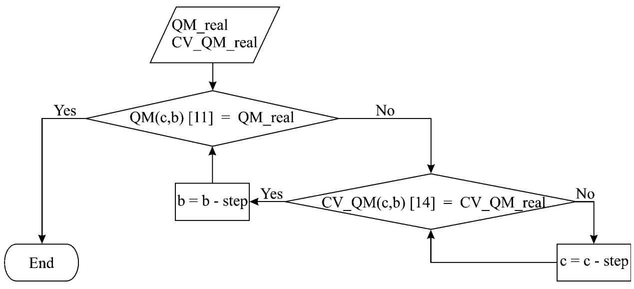

**Fig. 3** Iteration process for estimating the parameters of truncated Weibull function by the method of moments from quadratic mean and its coefficient of variance (Step 5 in the algorithm presented in Truncated Weibull function). The iteration steps, i.e. the change of the parameters *b* and *c*, were set to 0.1 and 0.01, respectively

of the unmeasured data, i.e. the data below the point of truncation, can be estimated.

### Evaluation of Weibull function to describe the diameter distributions of forest stands

The above-stated methodology was applied to observed data in order to test the appropriateness of the Weibull function for the description of diameter distributions. The necessary statistical characteristics, i.e. quadratic mean QM and coefficient of variation CV_QM, were calculated from the tree diameters observed on each plot. These observed statistical characteristics were used to estimate the parameters *c* and *b* from Eqs. 17 and 18, respectively. The calculated parameters of the Weibull function were applied to Eqs. 2 or 5 to obtain the modelled distribution curve. To enable visual comparison of the observed distribution and the Weibull pdf, the observed breast height diameters were divided into diameter classes.

Afterwards, the maximum difference between the cumulated frequencies of the observed and modelled Weibull cdf was tested by the Kolmogorov–Smirnov test at two significance levels 0.05 and 0.01. The modelled Weibull cdf was calculated from Eqs. 3 or 6. To complete the analysis, minimum and maximum diameters were calculated from the modelled inverse Weibull cdf, whereby the probability $F(x)$ or $F_T(x)$ of these values was estimated from the sampling size, i.e. from the number of trees in each specific case. Moreover, the algorithm for estimating the parameters of the Weibull function in the case of the truncated data (Fig. 3) was tested on the data from the managed forests in Litschau.

## Results

### Virgin forest Babia Hora

The analysis was performed on 12 variants of diameter distributions, which were derived by the aggregation of the data from the sample plots situated in the same elevation category and development stage. As shown in Table 1, the modelled quadratic mean QM and coefficient of variation CV_QM are identical with the statistics calculated from the observed data. According to Kolmogorov–Smirnov tests, only 3 distributions out of the 12 analysed variants differed significantly from the modelled Weibull distributions. Two of these belong to the development stage of breakdown, which is characterised by a bimodal distribution. The Weibull function is not able to simulate this kind of distribution. However, this disadvantage could be eliminated when the bimodal distribution is divided into two unimodal distributions representing certain layers of the forest stand, and for each part the parameters of Weibull function may be estimated using the method of moments.

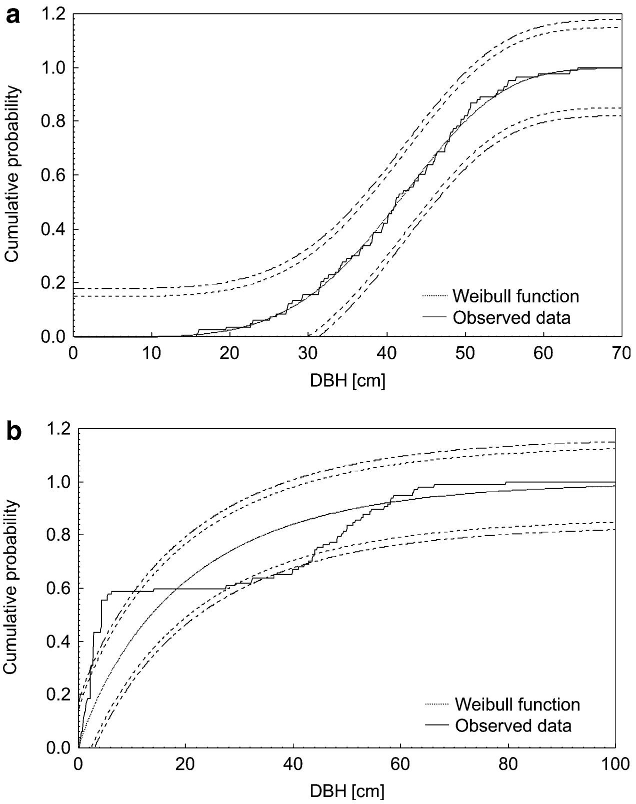

**Fig. 4** Kolmogorov–Smirnov test of the consistence of the observed diameter distribution with the modelled Weibull distribution for the best (**a**) and the worst (**b**) fit of Weibull function, where the dashed line represents the critical value of Kolmogorov-Smirnov test at the 5%, and dashed and dotted line represent the critical value at the 1% level

Comparing the observed minimum and maximum diameters with values calculated from the Weibull function shows that the Weibull function is able to represent the tails of the observed diameter distribution well (Table 1). The differences between the observed and modelled diameters are of course higher in the cases, when the curve of the Weibull function deviates more from the observed diameter distribution. The results of the frequency analysis for the cases, when the values of Kolmogorov-Smirnov test were the lowest and the highest are shown in Fig. 4a, b, respectively.

### Managed forests Litschau

On the data from Litschau, we analysed if the Weibull function is appropriate to describe diameter distributions of management forests. In this case, an analysed variant was represented by an observed diameter distribution of a particular tree species on a certain permanent inventory plot in a particular year.

The number of Norway spruce trees was sufficient on all 23 plots during all remeasurements. Therefore, all 115 diameter distributions could be evaluated for this species. The number of trees per variant varied between 27

**Table 1** Mathematical–statistical description of the observed and modelled diameter distribution and statistical test of the differences between them

| Stage elevation categories | Growth | | | | Maturity | | | | Breakdown | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | below 1,260 m | 1,260 to 1,360 m | 1,360 to 1,460 m | above 1,460 m | below 1,260 m | 1,260 to 1,360 m | 1,360 to 1,460 m | above 1,460 m | below 1,260 m | 1,260 to 1,360 m | 1,360 to 1,460 m | above 1,460 m |
| Observed data | | | | | | | | | | | | |
| AM | 16.09 | 12.93 | 17.81 | 8.11 | 46.62 | 46.12 | 41.00 | 23.17 | 21.42 | 26.64 | 37.64 | 21.07 |
| QM | 23.32 | 17.57 | 22.85 | 11.95 | 50.22 | 49.00 | 42.26 | 25.61 | 31.93 | 35.53 | 39.96 | 24.33 |
| SD_AM | 16.92 | 11.92 | 14.34 | 8.79 | 18.80 | 16.67 | 10.33 | 10.94 | 23.81 | 23.66 | 13.58 | 12.22 |
| SD_QM | 18.40 | 12.80 | 15.20 | 9.60 | 19.14 | 16.92 | 10.41 | 11.21 | 26.05 | 25.30 | 13.78 | 12.65 |
| CV_AM | 105.14 | 92.22 | 80.51 | 108.39 | 40.32 | 36.14 | 25.19 | 47.21 | 111.17 | 88.82 | 36.08 | 57.98 |
| CV_QM | 78.91 | 72.82 | 66.55 | 80.29 | 38.12 | 34.52 | 24.62 | 43.78 | 81.58 | 71.21 | 34.49 | 51.98 |
| Skewness | 1.91 | 2.02 | 1.41 | 1.83 | – 0.44 | – 0.01 | – 0.23 | 0.42 | 0.63 | 0.31 | 0.62 | 0.81 |
| Kurtosis | 5.88 | 7.56 | 5.01 | 5.97 | 3.47 | 2.97 | 2.76 | 3.53 | 1.72 | 1.64 | 4.94 | 3.26 |
| $D_{\mathrm{MIN}}$ | 0.50 | 0.50 | 0.90 | 0.30 | 0.30 | 7.40 | 15.80 | 1.40 | 0.30 | 0.60 | 2.20 | 1.20 |
| $D_{\mathrm{MAX}}$ | 81.00 | 65.30 | 73.30 | 44.70 | 100.00 | 89.00 | 64.50 | 62.60 | 79.60 | 76.50 | 82.00 | 58.60 |
| $N$ | 262 | 350 | 193 | 358 | 78 | 73 | 83 | 115 | 97 | 75 | 45 | 127 |
| Weibull function | | | | | | | | | | | | |
| Parameter $b$ | 15.695 | 13.319 | 19.092 | 7.801 | 52.388 | 51.590 | 44.906 | 26.140 | 20.219 | 27.674 | 42.076 | 23.654 |
| Parameter $c$ | 0.950 | 1.084 | 1.248 | 0.923 | 2.669 | 3.017 | 4.502 | 2.238 | 0.898 | 1.124 | 3.021 | 1.781 |
| AM | 16.06 | 12.91 | 17.79 | 8.10 | 46.57 | 46.08 | 40.98 | 23.15 | 21.31 | 26.52 | 37.58 | 21.05 |
| QM | 23.32 | 17.57 | 22.85 | 11.95 | 50.22 | 49.00 | 42.26 | 25.61 | 31.93 | 35.53 | 39.96 | 24.33 |
| SD_AM | 16.91 | 11.92 | 14.34 | 8.79 | 18.79 | 16.66 | 10.33 | 10.94 | 23.78 | 23.64 | 13.58 | 12.21 |
| SD_QM | 18.40 | 12.80 | 15.20 | 9.60 | 19.14 | 16.92 | 10.41 | 11.21 | 26.05 | 25.30 | 13.78 | 12.65 |
| CV_AM | 105.28 | 92.30 | 80.60 | 108.51 | 40.36 | 36.16 | 25.20 | 47.24 | 111.62 | 89.14 | 36.12 | 58.03 |
| CV_QM | 78.91 | 72.82 | 66.55 | 80.29 | 38.12 | 34.52 | 24.62 | 43.78 | 81.58 | 71.21 | 34.49 | 51.98 |
| Skewness | 2.16 | 1.77 | 1.43 | 2.26 | 0.29 | 0.16 | – 0.18 | 0.49 | 2.35 | 1.68 | 0.16 | 0.79 |
| Kurtosis | 10.11 | 7.57 | 5.82 | 10.85 | 2.80 | 2.73 | 2.81 | 3.01 | 11.61 | 7.05 | 2.73 | 3.59 |
| $D_{\mathrm{MIN}}$ | 0.04 | 0.06 | 0.28 | 0.01 | 10.27 | 12.47 | 16.85 | 3.14 | 0.12 | 0.60 | 11.98 | 1.56 |
| $D_{\mathrm{MAX}}$ | 95.63 | 67.99 | 72.21 | 53.24 | 90.93 | 83.60 | 62.46 | 52.41 | 110.04 | 101.69 | 65.50 | 57.35 |
| Difference and Kolmogorov–Smirnov test | | | | | | | | | | | | |
| $\Delta D_{\mathrm{MIN}}$ | – 0.46 | – 0.44 | – 0.62 | – 0.29 | 9.97 | 5.07 | 1.05 | 1.74 | – 0.18 | 0.00 | 9.78 | 0.36 |
| $\Delta D_{\mathrm{MAX}}$ | 14.63 | 2.69 | – 1.09 | 8.54 | – 9.07 | – 5.40 | – 2.04 | – 10.19 | 30.44 | 25.19 | – 16.50 | – 1.25 |
| K-St | 0.127** | 0.071 | 0.048 | 0.071 | 0.126 | 0.066 | 0.043 | 0.070 | 0.332** | 0.214** | 0.106 | 0.042 |

*AM* is the arithmetic mean diameter (cm), *QM* quadratic mean diameter (cm), *SD* standard deviation, *CV* coefficient of variation (%), $D_{\mathrm{MIN}}$ minimum diameter at breast height (cm), $D_{\mathrm{MAX}}$ maximum diameter at breast height (cm), *N* number of observations, Δ difference and *K-St* is the value of the Kolmogorov–Smirnov statistics
Significance level: *95%, **99%

and 1,093, whereby in 30.4% of the distributions the number was less than 50. The difference between the observed distribution and the distribution modelled with the Weibull function was found to be statistically significant at the 1% level in 21 variants, i.e. in 18.2% cases from the total 115 analysed variants (Table 2). Out of the significant cases, 12 variants, i.e. more than half, could be characterised as bimodal distributions suggesting that in these forests Norway spruce occurred in more than one layer.

The extent of the dataset for this tree species allowed us to perform also other analyses, which could explain the significant differences between the observed and modelled diameter distributions. In the first analysis, we tested the differences in mean diameter between the three artificial groups defined according to the results of Kolmogorov–Smirnov test: insignificant result, significant at the 5% level, and significant at the 1% level. Note that statistically significant variants with evident bimodal diameter distribution were excluded from this analysis.

This analysis revealed that the mean diameter of the forest stand strongly influences the shape of the diameter distribution. Young forest stands, characterised by a small quadratic mean diameter QM cannot be described by the Weibull function properly. According to our result, the Weibull function is with 99% probability not suitable for the description of diameter distributions of the forest stands with quadratic mean diameters below 6–7 cm.

Nevertheless, the interpretation of this result must be done with caution. The result of the test is influenced by the size of the dataset, in our case by the number of trees, from which the frequency analysis was performed. Therefore, although statistically significant differences between the observed and modelled distributions were detected mostly in the forest stands with a small mean diameter, these stands are composed of a larger number of trees which on one side decreases the error, but on the other hand causes that small differences become significant.

The results of this analysis fit well with the time trend in the development of the sample plots, since statistically significant differences between the observed and the modelled diameter distributions usually occurred at the beginning of the experiment (young forests or old forests stands with a high proportion of upper layer), or at the end of the experiment, when due to the applied singletree harvesting management vertical stand structure starts to differentiate.

Similarly, we analysed the differences between the three groups with regard to the coefficient of variation of diameters. We found that statistically significant differences between the observed and modelled diameter distributions can be detected if the observed variability of tree diameters in a stand is high, typically above 50%.

The analysed plots consisted of more tree species. However, due to the small number of trees of a particular species on a plot it was not possible to analyse all possible variants. Therefore, we selected only those variants, which included more than 30 individuals of each species. Thus, we obtained 30 variants representing Scots pine, common beech (*Fagus sylvatica* L.) and other broadleaves. Just as for Norway spruce, the estimation of the Weibull function using the method of moments was in most cases successful. Only four modelled diameter distributions were significantly (at the 1% level) different from the observed distributions (Table 2), whereby all significant variants are characterised by bimodal distribution.

The algorithm for the estimation of the parameters of the Weibull function for the truncated data (Truncated Weibull function) was applied to 93 diameter distributions with QM above 10 cm and to a minimum number of trees 30 individuals. The goal of this analysis was to document that the proposed method of the parameters estimation is correct. From the truncated data only the parameters of the truncated Weibull function should be estimated. The obtained values should coincide with the parameters of the complete Weibull function estimated from the complete data. If, however, truncated data are used to estimate the parameters of the complete Weibull function, the estimated values of the parameters contain a systematic error causing a significant bias in the function statistics. Its magnitude depends on the value of $T$, the data integrity below $T$, and the distance between

**Table 2** Results of Kolmogorov–Smirnov test about the identity of the observed and modelled diameter distribution separately for Norway spruce, Scots pine, common beech and other broadleaved species

| Tree species | Significance level | | | Total |
|---|---|---|---|---|
| | Not significant | Significant at 5% | Significant at 1% | |
| | Number of diameter distributions/relative proportion from the total number per species (%) | | | |
| Norway spruce | 86<br>74.78 | 8<br>6.96 | 21<br>18.26 | 115 |
| Scots pine | 19<br>82.61 | 2<br>8.70 | 2<br>8.70 | 23 |
| Common beech | 3<br>100.00 | 0<br>0.00 | 0<br>0.00 | 3 |
| Other broadleaves | 1<br>33.33 | 0<br>0.00 | 2<br>66.67 | 3 |
| Total | 109<br>75.69 | 10<br>6.94 | 25<br>17.37 | 144<br>100 |

QM and $T$. In younger and middle old forest stands the bias can reach 1.5 to 3 cm, in the cases where diameters were measured from the border characterising commercial timber.

Hence, within this analysis we tested the coincidence of the estimated parameters of the truncated and the complete Weibull function. For this aim, the minimum recording limit $T$ was set to 7 cm meaning that from each distribution trees with diameters below 7 cm were excluded. In the next step, the algorithmus for the estimation of the parameters of the truncated Weibull function was applied to these truncated data. The results of the analysis were strongly influenced by the original shape of the distribution function, i.e. if the observed complete distribution was significantly different from the complete modelled Weibull function, the values of the parameters estimated for the truncated Weibull function were also significantly different from the complete modelled Weibull function. This happened mainly for bimodal distributions, which were observed in 22 cases out of the total 93 distributions. After excluding these cases we found that from the remaining 71 distributions in 59 cases (83%), the estimated parameters of the truncated Weibull function deviated only slightly from their original values calculated from the complete data. In these cases, the differences of the parameters $b$ and $c$ between the complete and the truncated functions caused changes of QM and CV_QM of less than 0.5 cm and 5%, respectively. In the remaining cases the truncation of the data caused a more significant change to the shape of the distribution or to the bias in the estimation due to the small number of trees that characterised the diameter distribution.

In the following, we examined how the minimum recording limit $T$ influences the value of quadratic mean QM. The analysis was performed for three $T$ values: 3, 5, and 7, with 71 distributions, out of which 63 represented Norway spruce and 8 Scots pine. The influence of the $T$ value on the average difference was examined with the analysis of variance for the following variants, while the null hypothesis was that the average difference between the modelled and the real value of QM is equal to 0 if:

1. The difference is calculated from the data within the interval $T \rightarrow \infty$,
2. The difference between the modelled and the real value of QM is calculated from the data within the interval $0 \rightarrow T$. Here, we analysed two cases distinguished on the basis of the information about the occurrence of trees below diameter $T$ (which is of Boolean value, 0—they do not occur, 1—they occur):

(a) First, we analysed all the distributions regardless of the information about the occurrence of trees below diameter $T$;
(b) Second, the observed distributions with no trees below diameter $T$ were excluded, i.e. only filled distributions were analysed (those where the Boolean value was equal to 1).

3. The difference between the modelled and the real value of QM is calculated from the data within the interval $0 \rightarrow \infty$, in the cases (a) and (b) as defined earlier.

In all three cases, the modelled QM was determined from the parameters of the truncated Weibull function for the particular interval using Eq. (11).

The results of the first tested variant revealed that $T$ values do not have a significant influence on the values of QM. Positive values of average differences indicate that the model gives systematically higher QM than the observed QM. However, the absolute value of the difference is very small (below 0.035), which in forestry applications is negligible. This bias can be eliminated by decreasing the convergence factor in the algorithm (Truncated Weibull function). Tree species seems to have an effect on the value of average difference, too, while the bias of light demanding species (Scots pine) was only two-third of the bias of shade bearers (Norway spruce). However, this result is related to the distance between $T$ and QM. As the truncation limit $T$ approaches the value of QM, estimating the parameters of the Weibull function becomes more difficult and a smaller step of convergence factor is required. In the analysed data, Scots pine has higher QM values, i.e. its diameter distributions are moved to the right, whereas Norway spruce represents thinner stands, where the bias is greater.

The analysis of the variant 2(a) revealed that $T$ value has a significant effect on the values of modelled QM within the interval $0 \rightarrow T$, and that the average differences are positively biased. Similarly to the first variant, the tree species was found to influence model bias significantly. While for Norway spruce the bias magnitude was approximately 2 cm regardless of the $T$ value, the average differences of Scots pine were close to the values of the truncation limit $T$, and increased with increasing $T$. This is caused by higher QM values for the whole data range (i.e. $0 \rightarrow \infty$) and by the absence of trees with diameter below $T$.

However, if only filled diameter distributions within the interval $0 \rightarrow T$ were analysed, i.e. the cases where the observed QM was greater than 0 (variant 2b), $T$ value did not have a significant influence on the average difference between the observed and modelled QM. Although the absolute values of average differences were always below zero indicating that the model underestimates QM, they were not significantly different from 0. We assume that reducing the convergence factor would result in a more accurate result.

The test of the third variant documents that the proposed method for the estimation of the parameters of the Weibull function for the truncated data is correct and well-founded. The average difference between the modelled and observed QM was in all tested cases very close to zero. The significant average difference was found only for Norway spruce and $T = 7$ in both analysed cases 3(a) and 3(b) due to the small distance

**Table 3** Mathematical–statistical description of the observed and modelled diameter distribution and the statistical test of the differences between them for the selected managed forests SFE TU Zvolen

| Tree species | Forest stands | Minimum recording limit *T* | Number of trees | Observed data | | | | Weibull function | | | | | | Difference | | Kolmogorov–Smirnov test |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | QM | CV_QM | $D_{MIN}$ | $D_{MAX}$ | Parameter *b* | Parameter *c* | QM | CV_QM | $D_{MIN}$ | $D_{MAX}$ | $\Delta D_{MIN}$ | $\Delta D_{MAX}$ | |
| 1 | 6 | 8.0 | 2151 | 29.79 | 24.76 | 8.90 | 60.50 | 31.62 | 4.42 | 29.79 | 24.77 | 8.34 | 50.16 | − 0.56 | − 10.34 | 0.043** |
| 2 | 1 | 7.0 | 120 | 18.17 | 36.55 | 9.30 | 41.10 | 17.47 | 2.23 | 18.17 | 36.58 | 7.20 | 35.65 | − 2.10 | − 5.45 | 0.108 |
| 2 | 3 | 7.0 | 184 | 20.81 | 54.13 | 8.00 | 62.90 | 10.54 | 0.98 | 20.83 | 54.16 | 7.06 | 63.81 | − 0.94 | 0.91 | 0.090 |
| 2 | 6 | 8.0 | 1105 | 28.32 | 24.82 | 12.60 | 54.00 | 30.05 | 4.39 | 28.32 | 24.82 | 8.50 | 46.81 | − 4.10 | − 7.19 | 0.065** |
| 2 | 2 | 8.0 | 261 | 30.43 | 39.74 | 9.00 | 59.20 | 30.47 | 2.28 | 30.43 | 39.80 | 8.28 | 64.84 | − 0.72 | 5.64 | 0.060 |
| 20 | 4 | 7.0 | 4850 | 18.80 | 30.22 | 7.10 | 39.80 | 19.50 | 3.18 | 18.80 | 30.24 | 7.01 | 38.24 | − 0.09 | − 1.56 | 0.031** |
| 20 | 3 | 7.0 | 871 | 24.80 | 34.30 | 7.90 | 88.30 | 25.69 | 2.83 | 24.80 | 34.35 | 7.11 | 50.55 | − 0.79 | − 37.75 | 0.036 |
| 20 | 8 | 8.0 | 1475 | 26.22 | 24.68 | 8.20 | 52.00 | 27.81 | 4.40 | 26.22 | 24.68 | 8.28 | 43.70 | 0.08 | − 8.30 | 0.025 |
| 20 | 7 | 7.0 | 1769 | 26.37 | 26.08 | 8.20 | 57.50 | 27.98 | 4.15 | 26.37 | 26.08 | 7.28 | 45.46 | − 0.92 | − 12.04 | 0.034* |
| 20 | 9 | 8.0 | 1178 | 32.37 | 23.95 | 8.30 | 65.00 | 34.38 | 4.61 | 32.38 | 23.96 | 8.98 | 52.53 | 0.68 | − 12.47 | 0.032 |
| 20 | 1 | 7.0 | 692 | 33.67 | 21.08 | 11.90 | 57.70 | 35.71 | 5.37 | 33.67 | 21.08 | 10.77 | 50.66 | − 1.13 | − 7.04 | 0.069** |
| 21 | 3 | 7.0 | 943 | 24.12 | 49.32 | 6.80 | 78.00 | 19.48 | 1.44 | 24.11 | 49.39 | 7.02 | 75.56 | 0.22 | − 2.44 | 0.027 |
| 21 | 1 | 7.0 | 1001 | 29.65 | 36.38 | 7.40 | 64.10 | 30.66 | 2.68 | 29.65 | 36.42 | 7.13 | 63.13 | − 0.27 | − 0.97 | 0.042 |
| 22 | 3 | 7.0 | 136 | 13.92 | 41.19 | 7.00 | 40.00 | 6.36 | 1.06 | 13.93 | 41.21 | 7.04 | 34.37 | 0.04 | − 5.63 | 0.097 |
| 22 | 8 | 8.0 | 3413 | 15.02 | 30.09 | 8.10 | 34.10 | 13.71 | 2.34 | 15.03 | 30.08 | 8.00 | 34.14 | − 0.10 | 0.04 | 0.021 |
| 22 | 7 | 7.0 | 3832 | 15.47 | 31.97 | 7.70 | 72.80 | 15.11 | 2.55 | 15.47 | 31.97 | 7.01 | 34.77 | − 0.69 | − 38.03 | 0.055** |
| 22 | 2 | 8.0 | 731 | 17.67 | 38.38 | 2.00 | 55.80 | 14.23 | 1.68 | 17.68 | 38.39 | 8.02 | 45.29 | 6.02 | − 10.51 | 0.071** |
| 22 | 1 | 7.0 | 147 | 18.06 | 31.43 | 9.50 | 39.30 | 18.49 | 2.93 | 18.07 | 31.47 | 7.27 | 32.13 | − 2.23 | − 7.17 | 0.102 |
| 22 | 9 | 8.0 | 2894 | 18.26 | 33.79 | 8.00 | 46.10 | 17.49 | 2.35 | 18.27 | 33.82 | 8.01 | 42.71 | 0.01 | − 3.39 | 0.038** |
| 50 | 8 | 8.0 | 194 | 22.14 | 33.49 | 8.30 | 41.30 | 22.51 | 2.72 | 22.14 | 33.53 | 8.25 | 41.62 | − 0.05 | 0.32 | 0.048 |
| 50 | 7 | 7.0 | 240 | 22.94 | 33.19 | 8.30 | 41.80 | 23.79 | 2.93 | 22.94 | 33.22 | 7.34 | 42.63 | − 0.96 | 0.83 | 0.033 |
| 50 | 9 | 8.0 | 175 | 28.92 | 33.72 | 8.70 | 55.10 | 30.08 | 2.92 | 28.92 | 33.77 | 8.69 | 52.90 | − 0.01 | − 2.20 | 0.071 |

Tree species: 1-Spruce, 2-Fir, 20-Oak, 21-Beech, 22-Hornbeam, 50-Cherry

*QM* is the quadratic mean diameter (cm), *CV* coefficient of variation (%), $D_{MIN}$ minimum diameter at breast height (cm), $D_{MAX}$ maximum diameter at breast height (cm) and Δ the difference

Significance level: *95%, **99%

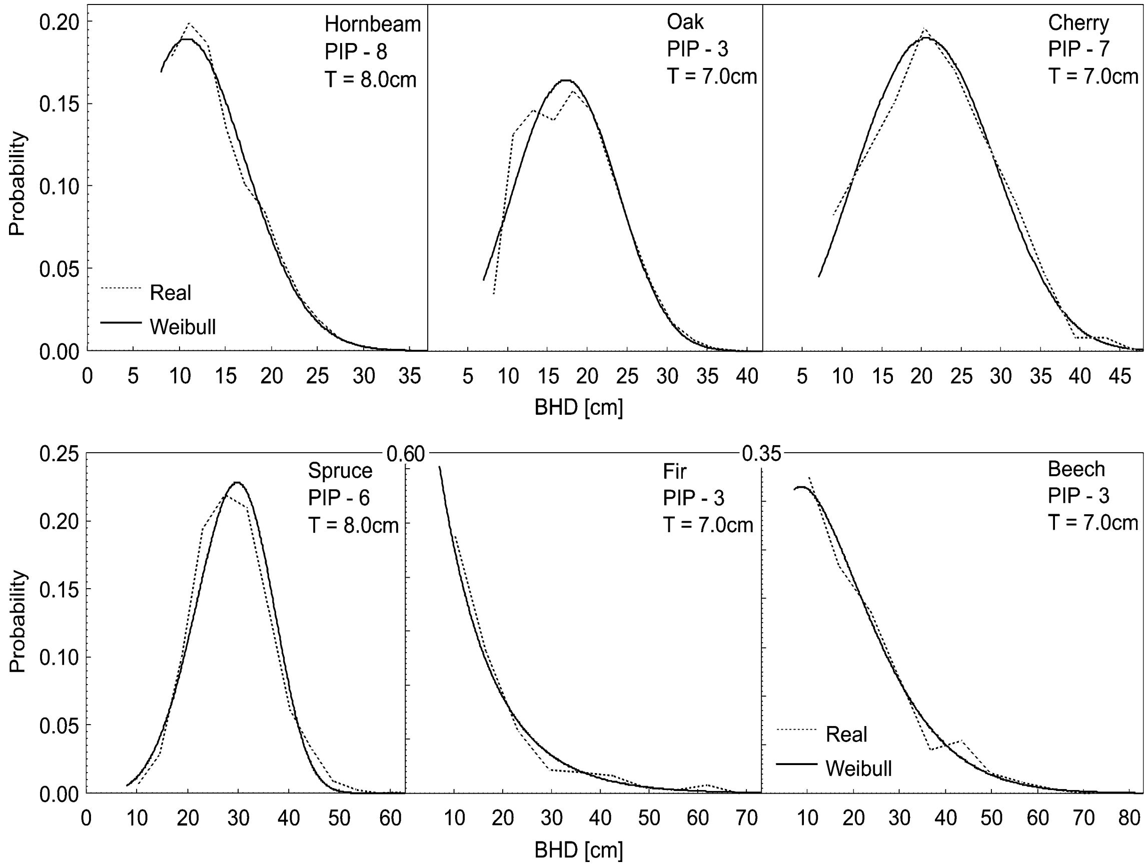

**Fig. 5** Example of the estimation of six different diameter distributions for six tree species that occurred within the selected managed forests SFE TU Zvolen

between $T$ and QM and the convergence factor. However, in forestry applications such values of average differences are negligible.

### Managed forests SFE TU Zvolen

For a sample of selected forest stands (PIP) we applied the method of moments to the truncated data. The point of truncation was $T = 7.0$ cm for forest stands No. 1, 3, 4 and 7, $T = 8.0$ cm in stands No. 2, 6, 8 and 9, and $T = 2.0$ cm for stand No. 5. From the collected data 22 different diameter distributions were created, whereby each variant described a single tree species. Since the forest stands are of a larger size, the diameter distributions are based on a relatively large sample of 120 to 4,850 trees. It emerged that the differences between the observed and modelled distributions were often significant because of the large number of trees (8 cases out of 22 tested variants), although the Kolmogorov–Smirnov values were low (Table 3). From the results we also note that the differences between the observed and the modelled values of $D_{\mathrm{MIN}}$ and $D_{\mathrm{MAX}}$ decrease with an increasing number of trees. Figure 5 documents the flexibility of the Weibull model for describing various shapes of distributions on an example of six different diameter distributions. Overall, the estimation of diameter distribution using the Weibull function seems to work well also in this case.

## Discussion and conclusions

The literature review confirmed that the method of moments is one of the most accurate methods for estimating the Weibull function parameters (Al-Fawzan 2000; Nanang 1998; Ueno and Ōsawa 1987; Shifley and Lentz 1985). Moreover, as it was shown in this work as well as in Shifley and Lentz (1985), the advantage of the method of moments is its minimal data requirement. To create the diameter distribution it is sufficient to know the mean tree diameter and the coefficient of variation of the diameters. Our analyses also showed that the presented procedure yields distributions with summary statitistics that are in agreement with the observed ones (Tables 1, 3). Therefore, it should not happen that the actual mean diameter that enters the model as an input variable will differ significantly from the mean diameter calculated from generated data. Other techniques of parameters estimation, e.g. regression methods that are based on the relation between the parameters of the Weibull function and selected stand characteristics (most frequently mean diameter, maximum or minimum diameter, mean height, etc.), can lead to biased estimates, since the relationships of these variables mainly with the parameter $c$ are very loose (Nagel and Biging 1995; Biging et al. 1994; Little 1983; Van Laar and Mosandl 1989; Clutter and Belcher 1978).

In addition, using the method of moments makes it easier to parameterise the distribution function. Taking into account the statistical character of the observed variables, we have to be conscious of the fact that the real values of the mean diameter and the coefficient of variation are only estimated, since they are calculated from the samples. Nevertheless these estimates are always better than those of a maximum dbh. Data representing the whole population are seldom available. Therefore, the observed values are usually affected by sampling error. In some cases, where the data are somehow distorted or where the sampling estimate is not sufficiently consistent, the Weibull function will not behave logically. For example, during the simulation of diameters we can obtain illogical (very high) values. If we use a very small probability $F(x)$ or $F_T(x)$, that corresponds the simulation of 10,000,000 trees, in Eqs. 4 or 7, the modelled maximum diameter should not exceed the double of the observed maximum diameter. If the result from Eqs. 4 or 7 is outside the predefined range, such value should be excluded and a new value should be generated. However, this happens very rarely and does not affect statistical characteristics of the function. An advantage of this method over other estimation techniques is that it is universal and does not require special parameterisations, e.g. for individual tree species (Table 3; Fig. 5). For example, Nagel and Biging (1995), who used the regression method, estimated the parameters of the Weibull function for each tree species separately. The work also presents a new algorithm for parameters estimation in cases, where tree diameters are measured from a certain recording limit. As Zutter et al. (1986) pointed out, the application of the complete Weibull distribution to describe truncated data can cause great systematic errors in the parameters. This bias is increasing if the value of QM is close to the pre-defined recording limit.

The Weibull function is not suitable for the description of diameter distribution in young forest stands. A similar finding was reported by Nanang (1998), who analysed the suitability of several different types of distributions for the description of diameter distributions. The author found that for the definition of the diameter distribution in young forests a log-normal distribution is more appropriate. Based on our experience as well as on the results of Nanang (1998) we suggest using the Weibull function only when the mean diameter is greater than 7 cm.

The information obtained can be applied with great advantage to modelling of the forest stand structure. Among the existing individual-tree forest growth simulators, the only one using the method of moments is according to our knowledge the growth simulator TWIGS (Miner et al. 1988) in its sub-model TREEGEN. This generator, however, utilises the three-parameter Weibull function and does not allow modelling diameter distributions if tree diameters are measured from a certain pre-defined recording limit. From our point of view, the two-parameter Weibull function is more suitable than the three-parameter function because logically the function should start from 0. This assumption should be valid also in very old forest stands. In such cases, although the function begins from 0, it is very improbable that during the diameter simulation we get a diameter close to 0.

**Acknowledgements** This work was carried out during the research stay of J. Merganič at the BOKU University of Natural Resources and Applied Life Sciences in Vienna, financed by ÖAD during 2002–2003. We are very grateful to Prof. Šmelko for providing us with the data from permanent inventory plots of FSE TU Zvolen for the presented analyses. We also thank Robert Williams and two anonymous reviewers for their comments, which helped us to improve the quality of the presented paper.

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JOURNAL OF FOREST SCIENCE, *52*, 2006 (9): 399–409

# Structural diversity change and regeneration processes of the Norway spruce natural forest in Babia hora NNR in relation to altitude

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**ABSTRACT**: The research was focused on exploring the dynamics and forms of regeneration processes and structural diversity of the Norway spruce virgin forest in Babia hora NNR in relation to altitude. In each developmental stage 19 sample plots were established. The structural diversity was assessed by the indices proposed by Clark and Evans (1954), Füldner (1995) and Jaehne and Dohrenbusch (1997). Concerning the spatial distribution of the trees in the virgin forest no tendency to their clustering with the increasing altitude was found in the zones below 1,460 m. The influence of the altitude was confirmed in the zone above 1,461 m where the groups of "family spruces" are typical. Diameter differentiation was significantly higher in the growth stage. Evaluation of this attribute in terms of the altitude detected significant diameter differentiation at an altitude below 1,260 m in the growth stage. According to the Füldner index the virgin forest has generally medium differentiated diameter structure. According to the complex index by Jaehne and Dohrenbusch (1997) the differentiation of the virgin forest decreases with the ascending altitude up to 1,460 m, where the compact forest ends. Evaluation of the seedbed revealed that 46.2% of the naturally regenerated individuals were growing on mineral soil, 52.4% on dead wood and 1.4% on wind-thrown roots. Regarding the developmental stages 46% of the individuals were found in growth stage, 23% in optimum stage and 31% in breakdown stage.

**Keywords**: Norway spruce; virgin forest; structural diversity; natural regeneration

Natural Norway spruce forests in the 7th spruce forest vegetation zone serve as a laboratory for the observation of their structure and regeneration processes under changing ecological conditions. Mountain forests in the orographic unit Babia hora represent the remains of natural forests on the flysch bedrock. A major part of the virgin forests in this massif has become the object of legislative protection in the form of national nature reserve (NNR). According to Korpeľ (1989) the knowledge of the basic structural attributes of Norway spruce natural forests allows us to identify their stage with the maximum functional efficiency. In the process of investigating these issues the structural diversity of these forest ecosystems is of great importance. Its quantification and understanding can significantly contribute to the explanation of the complicated ecological problems such as calamities. The quantification of the diversity through the mathematic formulas allows us to evaluate this problem objectively and to better understand the relations of a given forest ecosystem. A remarkable part of their diversity is structural diversity, which according to some authors is defined as specific arrangement of the components in the system (Gadow 1999) or as their positioning and mutual connections (Heupler 1982 in Lübbers 1999). According to Zenner (1999) the structure can be characterized horizontally, i.e. the spatial distribution of the trees, and vertically in their height differentiation. Lübbers (1999) added the amount and form of dead wood to these attributes. Gadow and Hui (1999) defined the structure as spatial distribution, mutual position and height differentiation of the trees in a forest ecosystem. For the mathematic description and quantification of this issue the index methods (indices) are mostly used. The most famous for the description of the horizontal distribution of trees on the area of a stand is the index by CLARK and EVANS (1954), based on the relation between the real distance to the nearest neighbour and the expected distance in the Poisson forest, i.e. in a forest where the spatial distribution of the trees is random (TOMPPO 1986).

Concerning different approaches, there are some indices that describe the differentiation of diameter, height and volume, respectively (FÜLDNER 1995) or complex indices describing more components of the structural diversity (PRETZSCH 1996, 1988; JAEHNE, DOHRENBUSCH 1997; ZENNER 1999; LÄHDE et al. 1999). By their help it is possible to add another hierarchical level of the stand diversity.

Regeneration processes in a Norway spruce natural forest are closely connected with the developmental stage and seedbed type. According to KORPEĽ (1989), and VORČÁK (2005) the amount of natural regeneration growing on dead wood increases with the altitude and most of it appears in the initial phase of the growth stage.

Analyzing the regeneration processes in the National Park Bavarian Forest REIF and PRZYBILLA (1998) found that besides the dead wood as a seedbed light has an important influence on the Norway spruce seedling density. This was already been previously by ZIERL (1972). The importance of the dead wood as a seedbed increases with the altitude of the Norway spruce natural forest. At the altitude above 1,400 m the dead wood represents the basic form of the seedbed (MAI 1999).

In Babia hora NNR JALOVIAR (2006) discovered that the ratio between the Norway spruce seedling weight and their root system weight is twice higher on the dead wood than on the mineral soil. HOLEKSA (1998), who investigated the Polish part of the orographic unit Babia hora, found that the competition and presence of high plants especially the fern *Athyrium distentifolium* inhibited the germination process of Norway spruce seeds and that the regeneration mostly (60%) took place on decaying dead wood and wind-thrown roots.

The objective of the paper is to describe the structural diversity and regeneration processes on the basis of 57 research plots with the size of 500 m$^2$ that were established in various stages of the natural forest developmental cycle and at various altitudes.

## STUDY AREA AND METHODS

Norway spruce natural forests of Babia hora were declared a nature reserve with the area of 117.6 ha in 1926, later the area was enlarged up to 503.94 ha under the name NNR Kotlina pod Babiou horou. The reserve is situated in the Oravské Beskydy Mts. in the cadastral unit Oravská Polhora on the slopes of W–SW aspect, in the altitude range 1,100–1,725 m. The bedrock is Magura-flysch and the most frequently represented soil types are podzolic Cambisols, Rankers and Podzols. Average annual temperature for the reserve is 2°C (for the vegetation period 6°C), average annual precipitation is 1,600 mm.

Forest communities of the reserve are characterized by a high level of preservation of the original status and belong to three vegetation zones: 6th spruce-beech-fir, 7th spruce, 8th dwarf pine. From the groups of forest types *Fagetum abietino piceosum* (Fap), *Sorbeto-Piceetum* (SP) and *Mughetum* (M) are present. Among the tree species Norway spruce dominates, scattered occur rowan and sycamore maple and beech and Silver fir are scarce. Virgin forests of Norway spruce below the upper timberline change gradually into the dwarf pine and juniper stand. Norway spruce has a typical narrow crown form and its wood has very fine tree rings. The breaks and the drying-out of crown tops are very common. Above 1,500 m a.s.l. the crown canopy of the spruce stands is permanently open, the crowns reach down to the ground and the stems are considerably tapering.

### Quantification of the structural diversity

For the research of the structure a stratified sampling performed in two levels was used. The first stratification level was the altitude divided into 4 altitudinal categories:

A – Norway spruce virgin forest below 1,260 m a.s.l.
B – Norway spruce virgin forest from 1,261 to 1,360 m a.s.l.
C – Norway spruce virgin forest from 1,361 to 1,460 m a.s.l.
D – Norway spruce virgin forest above 1,461 m a.s.l.

The second level for the location of the sample plots was the developmental stage of the virgin forest according to KORPEĽ (1989).

In the first 3 altitudinal categories and in each developmental stage 5 sample plots were established. In the last altitudinal category 4 sample plots were established in each developmental stage. In each developmental stage of the virgin forest 19 sample plots were established. The size of each sample plot stabilized by GPS was 500 m$^2$.

### Methods of the dendrometric attribute measurement

For the measurement the Vertex hypsometer, the calliper and the forest compass were used. From the sample plot centre the following dendrometric attributes were measured on the trees with dbh above 7 cm:

- tree height (m),
- height to the crown base (m),
- tree location in the grades from north (azimuth),
- distance of the tree from the sample plot centre (m),
- diameter at breast height (mm),
- crown width with the vectors $x_1$–$x_4$ according to the azimuth N, E, S, W (0.1 m).

### The assessment of natural regeneration

The objects of the analysis were the individuals with the height below 130 cm, which were registered on 10 subplots each with the size of 1 × 1 m. The subplots were established on the circle with the radius of 6.2 m from the sample plot centre and spacing between the subplots equal to 36°. At the regeneration analysis the following four seedbed types for the assessed individuals were distinguished:

- mineral soil,
- wind-thrown roots,
- stumps (broken part of the tree) with the maximum height of 1.3 m,
- dead wood.

On the subplots 4 height categories (Korpeľ 1989) of the natural regeneration individuals were distinguished:

- individuals with the height below 30 cm,
- individuals with the height from 31 to 50 cm,
- individuals with the height from 51 to 80 cm,
- individuals with the height from 81 to 130 cm.

For the structural diversity analysis of the spruce virgin forest we used the aggregation index according to Clark and Evans (1954), diameter differentiation index according to Füldner (1995) and the complex index according to Jaehne and Dohrenbusch (1997).

### *Aggregation index R* (Clark, Evans 1954)

The aggregation index was developed for the purposes of botanic and phytocoenologic studies. In forestry research this index was used very scarcely. Its importance increased with biodiversity and forest stand diversity studies. For the first time it was probably applied in the works of Pretzsch (1996) and Füldner (1995).

The aggregation index describes the horizontal distribution of trees using the relation of the mean distance between the reference tree and its nearest neighbour and the expected distance between them at the random distribution of trees in the stand. Mathematically it is defined as follows:

$$R = \frac{\frac{1}{n}\sum_{i=1}^{n} r_i}{0.5 \times \sqrt{\frac{Pl}{n}}} \qquad (1)$$

where: $r_i$ – distance of tree $i$ to the nearest neighbour,
$n$ – number of trees on the sample plot,
$pl$ – area of the sample plot (m$^2$).

Clark-Evans index $R$ can theoretically range from 0 at maximum tree clustering to 2.1491 at the regular hexagonal distribution of trees. Index value equal to 1 means that the trees are distributed on the stand area randomly. The stands with the index value > 1 show the tendency to the regular distribution and the index values < 1 the tendency to the clustering.

For the practical use it is not enough to know the index value, i.e. whether the distribution is clustered, regular or random. In the nature most variables have a stochastic character and this index belongs to this type of variables. Therefore, it is important to know whether the difference between the calculated index value and the value expected at random distribution is significant (i.e. the level of significance). This fact can be tested as follows:

$$t = \frac{r_R - r_T}{\sigma_{rT}} = \frac{r_R - r_T}{\frac{0.26136}{\sqrt{n \times \frac{n}{Pl}}}} \qquad (2)$$

where: $r_R$ – real distance to the nearest neighbour,
$r_T$ – expected distance to the nearest neighbour,
$\sigma_{rT}$ – standard error of mean for the expected distance to the nearest neighbour,
$n$ – number of trees on the sample plot,
$Pl$ – area of the sample plot (m$^2$).

If the calculated $t$-value is higher than 1.96 with 95% probability, we can state that the trees in the stand have a clustered or regular distribution according to the value of index $R$ for the given stand.

### *Diameter differentiation index TM* (Füldner 1995)

Tree differentiation is another important parameter of the structural diversity. It can be calculated from various tree attributes (diameter, perimeter, basal area, height, volume). Diameter is a commonly used attribute, because it is simple to measure. Füldner (1995) quantifies the differentiation by the following formula:

$$TM_i = \frac{1}{n}\sum_{j=1}^{n}(1 - d_{ij}) \tag{3}$$

where: $n$ – number of trees on the sample plot,
$d_{ij}$ – the relation between thinner and thicker dbh in the analyzed neighbour tree pair.

The index values range from 0 to 1. The stands with small diameter differentiation have the index values near 0, while the stands with high diameter differentiation reach the index values close to 1.

For better interpretation of the index his author suggested the 4-level scale of differentiation: small (0.0–0.3); average (0.3–0.5); big (0.5–0.7) and very big (0.7–1.0) differentiation. Some years later Aguirre et al. (1998) suggested to divide the *TM* index values into the 5-level scale in order to simplify the comparison of the stands as follows: low differentiation (0.0–0.2); medium differentiation (0.2–0.4); obvious differentiation (0.4–0.6); strong differentiation (0.6–0.8) and very strong differentiation (0.8–1.0).

### *Complex stand diversity index B*
### (Jaehne, Dohrenbusch 1997)

For the evaluation of the general differentiation the index according to Jaehne and Dohrenbusch (1997) was used, which evaluates complex diversity at the forest stand level. The authors created the *B*-index consisting of four variables of stand structural diversity:

- Index of tree species composition (*A*),
- Index of vertical structure (*S*),
- Index of spatial distribution (*V*),
- Index of crown differentiation (*K*).

#### 1. *Index of tree species composition (A)*

$$A = \log(N) \times (Z - Ma_{\max} + Ma_{\min}) \tag{4}$$

where: $N$ – number of tree species,
$Z$ – control parameter, the authors suggest the value 1.5,
$Ma_{\max}$ – relative proportion of the most abundant tree species,
$Ma_{\min}$ – relative proportion of the least abundant tree species.

#### 2. *Index of vertical structure (S)*

$$S = 1 - \frac{\sum_{i=1}^{n} BHD_{\min}}{\sum_{i=1}^{n} BHD_{\max}} \tag{5}$$

where: $n$ – number of measured trees (3 thickest and 3 thinnest trees),
$BHD_{\min}$ – dbh of the thinnest trees (cm),
$BHD_{\max}$ – dbh of the thickest trees (cm).

#### 3. *Index of spatial distribution (V)*

$$V = \left(1 - \frac{\sum_{i=1}^{n} Ab_{\min}}{\sum_{i=1}^{n} Ab_{\max}}\right) \times f \times st \tag{6}$$

where: $n$ – number of measured distances (3 shortest and 3 longest distances between neighbour trees),
$Ab$ – distance between trees (m),
$f$ – correction for the stand density (in the pole-stage and older stands it can be omitted)

$$f = Y - \frac{1}{\dfrac{\sum_{i=1}^{n} Ab_{\min}}{n} + \dfrac{\sum_{i=1}^{n} Ab_{\max}}{n}}$$

$st$ – factor considering coppice sprouts

$$st = N_{250} \times 0.1 + 1$$

$N_{250}$ – n umber of coppice sprouts per 250 m$^2$,
$Y$ – control parameter.

#### 4. *Index of crown differentiation (K)*

$$K = \left\{1 - \log\left(\frac{\sum_{i=1}^{n} Ka_{\min}}{n}\right)\right\} + \left(\frac{\sum_{i=1}^{n} Kd_{\text{mix}}}{\sum_{i=1}^{n} Kd_{\max}}\right) \tag{7}$$

where: $n$ – number of selected trees (2 trees with the smallest and 2 trees with the largest crown diameter),
$Ka_{\min}$ – the smallest height to the crown base (m),
$Kd_{\min}$ – the smallest crown diameter (m),
$Kd_{\max}$ – the largest crown diameter (m).

#### 5. *Complex stand diversity index (B)*

$$B = p \times A + q \times S + V + K \tag{8}$$

where: $A$ – index of tree species composition,
$S$ – index of vertical structure,
$V$ – index of spatial distribution,
$K$ – index of crown differentiation,
$p, q$ – factors of importance ($p = 4$, $q = 3$).

Jaehne and Dohrenbusch (1997) also offered the general evaluation of the stand diversity according to the *B*-index value:

$B \geq 9.0$ – very heterogeneous stand structure
$8.0 \leq B < 8.9$ – heterogeneous stand structure
$6.0 \leq B < 8.0$ – uneven stand structure
$4.0 \leq B < 6.0$ – homogeneous stand structure
$B < 4.0$ – monotonous stands.

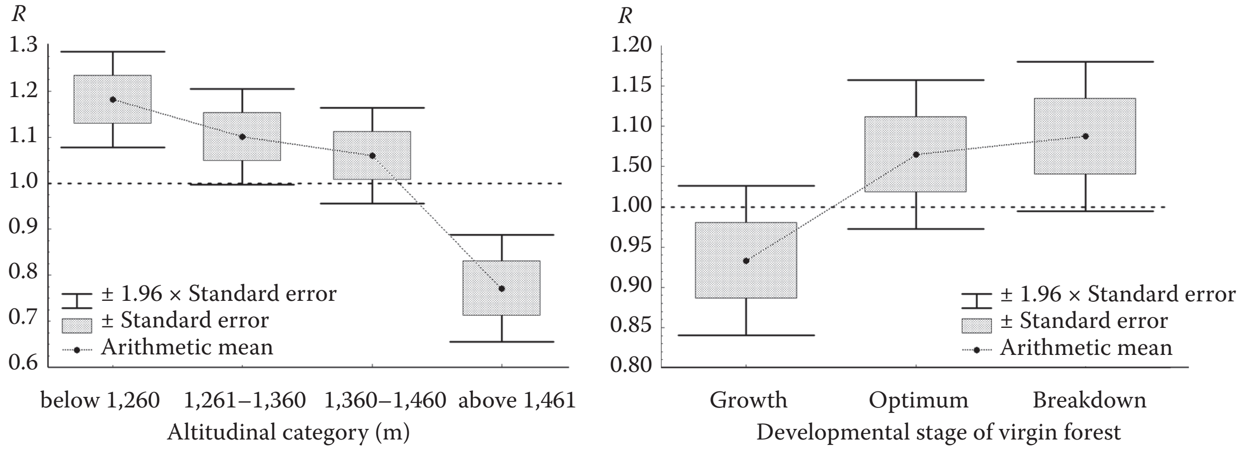

Fig. 1. The values of index *R* in individual altitudinal categories and developmental stages of the virgin forest

## RESULTS

### Structural diversity

The Norway spruce natural forest in Babia hora NNR is characterized by small-scale texture and high ecological stability. From the evaluation of its diversity on the basis of the aggregation index according to Clark and Evans (1954) we can state two substantial facts. The distances between the neighbouring trees show a tendency to clustering with increasing altitude regardless of the developmental stage. The trees with dbh above 7 cm have regular or random distribution at altitudes below 1,460 m, which is significantly changing to cluster distribution at an altitude above 1,461 m. This is due to the vegetative regeneration and the form of family spruces which create tree clusters (Fig. 1).

If we evaluate the developmental stage of the virgin forest regardless of the influence of the altitude, it can be said that the trees in the growth stage have a tendency to clustering ($R = 0.93$) and the Norway spruce virgin forest in other stages has a random distribution of trees ($R = 1.06$ and $R = 1.09$, respectively).

Generally we can state that in the Norway spruce virgin forest in Babia hora NNR the altitude has no significant impact on tree clustering except the upper timber line, where this tendency is confirmed by the cluster or group distribution of family spruces due to the vegetative regeneration. A moderate tendency to tree clustering was confirmed in the growth stage as well.

Füldner's index *TM* was used for the analysis of the diameter differentiation of the trees in the spruce virgin forest (Figs. 2 and 3). If we evaluate the impact of the developmental stage of the virgin forest on the *TM* index value, the highest diameter differentiation can be found in the growth stage. The analysis of variance confirmed that *T*-value in the growth stage is significantly higher than in the breakdown stage ($p = 0.0014$). The differences of *TM*-index between optimum and breakdown stage were not confirmed meaning that their diameter differentiation is equal (Fig. 2).

The analysis of the diameter differentiation according to altitudinal zones and developmental stages of Norway spruce virgin forest (Fig. 3) revealed that only the virgin forest in the altitudinal zone below 1,260 m in the growth stage can be characterized as evidently diameter differentiated. In all other altitudinal zones in this developmental stage the spruce virgin forest is moderately differentiated (Fig. 3) according to the scale by Aguirre et al. (1998). Generally we can state that according to Füldner's *TM* index the Norway spruce virgin forest in the growth and optimum stage has mainly medium differentiated diameter structure except the altitudinal zone 1,361–1,460 m, where in the optimum stage the virgin forest has a single-storied structure with small diameter differentiation.

Interesting values were found by the description of structural diversity according to index *B* (Jaehne, Dohrenbusch 1997). Mean value of the *B*-index

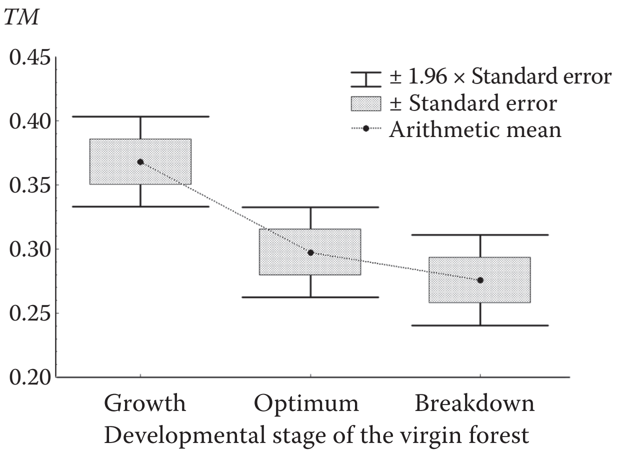

Fig. 2. The values of index *TM* in individual developmental stages of the virgin forest

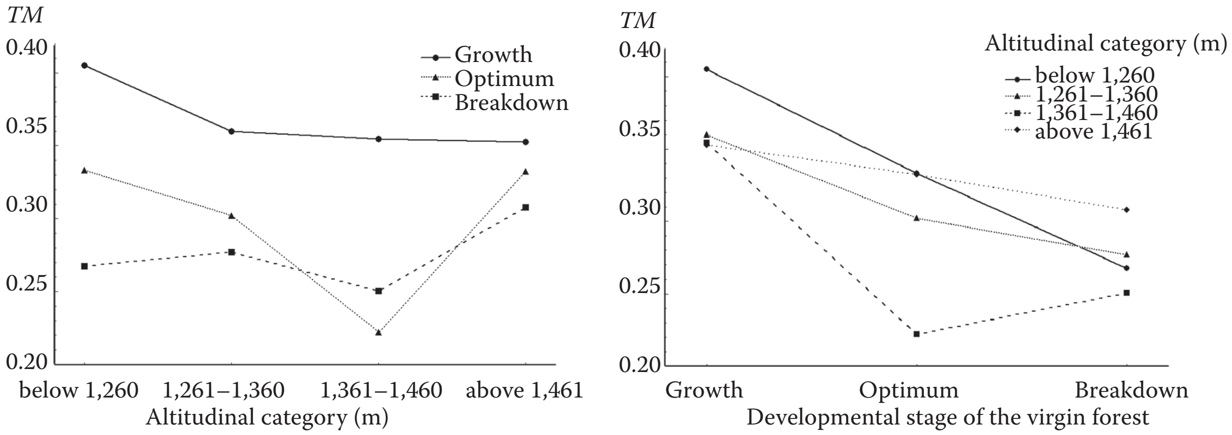

Fig. 3. Average values of index *TM* in individual altitudinal categories and developmental stages of the virgin forest calculated for the set of trees with dbh above 7 cm

calculated for all developmental stages and altitudes of the spruce virgin forest reached 7.5, which ranks the virgin forest to the category of uneven structure. If we evaluate the average *B*-index with regard to increasing altitude, it reaches its highest value in the virgin forest in the zone below 1,260 m (*B* = 9.8) indicating heterogeneous structure. The high value of the index in the first altitudinal zone is caused by a high admixture of rowan and by gradual exchange of the virgin forest developmental stages on small areas. On relatively large parts of the virgin forest the phase of selection forest structure occurs. With increasing altitude the index declines to the value *B* = 4.9 at an altitude 1,361–1,460 m, which stands for the homogeneous stand structure. Increased values of the index *B* = 7.7 in the altitudinal zone above 1,461 m are due to the cluster or group distribution of the virgin forest at the upper timber line (Fig. 4).

While evaluating the influence of the virgin forest developmental stage on the diversity the highest value of the index *B* = 11.8 appeared in the advanced phase of the breakdown stage and in the initial phase of growth stage *B* = 11.5 (Fig. 5). According to the authors of the index this is a virgin forest with very heterogeneous structure. The lowest values of the index were found in the optimum stage, i.e. the virgin forest has homogeneous to monotonous structure.

On the basis of the complex examination it can be said that with increasing altitude the heterogeneity of the spruce virgin forest declines to the altitude 1,460 m, where the compact forest ends and the virgin forest changes its structure to clusters or groups, which causes the increase of the *B*-index value.

### Regeneration processes

The regeneration process dynamics depending on the developmental stage of the virgin forest and the altitude is characterized in Table 1. From this table some relations are obvious. The evaluation of the number of individuals according to the altitude revealed lower values with the increasing altitude practically in all developmental stages of the spruce virgin forest (Table 1). The growth stage, which cre-

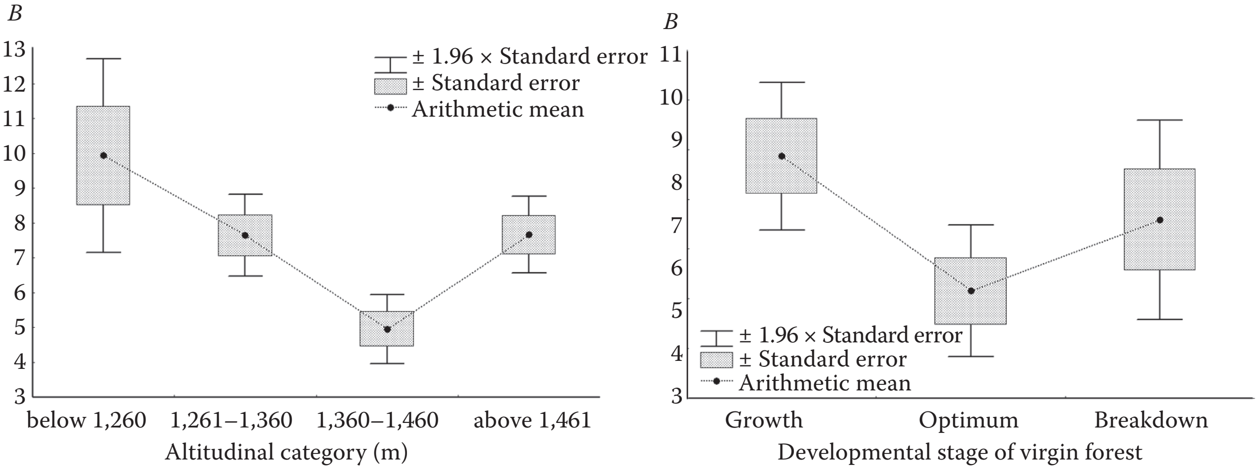

Fig. 4. The values of the complex index *B* in individual altitudinal categories and developmental stages of the virgin forest

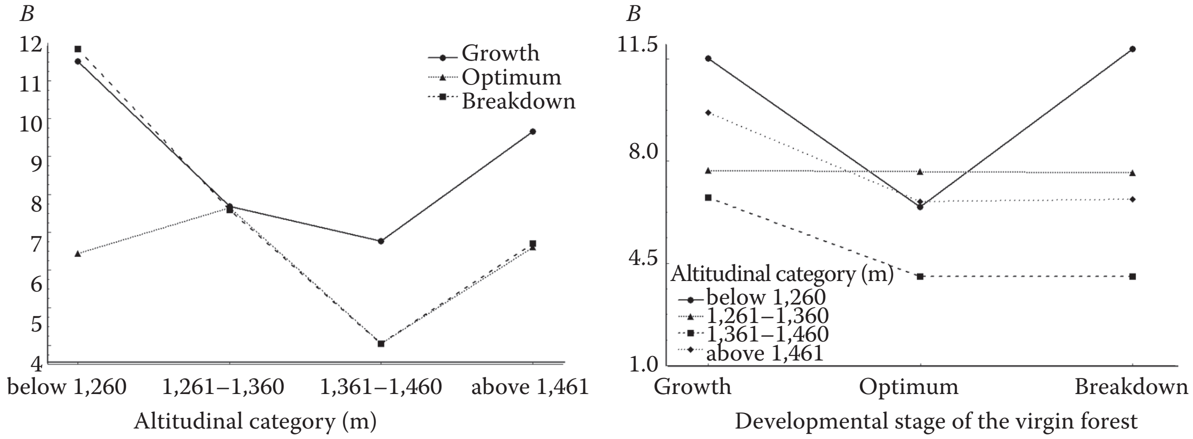

Fig. 5. Average values of the complex index *B* in individual altitudinal categories and developmental stages of the virgin forest

ates the best ecological conditions for the survival and growth of spruce seedlings, has the highest numbers of individuals of this tree species. A distinctive decrease occurred between the altitudinal zone below 1,260 m and the altitudinal zone 1,260 to 1,360 m, where the number of spruce individuals declined from the value 28,600 ha$^{-1}$ to 7,333 ha$^{-1}$. A notable decrease was also registered between the altitudinal zone 1,361–1,460 m and above this limit. The difference was caused by a strong decline of the temperature gradient and change in the overall ecological profile of the virgin forest. At an altitude above 1,461 m, there is a lack of warmth as well as lower germination capacity of spruce, which causes its lower number and lower emergence of vegetative regeneration, which is the leading form of stand regeneration in this altitudinal zone.

The optimum stage permanently has the lowest number of spruce individuals at all altitudes. This is due to unfavourable ecological conditions, above all to the lack of warmth, which is not able to heat up the seedbed at the existing crown canopy and thus to start the germination process to a larger extent. The other reason is the smaller volume of dead wood as the basic form of the seedbed (Table 1). Compared with the optimum stage the breakdown stage has a higher density of spruce regeneration in all altitudinal categories because of better ecological conditions and the presence of a higher amount of dead wood.

Generally we can state the Norway spruce virgin forest has a relatively long time period that is appropriate for the generation exchange below the altitude 1,460 m. In the higher altitudinal zone the vegetative natural regeneration prevails.

Table 1. Tree species structure of natural regeneration (trees/ha) per altitudinal category and seedbed type

| Developmental stage | Tree species | Altitudinal category (m a.s.l.) | | | | Total per stage |
|---|---|---|---|---|---|---|
| | | below 1,260 | 1,261–1,360 | 1,361–1,460 | above 1,461 | |
| Growth | spruce | 28,600 | 7,333 | 6,000 | 250 | 11,444 |
| | rowan | 3,000 | 2,667 | 333 | 0 | 1,778 |
| | total | 31,600 | 10,000 | 6,333 | 250 | 13,222 |
| Optimum | spruce | 8,400 | 2,600 | 1,400 | 250 | 3,316 |
| | rowan | 8,800 | 800 | 800 | 250 | 2,789 |
| | total | 17,200 | 3,400 | 2,200 | 500 | 6,105 |
| Breakdown | spruce | 22,000 | 3,200 | 2,000 | 750 | 7,050 |
| | rowan | 1,800 | 1,600 | 0 | 250 | 900 |
| | total | 23,800 | 4,800 | 2,000 | 1,000 | 7,950 |
| Total per altitudinal category | spruce | 19,667 | 4,563 | 2,643 | 417 | 7,193 |
| | rowan | 4,533 | 1,750 | 357 | 167 | 1,807 |
| | total | 24,200 | 6,313 | 3,000 | 583 | 9,000 |

Table 2. Tree species structure of natural regeneration (trees/ha) per altitudinal category and seedbed type

| Seedbed type | Tree species | Altitudinal category (m a.s.l.) | | | | Total per seedbed |
|---|---|---|---|---|---|---|
| | | below 1,260 | 1,261–1,360 | 1,361–1,460 | above 1,461 | |
| Soil | spruce | 8,000 | 1,625 | 715 | 0 | 2,737 |
| | rowan | 3,400 | 1,500 | 357 | 83 | 1,421 |
| | total | 11,400 | 3,125 | 1,072 | 83 | 4,158 |
| Wind-thrown roots | spruce | 67 | 125 | 0 | 0 | 53 |
| | rowan | 0 | 250 | 0 | 0 | 70 |
| | total | 67 | 375 | 0 | 0 | 123 |
| Lying dead wood | spruce | 10,667 | 2,250 | 1,286 | 333 | 3,824 |
| | rowan | 466 | 0 | 0 | 83 | 140 |
| | total | 11,133 | 2,250 | 1,286 | 416 | 3,964 |
| Stumps | spruce | 867 | 563 | 642 | 83 | 561 |
| | rowan | 733 | 0 | 0 | 0 | 193 |
| | total | 1,600 | 563 | 642 | 83 | 754 |
| Total per altitudinal category | spruce | 19,600 | 4,563 | 2,643 | 417 | 7,175 |
| | rowan | 4,600 | 1,750 | 357 | 166 | 1,825 |
| | total | 24,200 | 6,313 | 3,000 | 583 | 9,000 |

For the conservation of the virgin forest and the generation exchange the question of its seedbed is very important (Table 2). According to the evaluation of the numbers of spruce and rowan individuals growing on the particular seedbed, the most of the individuals were registered on dead wood (logs and stumps) 52.4%. On the mineral soil 4,158 ha on average were recorded, which means 46.2%. If we evaluate the relation between the seedbed type and altitude, an interesting fact appears regarding the wind-thrown roots. The higher number of regenerated individuals on this seedbed in the altitudinal range 1,261–1,360 m suggests that there is a higher amount of wind-thrown trees there. On the other side, with the increasing altitude the proportion of the naturally regenerated individuals of spruce on dead wood grows. In the last altitudinal zone the regeneration on this seedbed type represented even 86% of the individuals.

The importance of dead wood increases if we evaluate the relation between the area proportion of the seedbed type and the amount of natural regeneration (Table 3). Although the mineral soil as a seedbed type constitutes 94.4–97.8% of the area, the number of the regeneration individuals on this seedbed type decreases with the altitude. On the other hand, the area of the lying dead wood declined with the increasing altitude to 2%, and 0.8% at an altitude above 1.61 m. If we consider the seedbed types that are connected with the tree component of the virgin forest (stumps,

Table 3. Proportion of seedbed types for regeneration in the total examined area per altitudinal category (m$^2$/%)

| Altitudinal category (m a.s.l.) | | Seedbed type | | | |
|---|---|---|---|---|---|
| | | soil | wind-thrown roots | lying dead wood | stumps |
| Below 1,260 | (m$^2$) | 9,433.7 | 0.3 | 527.2 | 38.8 |
| | (%) | 94.34 | 0.00 | 5.27 | 0.39 |
| 1,261–1,360 | (m$^2$) | 9,533 | 0.9 | 437.2 | 28.9 |
| | (%) | 95 | 0.01 | 4.37 | 0.29 |
| 1,361–1,460 | (m$^2$) | 9,784 | 0 | 199.2 | 16.8 |
| | (%) | 97.80 | 0 | 2 | 0.20 |
| Above 1,461 | (m$^2$) | 9,914.7 | 0 | 79.5 | 5.8 |
| | (%) | 99.10 | 0 | 0.80 | 0.10 |
| Total | (m$^2$) | 9,666.2 | 0.6 | 310.7 | 22.5 |
| | (%) | 96.66 | 0.01 | 3.11 | 0.23 |

logs, wind-thrown roots), their area represented 3.34% while the proportion of the regeneration growing on them was 53% of the number of individuals.

## DISCUSSION AND CONCLUSIONS

The evaluation of the structural diversity using only one index can lead to incorrect conclusions. If we evaluate the tree distribution of the Norway spruce virgin forest on the basis of the aggregation index according to Clark and Evans (1954), our results confirm the findings from the Norway spruce natural forest in Poľana NNR (Holeksa et al. 2006). The changes in the index values in relation to altitude are also nearly the same in this explored object. Both spruce virgin forests appear to have random or moderately regular distribution of the trees on the area of the virgin forest. However, this index says nothing about the range of tree diameters and heights, or about other dendrometric attributes.

Füldner's index *TM* pointed to the tendency of certain diameter differentiation especially in the spruce virgin forest below the altitude 1,260 m. This index was not used in the analysis and evaluation of the structural diversity of the spruce virgin forest in Poľana NNR.

The complex index according to Jaehne and Dohrenbusch (1997) completed the information about the structural diversity of the Babia hora spruce virgin forest. The values of this index confirmed the decline of the structural diversity with the increasing altitude up to the altitudinal zone 1,460 m. Above this limit its values are increasing again. The heterogeneous forest structure as one of the highest diversity levels at an altitude below 1,260 m is caused by a high admixture of rowan and gradual exchange of the developmental stages on small areas (200–400 m$^2$). Such diversity was not confirmed in the spruce virgin forest in Poľana NNR (Holeksa et al. 2006), where the virgin forest has a monotonous structure. The reasons for this status are better ecological conditions, soil conditions and a large-scale character of the developmental stages in this virgin forest.

The regeneration processes of Norway spruce and rowan in this orographic unit confirmed that rowan as an admixture in this forest ecosystem created appropriate ecological conditions for the regeneration of Norway spruce. In all developmental stages of the virgin forest the number of individuals of both tree species decreases with the increasing altitude. Similar results were reported by Holeksa (1998) from the Polish part of the orographic unit Babia hora and from the orographic unit Poľana as well (Holeksa et al. 2006). Concerning the seedbed types the results confirmed that 53% of spruce seedlings emerged on dead wood and 47% on mineral soil, while the proportion of dead wood was only 3.34%. These results correspond almost entirely with the results of Holeksa (1998). At an altitude above 1,460 m the natural regeneration on dead wood represented 86% and on the mineral soil only 14%. This result confirms the data presented in the paper of Mai (1999).

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Received for publication March 10, 2006
Accepted after corrections May 10, 2006

## Zmena štrukturálnej diverzity a regeneračné procesy smrekového prírodného lesa v NPR Babia hora v závislosti od nadmorskej výšky

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**ABSTRAKT**: Výskum bol zameraný na zistenie dynamiky a formy regeneračných procesov a štrukturálnej diverzity smrekového pralesa v NPR Babia hora v závislosti od nadmorskej výšky. V jednotlivých vývojových štádiách bolo založených zhodne 19 pokusných plôch. Pri posudzovaní štrukturálnej diverzity boli použité rôzne indexy (Clark, Evans 1954; Füldner 1995; Jaehne, Dohrenbusch 1997). Pokiaľ sa týka rozmiestnenia stromov v pralese, nepotvrdil sa vplyv nadmorskej výšky na zhlukovatenie stromov až vo výškovom rozpätí nad 1 461 m, kde sa jedná o skupinové zoskupenie rodinných smrekov. Hrúbková diferenciácia bola štatisticky významne väčšia v štádiu dorastania. Pri hodnotení tohto znaku z hľadiska vplyvu nadmorskej výšky sa významná diferenciácia potvrdila v nadmorskej výške pralesa do 1 260 m v štádiu dorastania. Celkovo na základe Füldnerovho indexu možno konštatovať, že prales má stredne diferencovanú hrúbkovú štruktúru. Na základe komplexného indexu (Jaehne, Dohrenbusch 1997) možno súdiť, že rozrôznenosť pralesa klesá so stúpajúcou nadmorskou výškou do výšky 1 460 m, kde končí kompaktný les. Vyhodnotením klíčneho lôžka bolo zistené, že 46,2 % jedincov obnovy sa nachádzalo na pôde, 52,4 % na moderovom dreve a 1,4 % sa vyskytovalo na kopčekoch po vývratoch.

**Kľúčové slová**: smrek; prales; štrukturálna diverzita; prirodzená obnova

Výskum bol zameraný na zistenie dynamiky a formy regeneračných procesov a štukturálnej diverzity smrekového pralesa v NPR Babia hora, nachádzajúceho sa v nadmorskej výške od 1 190 m do 1 482 m. Analýza sa vykonala v 4 výškových rozpätiach do 1 260 m, 1 261–1 360 m, 1 361–1 460 m a nad 1 460 m n. m. diferencovane podľa vývojového štádia pralesa (Korpeľ 1989) na 57 kruhových skusných plochách s výmerou 500 m$^2$. Kritériom pre rozdelenie na výškové zóny po 100 m bolo výškové rozvrstvenie pralesa. V jednotlivých štádiách (štádium dorastania, štádium optima a štádium rozpadu) bolo založených zhodne 19 pokusných plôch. Plochy boli stabilizované systémom GPS.

Pre posudzovanie a vyhodnotenie štrukturálnej diverzity boli použité rôzne indexy (Clark, Evans 1954; Füldner 1995; Jaehne, Dohrenbusch 1997).

Pokiaľ sa vykonala analýza rozmiestnenia stromov, potvrdil sa vplyv nadmorskej výšky až vo výškovom rozpätí nad 1 460 m, kde vegetatívna obnova má vplyv na skupinové (zhlukové) rozmiestnenie stromov cez proces rodinných smrekov. Smrekový prales v ostatných výškových zónach má náhodné rozmiestnenie stromov (obr. 1).

Pri posúdení hrúbkovej diferenciácie stromov na výskumných plochách hodnotenej Füldnerovým indexom sa potvrdil štatisticky významný rozdiel pralesa v štádiu dorastania (obr. 2). Faktor nadmorskej výšky a jeho vplyv na tento znak štruktúry pralesa bol štatisticky významne potvrdený v nadmorskej výške do 1 260 m v štádiu dorastania (obr. 3). Komplexnou analýzou pomocou tohto indexu bolo potvrdené, že prales v celom výškovom rozpätí má stredne diferencovanú hrúbkovú štruktúru.

Komplexná štrukturálna diverzita hodnotená indexom autorov Jaehne a Dohrenbusch (1997) hovorí o tom, že rozrôznenosť pralesa klesá so stúpajúcou nadmorskou výškou do výšky 1 460 m, kde končí kompaktný les (obr. 5). Priemerná hodnota zistená pre celú rezerváciu ($B$ = 7,5) zaraďuje smrekový prales medzi porasty s nerovnomernou výstavbou, výškovo diferencované. Najvyššia hodnota bola zistená v štádiu rozpadu 11,8 a v štádiu dorastania 11,5 v nadmorskej výške do 1 260 m (obr. 5). V tejto výškovej zóne ide o štruktúru pralesa s veľmi rôznorodou výstavbou.

V otázke prirodzenej obnovy (jedince do výšky 1,30 m) bol zistený poznatok, že vo všetkých štádiách vývojového cyklu smrekového pralesa ich počet klesá so stúpajúcou nadmorskou výškou (tab. 1).

Vyhodnotením vplyvu klíčneho lôžka na početnosť prirodzenej obnovy bolo zistené, že 46,2 % jedincov sa nachádzalo na minerálnej pôde, 52,4 % na moderovom dreve a 1,4 % sa nachádzalo na kopčekoch po vývratoch (tab. 2). Na druhej strane plošný podiel pôdy reprezentoval 96,66 % a moderové drevo len 3,11 % (tab. 3). Možno konštatovať, že moderové drevo je zásadný komponent z pohľadu typu klíčneho lôžka pri regeneračných procesoch tohto smrekového pralesa.

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**Beskydy,** 19 (2006): 143-148
Ediční středisko MZLU v Brně

# ŠTRUKTURÁLNA DIVERZITA SMREKOVÉHO PRÍRODNÉHO LESA V SUPRAMONTANNOM STUPNI NPR BABIA HORA

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**Abstract:** ♠Vorčák, J., ♣Merganič, J., ♣Merganičová, K. (♠Forestry Trades School Tvrdošín, Medvedzie 135, SK-027 47 Tvrdošín, Slovak Republic, ♣FORIM – Forest research, inventory and monitoring, Kpt. Nálepku 277/11, SK-073 01 Sobrance, Slovak Republic), *Structural diversity of natural spruce forests in the subalpine forest belt of National Nature Reserve of Babia hora*, Beskydy, 2006 (19):143-148

The importance of forest stand diversity lies in its relationship with the processes and functionality of ecosystems. The aim of the presented paper was to analyse the influence of elevation and development stage on the value of the complex structural diversity index proposed by Jaehne & Dohrenbusch (1997). To a large extent, the structure of mountain forests of Babia hora is affected by harsh climate conditions, as confirmed by the obtained results. Elevation was found to influence structural diversity more than the development stage of the virgin forest. The average value of the complex structural diversity index $B = 7.5$ indicates that the forests of Babia hora are heterogeneous and of uneven stand structure. As elevation increases, structural diversity of the forest stands decreases until the forest stands begin to be naturally sparsely stocked. With further increase of elevation structural diversity increases, too. In the conditions of Babia hora, this change happens at approximately 1400 m above sea level. The analysis of the development stages revealed that the stage of growth has the highest structural diversity, while the lowest structural diversity is characteristic of the maturity stage.

**Keywords:** structural diversity, mountain forests, diversity quantification

## Úvod a problematika

Trvalosť lesov a ich stabilita je závislá nielen od miery vplyvu exogénnych činiteľov, ale je daná aj genetickým základom a usporiadaním prvkov v ekosystéme. Ekologická stabilita lesov teda úzko závisí od ich diverzity (Miegroet et al. 1996), čo ešte vo väčšej miere platí o horských lesoch, ktoré sú vystavené vo zvýšenej miere pôsobeniu stresových faktorov. Parametre stability vychádzajú z rôznorodosti lesa v čase a priestore, z produkcie a produktivity lesných ekosystémov a v neposlednej miere zo štruktúry a výstavby. Vo všeobecnosti je stav horských lesov výslednicou genetickej, druhovej a ekosystémovej diverzity za súčinnosti pôsobenia exogénnych činiteľov (Vorčák 2005).

Významnou zložkou diverzity lesných ekosystémov je popri druhovej diverzite štrukturálna diverzita porastov, ktorá sa skladá z biotických a abiotických prvkov a môže sa tiež charakterizovať ako eko-morfologická alebo priestorovo-ekologická diverzita (Lexer et al. 2000). Pre správnu kvantifikáciu štrukturálnej diverzity je v prvom rade nutné vysvetliť niektoré základné pojmy. Pojem štruktúra vyjadruje vo všeobecnosti špecifické usporiadanie prvkov v systéme (Gadow 1999), ich umiestnenie v danom systéme a vzájomné prepojenie (Heupler 1982 in Lübbers 1999). V náväznosti na túto definíciu sa štruktúra lesa definuje ako priestorové rozmiestnenie biomasy, čiže stromov a ich vlastností (hrúbka, výška a pod.) (Zenner 1999, Gadow 1999, Gleichmar & Gerold 1998).

Štruktúru lesa môžeme vnímať na viacerých hierarchických úrovniach (Kint et al. 2000). Na úrovni krajiny je definovaná ako rozrôznenie porastových typov (O´Hara 1998) a stupeň fragmentácie biotopov (Andrén 1994). Pre výskum biodiverzity sa však za dôležitejšiu považuje porastová úroveň (Kuuluvainen et al. 1996), pretože plošné a vertikálne rozmiestnenie stromov v poraste definuje trojdimenzionálny priestor biotopu vtákov, hmyzu, cicavcov, epifytov, nedrevnej vegetácie a pôdnych mikroorganizmov (Ratcliffe et al. 1986, Kuuluvainen et al. 1996). Porastová štruktúra človekom nenarušených lesných ekosystémov v sebe navyše nesie informácie o dynamike vývoja pralesov (Hofgaard 1993). V hospodárskych lesoch je však táto veličina nemenej významná, pretože napomáha zhodnotiť ich súčasný stav, ich tzv. blízkosť k prírode a v konečnom dôsledku aj ich ekologickú stabilitu (Pretzsch 1995, 1998, Zenner 1999).

Ako uvádza Merganič et al. (2003), štruktúra lesa sa popisuje rôznymi štruktúrnymi prvkami, ako je priestorové rozmiestnenie, hustota, diferenciácia a zmiešanie jedincov (Zenner 1999). Pri správnom hodnotení štruktúry porastov je však potrebné vnímať ju komplexne ako charakteristiku zloženú z:

- horizontálnej štruktúry, t.j. plošného, dvojdimenzionálneho rozmiestnenia stromov na ploche porastu
- vertikálnej štruktúry vyjadrenej rozrôznením porastu vo vertikálnom smere.

Lübbers (1999) k týmto dvom zložkám pridáva ešte aj tzv. mikroštruktúry, pod ktorými rozumie napr. množstvo moderového dreva v poraste, formy stromov a pod.

Gadow & Hui (1999) definujú štruktúru ako súbor nasledovných charakteristík:

- pozícia - plošné rozmiestnenie stromov po ploche porastu
- zmiešanie - vzájomná pozícia druhov drevín v poraste
- diferenciácia - relatívne veľkostné zmeny medzi susediacimi stromami v horizontálnom a vertikálnom smere

Pre popis a kvantifikáciu horeuvedených zložiek je možné použiť viacero metód, pričom najvyššiu výpovednú hodnotu majú indexové metódy, ktoré používajú matematicko-štatistické výpočty.

## Materiál a metodika

Pri štúdiu vzťahov fungujúcich v ekosystéme hrajú významnú úlohu prírodné lesy, keďže sú činnosťou človeka najmenej narušené. Medzi takéto ekosystémy patria aj horské smrekové lesy v masíve Babej hory v Oravských Beskydách, ktoré boli predmetom výskumu pri riešení projektu financovaného Ministerstvom životného prostredia „Monitoring diverzity horských lesov severnej Oravy“ (Merganič et al. 2003). V rámci tohto projektu bolo založených 57 skusných plôch kategorizovaných do štyroch kategórií nadmorskej výšky (do 1260, 1260-1360, 1360-1460, nad 1460 m n. m) a troch vývojových štádií prírodného lesa (štádium dorastania, optima a rozpadu). Informačné spektrum tvorilo cca 55 kvantitatívnych a kvalitatívnych znakov, ktoré sa na skusnej ploche merali a hodnotili. Celkovo bolo pomeraných a ohodnotených 2846 ks stromov s výškou nad 1.3m a 513ks jedincov s výškou do 1.3m. Vytvorená databáza umožňuje vierohodnú rekonštrukciu lesného porastu ako aj podrobnú analýzu vzťahov medzi jednotlivými jedincami, resp. stromovou a ostatnými zložkami lesného ekosystému, ktoré sa dajú charakterizovať parametrami štrukturálnej diverzity.

V predkladanej práci analyzujeme vplyv nadmorskej výšky a vývojového štádia na index štrukturálnej diverzity. Pre kvantifikáciu štrukturálnej diverzity sme použili komplexný index štrukturálnej diverzity zahŕňajúci viac zložiek štruktúry porastov navrhnutý autormi Jaehne & Dohrenbusch (1997). Ide o tzv. *B* - index, ktorý pozostáva zo štyroch parciálnych premenných štrukturálnej diverzity porastu:

- Index drevinovej skladby porastu (*A*)
- Index vertikálnej štruktúry (*S*)
- Index priestorového rozdelenia (*V*)
- Index diferenciácie korún (*K*)

Ich presná matematická formulácia je uvedená v prácach Jaehne & Dohrenbusch (1997), Merganič et al. (2003) a Vorčák (2005).

Na základe vypočítanej hodnoty indexu *B* navrhli Jaehne & Dohrenbusch (1997) aj stupnicu pre celkové vyhodnotenie porastovej štrukturálnej diverzity:

$B \geq 9.0$ - veľmi rôznorodá výstavba porastu
$8.0 \leq B < 8.9$ - rôznorodá výstavba porastu
$6.0 \leq B < 8.0$ - nerovnomerná výstavba porastu
$4.0 \leq B < 6.0$ - rovnomerná výstavba porastu
$B < 4.0$ - monotónne porasty

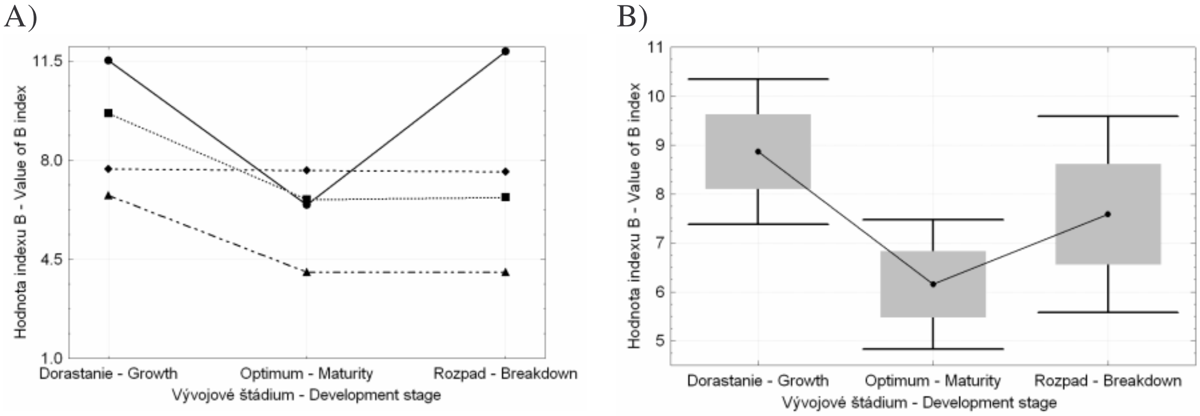
## Výsledky a diskusia

Horské smrekové lesy v masíve Babej hory vykazujú vo všeobecnosti vysoký stupeň diverzity - rôznorodosti a prirodzenosti. Majú pestrú výstavbu a textúru, čo nepriamo dokazuje aj priemerná hodnota komplexného indexu diverzity $B = 7.5$, ktorá podľa stupnice navrhnutej Jaehne & Dohrenbusch-om (1997) radí skúmané porasty k porastom s nerovnomernou výstavbou. Takéto porasty sú na základe popisu uvedených autorov výškovo diferencované, tvorené dvoma až troma porastovými vrstvami s náhodným rozmiestnením stromov na ploche a len zriedkavo dochádza k ich zhlukovaniu. Tento popis v plnej miere odzrkadľuje skutočný stav smrekových porastov v masíve Babej hory.

Analýza vplyvu skúmaných faktorov (nadmorská výška a vývojové štádium) na *B* index potvrdila ich signifikantný vplyv. Vplyv vývojového štádia sa preukázal na 95% a vplyv nadmorskej výšky na 99.9% hladine spoľahlivosti. Kombinovaný vplyv oboch faktorov na hodnoty indexu *B* sa nepotvrdil, preto je možné konštatovať, že zásadný vplyv na jeho hodnoty má faktor nadmorská výška.

Podrobnejšou analýzou priemerných hodnôt indexu *B* v rámci vývojového štádia zisťujeme, že najvyššiu hodnotu 8.9 dosahuje štádium dorastania, čo je na hranici medzi porastami s rôznorodou a veľmi rôznorodou výstavbou. Štádium optima má naopak najnižšiu hodnotu 6.2, kým štádium rozpadu je charakterizované priemernou hodnotou 7.5 (obr. 1). V posledných dvoch prípadoch ide o nerovnomernú výstavbu porastov.

V závislosti od nadmorskej výšky hodnota indexu *B* klesá od 1. po 3. výškovú kategóriu (obr. 2). Najvyššia hodnota indexu *B* v 1. výškovej kategórii – 9.9 - je spôsobená vysokým podielom prímesi jarabiny a charakterom porastov, keďže tu prebieha plynulé striedanie vývojových štádií a fáz na malých plochách. Časti porastov sa tu vyskytujú vo výbernej fáze, čo zvyšuje štrukturálnu diverzitu porastov vyjadrenú pomocou vyššie uvedeného indexu. Tieto porasty môžeme podľa stupnice Jaehne & Dohrenbusch (1997) označiť ako porasty s veľmi rôznorodou výstavbou. V 3. výškovej kategórii má index najmenšiu hodnotu 4.9. Táto hodnota indikuje rovnomernú výstavbu porastu charakteristickú pre porasty tvorené jednou drevinou – v našom prípade smrekom, ktoré majú jednovrstvovú výstavbu s nerovnomerným až náhodným rozmiestnením stromov na ploche. Od 3. výškovej kategórie hodnota indexu *B* opäť stúpa a v 4. výškovej kategórii dosahuje hodnotu 7.7. Rast hodnoty indexu v 4. výškovej kategórii je spôsobený rozpojeným charakterom porastov (bioskupiny) a jarabinou, ktorá sa v rozpojených porastoch začína opäť uplatňovať. Na základe stupnice uvedených autorov ide o porasty s nerovnomernou výstavbou.



Obr. 1: Priebeh priemerných hodnôt indexu štrukturálnej diverzity *B* v závislosti od vývojového štádia prírodného lesa a kategórie nadmorskej výšky (A) a testovanie vplyvu vývojového štádia na hodnoty indexu *B* (B).

Legenda:

Obrázok A: kategória nadmorskej výšky ● do 1260 m n.m., ■ od 1261 do 1360 m n.m., ◆ od 1361 do 1460 m n.m., ▲ nad 1461 m n.m.

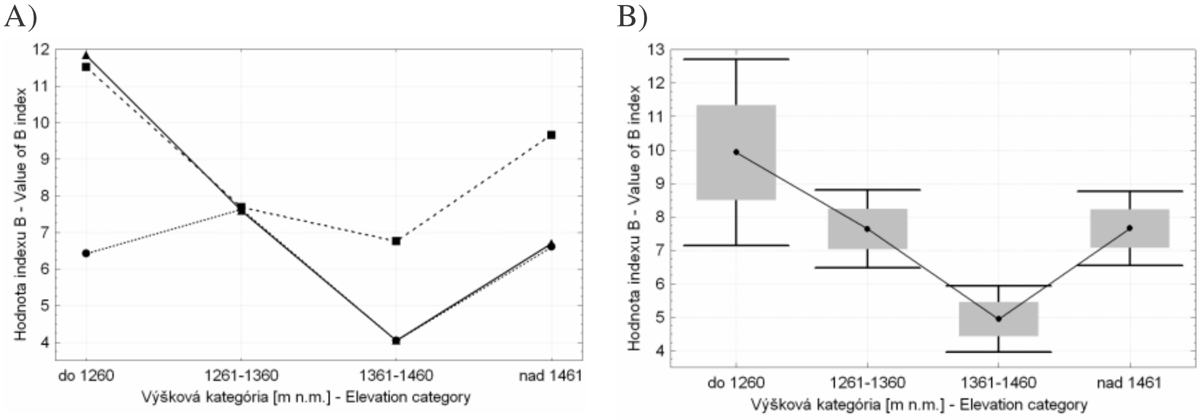
Obrázok B: ● aritmetický priemer, ▨ ± stredná chyba, ⊥ ± 1.96 · stredná chyba

Fig. 1: Average values of the complex structural diversity index *B* in individual development stages considering elevation categories (A) and test of the influence of the development stage on index *B* (B).

Legend:

Figure A: elevation category ● below 1260 m, ■ 1261-1360 m, ◆ 1361-1460 m, ▲ above 1461 m

Figure B: ● arithmetic mean, ▨ ± standard error, ⊥ ± 1.96 · standard error



Obr. 2: Priebeh priemerných hodnôt indexu štrukturálnej diverzity *B* v závislosti od kategórie nadmorskej výšky a vývojového štádia prírodného lesa (A) a testovanie vplyvu kategórie nadmorskej výšky na hodnoty indexu *B* (B).

Legenda:

Obrázok A: Vývojové štádium prírodného lesa ■ Dorastanie, ● Optimum, ▲ Rozpad

Obrázok B: ● aritmetický priemer, ▨ ± stredná chyba, ⊥ ± 1.96 · stredná chyba

Fig. 2: Average values of the complex structural diversity index *B* in individual elevation categories with regard to the development stages (A,) and test of the elevation influence on index *B* (B).

Legend:

Figure A: Development stage of the virgin forest ■ Growth, ● Maturity, ▲ Breakdown

Figure B: ● arithmetic mean, ▨ ± standard error, ⊥ ± 1.96 · standard error

Absolútne najvyššia hodnota indexu bola zaznamenaná v 1. výškovej kategórii v štádiu dorastania (11.5) a v štádiu rozpadu (11.8). Ide o porasty vo fáze obnovy, ktoré podľa slovného hodnotenia Jaehne & Dohrenbusch-a (1997) zaraďujeme k porastom s veľmi rôznorodou výstavbou. Sú to porasty s výbernou štruktúrou a prímesou jarabiny, ktorá často tvorí pňovú výmladnosť, čím zvyšuje ich diverzitu. Porasty majú vertikálny zápoj s výrazným zoskupením hlavne mladších stromov tvoriacich hlúčiky. Najnižšiu rovnakú hodnotu má index v štádiu optima a rozpadu v 3. výškovej kategórii (4.04). Ide o porasty na hranici monotónnych porastov a porastov s rovnomernou výstavbou. Táto výšková kategória má prevažne charakter jednovrstvových smrekových porastov (halové porasty) s nerovnomerným až náhodným rozmiestnením stromov na ploche, ktoré sa podľa nášho predpokladu začnú rozpadať, aby sa vytvorili vhodné podmienky pre regeneračné procesy. S časovým predstihom a vo veľkoplošnej forme sa takýto vývoj uskutočňuje v Západných Tatrách v Látanej doline (Vorčák 2005).

## Záver

Význam diverzity spočíva v jej silnom prepojení na procesy i funkčnosť ekosystému. Diverzita má blízky vzťah k naturálnej produkcii, ale v hlavnej miere je spájaná so stabilitou ekosystému. V tejto súvislosti má dôležité postavenie oblasť zaoberajúca sa metódami kvantifikácie diverzity, pretože matematická kvantifikácia umožňuje nielen objektívnejšie ohodnotiť, ale v konečnom dôsledku aj lepšie pochopiť a popísať vzťahy, ktoré v ekosystéme fungujú. Medzi prvé práce zaoberajúce sa kvantifikáciou štrukturálnej diverzity horských lesov na Slovensku patrí práca „Monitoring diverzity horských lesov severnej Oravy“. (Merganič et al. 2003), z ktorej vychádza aj tento príspevok.

V predkladanej práci sa analyzuje vplyv nadmorskej výšky a vývojového štádia na komplexný index štrukturálnej diverzity navrhnutý autormi Jaehne & Dohrenbusch (1997). Skúmané horské lesy v supramontánnom stupni sa nachádzajú v drsných prírodných podmienkach, ktoré v rozhodujúcej miere formujú ich štruktúru. Toto konštatovanie sa v plnej miere potvrdilo, pretože z dosiahnutých výsledkov vyplýva, že nadmorská výška významnejšie ovplyvňuje štrukturálnu diverzitu a tým aj štruktúru vývojových štádií prírodného lesa. Súhrnne je možné konštatovať, že štrukturálna diverzita so zvyšujúcou sa nadmorskou výškou klesá až do výšky, kde dochádza k prirodzenému rozpojeniu porastov, odkiaľ opäť stúpa. V podmienkach Babej hory ide o výškovú hranicu okolo 1400 m n.m. Zároveň sa jednoznačne potvrdilo, že najvyššiu štrukturálnu diverzitu majú porasty v štádiu dorastania, nasleduje štádium rozpadu a optima.

Dlhodobo formovaná štruktúra porastov sa vyznačuje vlastnosťami, ktoré udržujú tieto ekosystémy v rovnováhe a stabilite. Štrukturálna diverzita lesných porastov je veľmi dôležitou charakteristikou stavu a vývoja porastov. Úzko súvisí so všetkými procesmi prebiehajúcimi v ekosystéme. Poznanie a rešpektovanie týchto zákonitostí môže významne napomôcť lesnému hospodárovi pri obhospodarovaní lesa v meniacich sa ekologických podmienkach a zabezpečiť jeho trvalosť. Vysoký stupeň stability porastov na Babej hore dokazuje aj skutočnosť, že ani vetrové smršte v novembri 2002 a 2004 výrazne nepoškodili tento ekosystém, hoci v ďalších horských oblastiach na Slovensku vznikli katastrofálne škody.

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**Beskydy**, 20 (2007): 275-282
Ediční středisko MZLU v Brně

# EKOLOGICKÁ STABILITA LESNÝCH PORASTOV V NPR BABIA HORA

**J. Vorčák, J. Merganič, K. Merganičová**

**Abstract :** [1]Vorčák, J., [2]Merganič, J., [2]Merganičová, K. ([1]Forestry Trades School Tvrdošín, Medvedzie 135, SK-027 47 Tvrdošín, Slovak Republic, [2]FORIM – Forest research, inventory and monitoring, Kpt. Nálepku 277/11, SK-073 01 Sobrance, Slovak Republic)*, Ecological stability of forest stands in the National Nature Reserve of Babia hora*, Beskydy, 2007 (20): 275-282

In the presented paper we analyse the static stability of forest stands in the National Nature Reserve Babia hora using five indicators: depigmentation, defoliation, crown ratio, h/d ratio, and tree damage. The data were collected using stratified sampling in 57 circle sample plots each of 500 $m^2$. The values of the examined features indicate that the mountaineous forests of Babia hora belong to highly stable ecosystems formed in the long term into the state of internal homeostasis, which guarrantees permanent stability in different extreme situations.

**Keywords**: static stability, depigmentation, defoliation, h/d ratio, crown ratio, tree damage

## Úvod a problematika

Pod stabilitou lesného ekosystému sa v ekologickom zmysle slova rozumie buď nemennosť štruktúry a funkcií jeho komponentov (autotrofné a heterotrofné organizmy) nezávisle od toho, či je ekosystém vystavený vplyvom rozličných exogénnych faktorov (COOLIER et al. 1977 in STOLINA 1985), prípadne, ak bol ekosystém narušený, jeho schopnosť opätovne vytvoriť prostredníctvom biotických mechanizmov pôvodnú štruktúru a funkciu jeho zložiek (ODUM 1977). Ide teda o dvojaký prejav stability lesného ekosystému, pričom v prvom prípade sa jedná o rezistenciu a v druhom o rezilienciu ekosysténu. Tieto schopnosti lesného ekosystému sa často nazývajú spoločným názvom homeostatická spôsobilosť, prípadne jednoducho - homeostáza ekosystému (MARGALEF 1969 in STOLINA 1985).

Schopnosť lesného ekosystému zotrvávať v dynamicky (zdanlivo) nemennom stave je výslednicou progresívnej sukcesie. Počas tohto procesu sa postupne vytvára stále pestrejšie a zrelšie spoločenstvo rastlín a živočíchov tesne späté s danými podmienkami abiotického prostredia. Homeostáza je teda vo všeobecnosti označovaná schopnosť systému udržiavať prvky pomocou autoregulačných procesov v relatívne statickom stave. Čím je ekosystém na vyššom organizačnom stupni, tým má väčšiu homeostatickú (vyrovnávaciu) schopnosť (ODUM 1997). Pri sústavnom pôsobení exogénnych činiteľov nemusí homeostatická schopnosť vrátiť systém do východiskového stavu, ale môže ho dostať na novú úroveň – trajektóriu (MÍCHAL 1992).

Princíp stability v ekosystéme a spoločenstve rastlín vychádza z dvoch predpokladov: po prvé z druhovej skladby a po druhé odolnosti rastlinného spoločenstva proti poškodeniu, vplyvu exogénnych činiteľov (MIEGROET 1996). Parametre stability, ktoré charakterizujú stav lesného ekosystému, sú (MIEGROET 1996):

- diverzita druhov vyjadrená v čase a priestore
- produkcia a produktivita

- štruktúra a výstavba.

U lesných ekosystémov sa optimálnou stabilitou vyznačujú prírodné lesné ekosystémy, pralesy (JENÍK 1979, KORPEĽ 1989). Ich charakteristickým znakom je výrazná diverzita a funkčná ustálenosť, automatické vyrovnávanie výkyvov v štruktúre a funkcii komponentov vyvolaných vplyvmi exogénnych činiteľov (STOLINA 1985). Dosiahnutie ekologickej stability je považované za najvyšší princíp v súčasnej koncepcii prírode blízkeho pestovania lesa (KORPEĽ & SANIGA 1995).

Významným ukazovateľom ekologickej stability horských lesov je statická stabilita (VORČÁK 2005). Vyjadruje predovšetkým odolnosť porastu proti vetru, námraze a snehu. Pre statickú stabilitu má veľký význam výskyt vitálnych stromov hornej vrstvy s hlbokými korunami.

Stabilita lesa, ako predpoklad trvalej drevnej produkcie a verejnoprospešných funkcií lesa, bola sformovaná dávno pred dnešným chápaním ekologickej stability (MOROZOV 1949 in STOLINA 1985, KONŠEL 1931 in STOLINA 1985, RUBNER 1934 in STOLINA 1991).

## Materiál a metodika

Empirický materiál pochádza z NPR Babia hora, ktorá patrí orograficky do sústavy vonkajších Západných Karpát, časti Oravských Beskýd, do komplexu horského masívu Babej hory. Národná prírodná rezervácia zaberá celkovo 503.94 ha a je umiestnená na západných, južných a juhozápadných svahoch Babej hory v nadmorskej výške 1100 až 1725 m n. m. (KORPEĽ 1989). Geologické podložie je tvorené zo súvrství rozličných flyšových hornín. Hlavnými pôdnymi predstaviteľmi sú litozem, kambizem a podzol. Priemerné ročné teploty dosahujú vo vrcholových polohách 2°C, v nižších polohách 4°C. Dlhodobý priemerný ročný úhrn zrážok je 1400 mm.

Lesné porasty sú tvorené prevažne smrekom obyčajným (*Picea abies* L.), vtrúsene sa vyskytujú jarabina vtáčia (*Sorbus aucuparia* L.), jedľa biela (*Abies alba* Mill.) a v spodnej časti územia aj buk lesný (*Fagus sylvatica* L.). Les vystupuje približne do nadmorskej výšky 1460 - 1480 m n. m. Nad touto hranicou lesa nastupuje pásmo kosodreviny vystriedané v najvyšších častiach NPR alpskými lúkami.

V rámci inventarizácie zalesneného územia NPR Babia hora (MERGANIČ *et al.* 2003) bolo založené 57 kruhových skusných plôch o rozlohe 500 m$^2$ tak, aby boli rovnomerne rozdelené medzi tri vývojové štádiá (dorastanie – optimum - rozpad podľa KORPEĽA 1989) a štyri výškové kategórie, t.j. do 1260 m n. m, 1260 – 1360 m n. m., 1360 – 1460 m n.m. a nad 1460 m n. m. Informačné spektrum tvorilo cca 55 kvantitatívnych a kvalitatívnych znakov, ktoré sa na skusnej ploche merali a hodnotili. Celkovo bolo pomeraných 2846 ks stromov. Vytvorená databáza umožňuje vierohodnú rekonštrukciu lesného porastu ako aj podrobnú analýzu vzťahov medzi jednotlivými jedincami, resp. medzi stromovou zložkou a ostatnými zložkami lesného ekosystému.

Pre účely tejto práce boli zo zisťovaného informačného spektra použité údaje o depigmentácii, defoliácii, poškodení stromu a informácie odvodených veličín kvantifikujúcich korunovosť a štíhlostný koeficient. Depigmentácia a defoliácia bola určovaná s presnosťou na 5% podľa medzinárodne platnej metodiky pre monitoring stavu lesov (LVÚ ZVOLEN 1998). Poškodenie stromu sa chápe ako frekvencia výskytu poškodenia bez ohľadu na druh poškodenia. Korunovosť je definovaná ako pomer dĺžky koruny k výške stromu, štíhlostný koeficient ako pomer výšky stromu k prsnej hrúbke. Stromy so štíhlostným koeficientom väčším ako 100 sú staticky labilné, v intervale 80 – 100 stabilné, s koeficientom menším ako 80 veľmi stabilné a menším ako 55 sú solitéry s najvyššou stabilitou (MÍCHAL et al. 1992). Uvedené veličiny boli spracované samostatne pre dva podsúbory stromov a to pre stromy hornej a strednej vrstvy definovaných v zmysle biosociologického postavenia stromov v poraste podľa Kraftovej stupnice. Hornú vrstvu tvorili stromy nadúrovňové a úrovňové (1, 2) a strednú vrstvu stromy čiastočne úrovňové (3). Štatistická analýza sa vykonala použitím dvojfaktorovej analýzy variancie.

## Výsledky a diskusia

### *Depigmentácia*

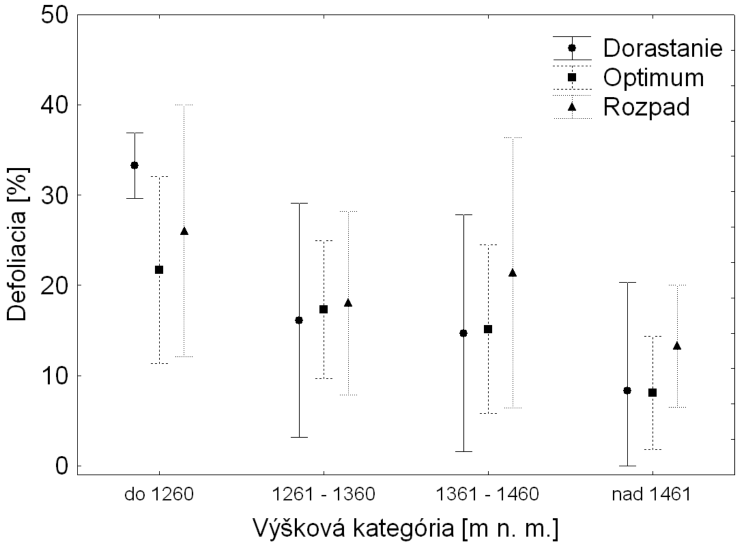
Depigmentácia je výslednicou pôsobenia pôdno-klimatických faktorov, zmeny chemizmu atmosféry a žiarenia na fyziologické procesy rastlín, ako aj výsledkom vnútrocenotických vzťahov. Prejavuje sa zmenou zafarbenia asimilačných orgánov rastlín. Výrazný vplyv na depigmentáciu má tiež ozón ($O_3$), ktorý má fotooxidačný účinok. Prvým prejavom poškodenia ozónom je úbytok chlorofylu prejavujúci sa blednutím zeleného zafarbenia ihlíc a listov, prípadne vznikom chlorotických a neskoršie nekrotických škvŕn. Celková priemerná depigmentácia hornej porastovej vrstvy bez rozdielu kategórie nadmorskej výšky (v.k.) a vývojového štádia činila 0,84±0,16% (± stredná chyba) a strednej porastovej vrstvy 1,43±0,37%. V obidvoch prípadoch ide o zanedbateľnú hodnotu a predstavuje stupeň 0 podľa použitej stupnice. Pri slovnom popise stupňa ide o stav asimilačných orgánov bez zmeny zafarbenia. Vyššie percento depigmentácie je v strednej vrstve pravdepodobne ako následok vnútrocenotických vzťahov a prístupu svetla. Horná vrstva je charakteristická nárastom depigmentácie s nárastom nadmorskej výšky, čo je podľa nášho predpokladu ovplyvnené ozónom ($O_3$). Ozón, ako fotooxidant zvyšuje svoj účinok práve s nárastom nadmorskej výšky. Ide v podstate o normálny trend, čo dokazujú aj zistenia autorov INNES et al. (1998), FLEISCHER (1999). V strednej porastovej vrstve je vo všetkých vývojových štádiách najvyššia depigmentácia v 3. v.k., v štádiu optima dosahuje priemerná hodnota 3,4%. Najvyššiu hodnotu depigmentácie 16% sme namerali na skusnej ploche 47, ktorá sa nachádzala v 3. v.k. v štádiu optima. Ide o stupeň 1, čo je definované ako slabá zmena zafarbenia asimilačných orgánov.

### *Defoliácia*

Defoliácia, čiže strata asimilačných orgánov (SAO), je považovaná za jeden z hlavných ukazovateľov zdravotného stavu drevín (RAČKO 1994). Podobne ako depigmentácia aj defoliácia je odrazom zmeny vo fyziologických procesoch rastlín a závisí od obdobných faktorov. Najvýraznejší vplyv na defoliáciu má zmena chemizmu atmosféry, čiže znečistenie ovzdušia imisiami. Dlhodobé priame pôsobenie imisií sa na lesných porastoch prejavuje postupnou stratou ihličia, rednutím korún a následným uschýnaním jedincov, ale aj celých porastov. Nepriame pôsobenie imisií spôsobuje chronické fyziologické zmeny, čo spôsobuje nárast dispozície na poškodzovanie prírodnými škodlivými činiteľmi biotického i abiotického charakteru (STOLINA 1989).

Celková priemerná hodnota defoliácie za skúmanú oblasť je u hornej porastovej vrstvy 18±1,45% a strednej 17±1,7%, čo je v oboch prípadoch 1. stupeň, t.j. slabá defoliácia. Najvyššia priemerná hodnota defoliácie bola v hornej vrstve v 1. v.k. v štádiu dorastania a predstavovala 33±1,3% - stredná defoliácia (obr. 1). Od 2. v.k. rastie defoliácia v obidvoch porastových vrstvách od štádia dorastania cez optimum a najvyššia je v rozpade. Na Babej hore mala na defoliáciu hornej porastovej vrstvy signifikantný vplyv nadmorská výška (IS 99.9%) a strednej vrstvy vývojové štádium (IS 95%). Priemerná hodnota defoliácie s rastúcou nadmorskou výškou klesá, čo je spôsobené prirodzeným rozpojením porastov. Vďaka tomu sú doposiaľ lesy v masíve Babej hory veľmi málo postihnuté stratou asimilačných orgánov aj napriek tomu, že v minulosti (80. roky minulého storočia) boli vystavené intenzívnemu pôsobeniu polutantov z miestnych i diaľkových zdrojov znečistenia. V uvedenom období boli lesy v nižších polohách vyhlásené za imisné lesy.

Najvyššie hodnoty defoliácie (50%) boli namerané na ploche 21 a 22 v 2. v.k. v štádiu dorastania. Ide o 2. stupeň poškodenia (stredná defoliácia). Porasty na týchto plochách sa nachádzali vo fáze dožívania so silne preriedenou hornou etážou.



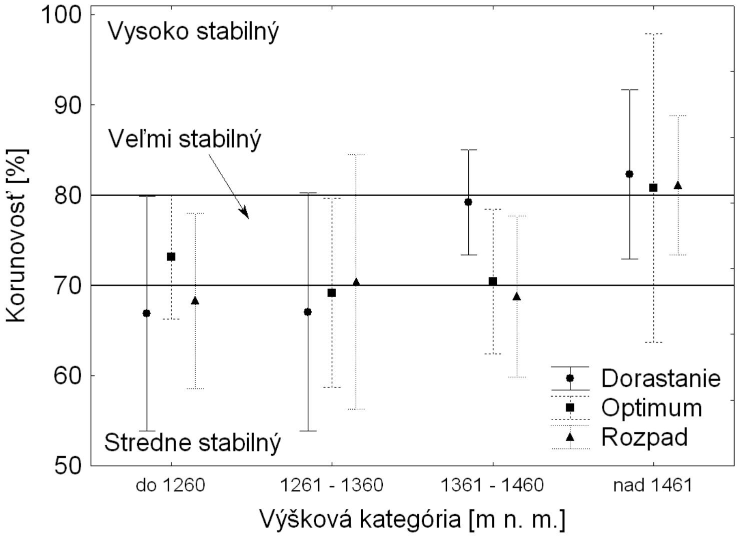
Obr. 1 Priebeh priemerných hodnôt defoliácie hornej vrstvy stromov v rámci vývojového štádia a kategórie nadmorskej výšky.
Legenda:
Vývojové štádium: ● Dorastanie, ■ Optimum, ▲ Rozpad;
●, ■, ▲ aritmetický priemer, ⌶ ± 1.96 · stredná chyba

Fig. 1 Average defoliation of the upper tree layer in individual development stages and elevation categories.
Legend:
Development stage: ● Growth, ■ Maturity, ▲ Breakdown;
●, ■, ▲ arithmetic mean, ⌶ ± 1.96 · standard error

***Korunovosť***

Korunovosť je nielen významným ukazovateľom statickej stability, ale hrá dôležitú úlohu aj vo fyziológii drevín a odolnosti voči klimatickým zmenám a znečisteniu ovzdušia. Smrekové horské lesy v masíve Babej hory sú typické tým, že stromy majú silné a hlboké koruny. Okrem genetických vlastností smreka danej klimaticko - geografickej oblasti je to pravdepodobne aj dôsledok rozpojenej štruktúry porastov.

Celková priemerná hodnota korunovosti hornej porastovej vrstvy je 73±1,2%, na základe čoho sa porasty hodnotia ako veľmi stabilné, stredná vrstva má korunovosť 70±1,8% (stredne stabilný porast). Obdobne ako u defoliácie bol na Babej hore v hornej vrstve zistený signifikantný vplyv nadmorskej výšky na korunovosť (IS 99.9%) a v strednej vplyv vývojového štádia (IS 99%). Všeobecne rastie korunovosť s nadmorskou výškou, čo sa vzhľadom na vývoj štruktúry porastov v závislosti od nadmorskej výšky u prirodzených lesných ekosystémov (VORČÁK 2005) očakávalo. Najvyššie hodnoty dosahuje korunovosť v štádiu dorastania u obidvoch porastových vrstiev. V hornej porastovej vrstve v štádiu dorastania dosahuje v 4. v.k. hodnotu 82±2,3%, v strednej porastovej vrstve 81±0,8%, čo predstavuje vysoko stabilné porasty, teda najvyšší stupeň stability (obr. 2). Absolútne najvyššia hodnota korunovosti 92% bola na ploche č. 8, ktorá sa nachádzala v štádiu dorastania, v 1. v.k. v strednej porastovej vrstve. Najnižšia priemerná hodnota 54% bola v štádiu rozpadu v 1.v.k. v strednej porastovej vrstve. Na základe porovnania našich výsledkov s výsledkami autorov MORAVČÍK et al. (2002) môžeme konštatovať, že z hľadiska korunovosti patria porasty na Babej hore v zmysle agregovaných stupňov prirodzenosti do stupňa 1, ktorý predstavuje pralesy. Toto zaradenie je potvrdením skutočného stavu porastov v danej lokalite aj na základe štrukturálnych indexov (MERGANIČ *et al.* 2003, VORČÁK 2005).



Obr. 2 Priebeh priemerných hodnôt korunovosti hornej vrstvy stromov v rámci vývojového štádia a kategórie nadmorskej výšky.
Legenda:
Vývojové štádium: ● Dorastanie, ■ Optimum, ▲ Rozpad;
●, ■, ▲ aritmetický priemer, ⌶ ± 1.96 · stredná chyba

Fig. 2 Average values of crown ratio of the upper tree layer in individual development stages and elevation categories.
Legend:
Development stage: ● Growth, ■ Maturity, ▲ Breakdown;
●, ■, ▲ arithmetic mean, ⌶ ± 1.96 · standard error

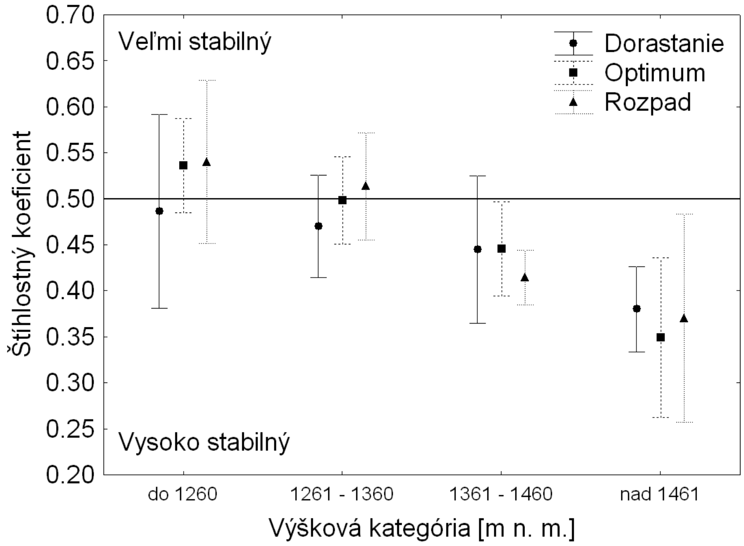
***Štíhlostný koeficient***

Štíhlostný koeficient ako hlavný ukazovateľ statickej stability lesných porastov má význam pri posudzovaní porastov hlavne s ohľadom na mechanicky pôsobiace škodlivé činitele (vietor, sneh a námraza). Celková priemerná hodnota štíhlostného koeficientu je v hornej vrstve 0,45 a strednej vrstve 0,49. Porasty hornej aj strednej vrstvy môžeme charakterizovať ako vysoko stabilné (MÍCHAL et al., 1992). Dôkazom vysokej stability a odolnosti porastov na Babej hore môže byť aj fakt, že extrémne silné vetry v jeseni 2002 a 2004 výrazne nepoškodili porasty v uvedenej oblasti, napriek tomu, že v nižších polohách v hospodárskych lesoch vznikli kalamity v rozsahu desiatok tisíc metrov kubických.

V hornej ako aj strednej vrstve bol na Babej hore zistený signifikantný vplyv nadmorskej výšky na štíhlostný koeficient (IS 99.9%). S rastom nadmorskej výšky výrazne klesá jeho hodnota v obidvoch porastových vrstvách (obr. 3).

Stabilita lesných porastov v jednotlivých vývojových štádiách prírodných a prirodzených lesných ekosystémov je rozdielna. Podľa Stolinu (1982) je najvyššia stabilita v štádiu dorastania, nižšia v optime a najnižšia v rozpade. Štíhlostný koeficient ako hlavný ukazovateľ statickej stability lesných porastov toto potvrdzuje len v 4 v.k. (obr. 3).

Najvyššia hodnota štíhlostného koeficientu 0,72 (stredne stabilný porast) bola zistená na skusnej ploche 17, ktorá sa nachádzala v 2. v.k. v štádiu optima v strednej porastovej vrstve. Najnižšia hodnota 0,29 (vysoko stabilný porast) bola v štádiu optima v 4.v.k. v hornej porastovej vrstve.



Obr. 3 Priebeh priemerných hodnôt štíhlostného koeficienta hornej vrstvy stromov v rámci vývojového štádia a kategórie nadmorskej výšky.
Legenda:
Vývojové štádium: ● Dorastanie, ■ Optimum, ▲ Rozpad;
●, ■, ▲ aritmetický priemer, ⌶ ± 1.96 · stredná chyba

Fig. 3 Average values of h/d ratio of the upper tree layer in individual development stages and elevation categories.
Legend:
Development stage: ● Growth, ■ Maturity, ▲ Breakdown;
●, ■, ▲ arithmetic mean, ⌶ ± 1.96 · standard error

## *Poškodenie stromu*

Poškodenie stromu je súčasťou hodnotení zdravotného stavu porastov a ako pomocná veličina sa využíva aj pri stanovení ekologickej stability lesných porastov. Poškodenie stromov hornej vrstvy na Babej hore dosahuje priemerne 51±3,93%. S rastom nadmorskej výšky na Babej hore signifikantne rastie aj priemerné percento poškodenia stromov (IS 99.9%) a v 4. v.k. dosahuje až 75±9,18%. V 4. v.k. dosahuje poškodenie v hornej porastovej vrstve v štádiu optima dokonca 89±4,47% a v strednej porastovej vrstve 89±4,13%. Predpokladáme, že toto vysoké percento poškodenia je normálnym javom na hornej hranici lesa spôsobené extrémnou klímou. V zimnom období sú vrcholky stromov poškodzované námrazou a snehom za súčinnosti vetra. Treba podotknúť, že poškodenie smreka lesnou zverou v týchto podmienkach je zanedbateľné, prevláda poškodenie spôsobované námrazou a snehom.

## Záver

Horské lesy predstavujú jedinečný a polyfunkčný ekosystém. Sú mimoriadne dôležitým stabilizačným prvkom v krajine, ale zároveň aj rovnako dôležitým regulačným faktorom hydrických, eróznych, protilavínových a ďalších procesov. Stabilita tohto ekosystému má preto značný význam. V predkladanej práci analyzujeme statickú stabilitu lesných porastov v NPR Babia hora v Oravských Beskydách na podklade piatich ukazovateľov: depigmentácie, defoliácie, korunovosti, štíhlostného koeficienta a poškodenia stromov. Z analýzy vyplýva, že s nárastom nadmorskej výšky rastie aj percento depigmentácie, čo je podľa nášho predpokladu spôsobené vplyvom ozónu ($O_3$), ako fotooxidanta. V priemere je však hodnota depigmentácie veľmi malá a porasty v NPR Babia hora je možné zaradiť do stupňa 0 - bez zmeny zafarbenia. Z hodnotenia defoliácie vyplýva, že defoliácia s nadmorskou výškou naopak klesá. Vzhľadom na defoliáciu je možné zaradiť porasty v NPR Babia hora do 1. stupňa. Depigmentácia a defoliácia sa výrazne neprejavila ani v minulosti v období 80. rokov, kedy boli nižšie položené smrekové porasty výrazne poškodené ako dôsledok vysokého imisného tlaku z miestnych i diaľkových zdrojov, hoci je masív vystavený prevládajúcemu prúdeniu vzduchu, ktorý prinášal množstvo polutantov. Veľkosť koruny - korunovosť integruje v sebe dôležitú úlohu vo fyziológii drevín a odolnosti voči komplexu škodlivých činiteľov. Na základe korunovosti hodnotíme porasty v NPR Babia hora ako veľmi stabilné porasty. Hodnota korunovosti hornej porastovej vrstvy signifikantne rastie s nadmorskou výškou. Najvyššie hodnoty dosahuje korunovosť v štádiu dorastania. Typickým javom horských lesov v masíve Babej hory je, že jednotlivé smreky majú hlboké a úzke koruny formované okrem genetických vlastností aj ich vzájomným priestorovým usporiadaním. Hodnoty štíhlostného koeficienta ako hlavného ukazovateľa statickej stability lesných porastov potvrdili vysokú rezistenciu horských lesov v masíve Babej hory. Tento ukazovateľ signifikantne klesá s rastúcou nadmorskou výškou a vo výškovej kategórii nad 1460 m n. m. dosahuje veľmi nízke hodnoty (0.35). Ide o vysoko stabilné porasty, kde takmer nie je možné poškodenie porastov veternou smršťou, pokiaľ koruny nie sú zaťažené námrazou. Z hľadiska poškodenia stromu je hlavný faktor sneh a námraza, ktoré sa najvýraznejšie prejavili vo výškovej kategórii nad 1460 m n. m.

Na základe hodnotených ukazovateľov ekologickej stability môžeme záverom konštatovať, že horské lesy v masíve Babej hory predstavujú vysoko stabilný ekosystém dlhodobo formovaný v danej klimaticko – geografickej oblasti do stavu vnútornej homeostázy. Táto zabezpečuje trvalú stabilitu v rôznych extrémnych situáciách. Významnú úlohu pre tento smrekový ekosystém hrá aj surová klíma, hlavne teplota a zrážky. Tieto faktory priaznivo pôsobia proti šíreniu biotických škodlivých činiteľov (hmyzu) a výrazne vplývajú na štruktúru a rastové pomery porastov.

## Poďakovanie

Ďakujeme Ministerstvu životného prostredia Slovenskej republiky za finančnú podporu poskytnutú pre riešenie tejto práce.

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Eur J Forest Res (2004) 123: 75–85
DOI 10.1007/s10342-004-0013-y

ORIGINAL PAPER

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# Relations between selected geomorphology features and tree species diversity of forest ecosystems and interpolation on a regional level

Received: 8 July 2003 / Accepted: 3 December 2003 / Published online: 3 April 2004

**Abstract** The presented paper analyses the relations between four features of geomorphology, i.e. aspect, slope, elevation and type of terrain, and tree layer diversity of forest ecosystems. The forest stand diversity is quantified by nine species diversity indices (*N0, R1, R2, H′*, N1, N2, E1, E3, E5). The data used in this study come from the regional forest inventory of the Forest School Enterprise, Technical University Zvolen, Slovakia. Within this inventory, 26 permanent tracts were established in the systematic grid of 2×2 km, whereby a total of 120 sample plots were created on which 1,728 trees were measured. Analyses showed that species diversity is closely related only to elevation. However, the relationships of the individual species diversity indices to elevation are loose since the correlation coefficients do not exceed values of 0.30–0.40. The second part of this work presents spatial interpolation of species diversity degrees on a regional level using geostatistical methods. For this regionalisation, we used an indicator kriging. The final kriging map was found to be a suitable tool for the interpretation of tree species diversity over the investigated area.

**Keywords** Geomorphology · Tree species diversity · Diversity indices · Geostatistics · Indicator kriging

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## Introduction

Since the second half of 20th century, changes in biological diversity of natural ecosystems have become a global problem due to extensive human activities. The year 1992, when the “Convention on Biological Diversity” was approved by the United Nations Conference on Environment and Development in Rio de Janeiro, can be considered as pivotal in this field.

In the Convention on Biological Diversity (Article 2), the term biological diversity is defined as “the variability among living organisms from all sources including inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part”. This term, as described in the convention, thus covers not only the diversity within species and between species, but also the diversity of ecosystems (Sibl et al. 1996). The trilogy of the standard components included in the definition, i.e. diversity on genetic, species and ecosystem levels, has become a conventional definition of biodiversity (Hunter 1999).

As the above definition implies, species diversity represents one of the important biodiversity components. Diversity within a certain space and time is determined by the combination of abiotic constraints, biotic interactions, and disturbances (Frelich et al. 1998; Spies and Turnier 1999). Abiotic factors, such as elevation, slope, aspect, soil texture, climate etc., specify the conditions of physical environment and thus the primary species distribution. These relationships were already observed and studied in the 19th century (Hansen and Rotella 1999). Although modern ecologists have focused mainly on other influencing factors, e.g. natural disturbances, the influence of abiotic conditions on species diversity has recently begun to gain the attention of researchers (Burns 1995; Rosenzweig 1995; Austin et al. 1996; Ohmann and Spies 1998; Hansen and Rotella 1999). However, most of these works analyse environmental factors only with regard to the number of tree species, representing just one part of species diversity.

In the presented work, we analyse the relations between four geomorphology features (aspect, slope, elevation and type of terrain) and tree species diversity of forest ecosystems. Species diversity is here quantified by the index methods most frequently used for its assessment. Possible detected relationships could not only extend the current knowledge in this field, but could also be used for modelling tree species diversity on regional and larger scales. The regionalisation of tree species diversity over the whole surveyed area using geostatistical methods is presented in the second part of the paper. This interpolation is based on the output of the model BIODIVERSS (Merganič 2001; Merganič and Šmelko, submitted), a complex species diversity quantification method.

## Materials and methods

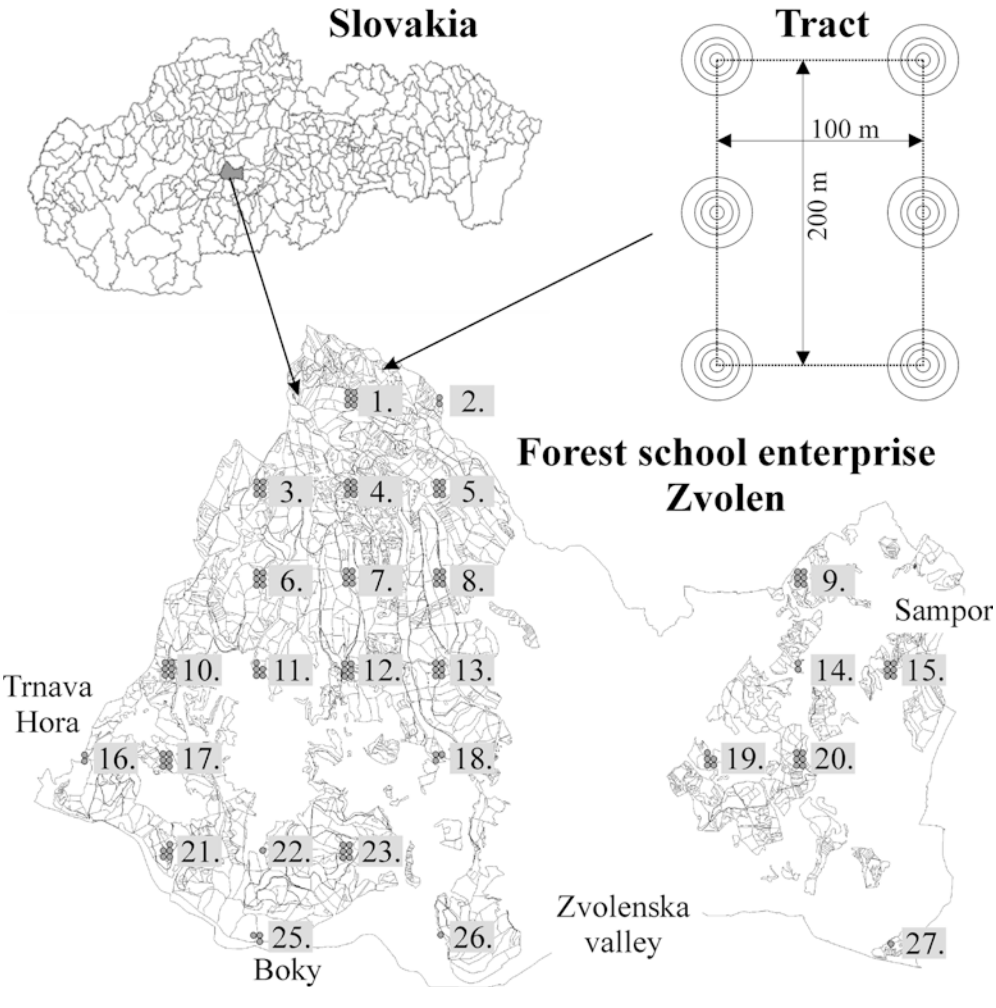
### Data sampling

Data were collected within the regional forest inventory (RFI 98) of the Forest School Enterprise (FSE) of the Technical University Zvolen (Slovak Republic) in the year 1998. This enterprise covers a total area of 8,043.59 ha, of which 7,743.74 ha is covered by forests. The forested area comprises 32.18% managed forests, 6.91% protected forests and the rest are forests for special purposes (e.g. recreation, research, hunting etc.). Regarding the species composition of forest stands, deciduous species (mainly common beech, oak and hornbeam) dominate (72.9%) over coniferous ones, represented by Norway spruce, silver fir and Scots pine. In the forest communities, mixed spruce-fir-beech, pure beech and mixed oak-beech types prevail. The most abundant plant sociological groups within the area of FSE, classified according to the classification schemes of Zlatník (1976) and Hančinský (1977), are Querceto-fagetum, Fageto-quercetum and Fagetum pauper.

The sampling design applied in the RFI 98 was a quadratic (2×2 km) net of tracts, each 100×200 m in size, established in FSE during research on a large-scale forest inventory for Slovak conditions (Šmelko et al. 1986, 1988). Each tract is composed of six sample plots with 100 m distance between neighbouring plots in the tract (Fig. 1). Thus, the design can be characterised as a systematic, one-level grouping sampling.

The advantage of using tracts within the forest inventory is their ease of handling and that, at the end, the working group returns to its starting point.

**Fig. 1** Forest school enterprise—its general location and location of analysed tracts



Although higher accuracy could have been achieved by applying individual sample plots distributed around the whole survey area, this kind of inventory would, on the other hand, raise the inventory costs substantially.

The individual plots within each tract are composed of five concentric circles, on which groups of trees with a certain size were measured according to the following scheme (Šmelko 2000):

| | | |
|---|---|---|
| 1. | tree height < 1.3 m | 1 m$^2$(square) |
| 2. | tree height > 1.3 m and dbh < 8 cm | r = 2.52 m (20 m$^2$) |
| 3. | 8.1–16.0 cm (dbh) | r = 5.64 m (100 m$^2$) |
| 4. | 16.1–28.0 cm | r = 7.98 m (200 m$^2$) |
| 5. | above 28.1 cm | r = 12.62 m (500 m$^2$) |

Specification of the individual concentric plots was based on current knowledge, experience, and a simulation experiment, within which the forest stand conditions of Slovakia (Halaj 1957) were taken into account. Analyses confirmed that the suggested design of concentric circle sample plots and the algorithm of elaboration of results is a suitable new approach for sampling forest inventory and surveys on a larger scale. The plots have good biometric characteristics, react sensitively to the structure of forest stands, and are equally accurate but economically more efficient than classic constant circles (Šmelko 2000).

Within the regional forest inventory, 27 tracts were established with a total of 121 sample plots. In the presented study, 26 tracts and 120 plots were used, whereby only data from the last three concentric circles, representing the tree layer of forest ecosystems, were taken into account. In total 1,728 trees with diameters at breast height over 8 cm were measured on 120 selected plots.

From the whole RFI 98 information spectrum, consisting of 55 elements which were assessed on every permanent sample plot, the following four geomorphology features were selected for this analysis:

- aspect, defined by the cardinal points as N, NE, E, SE, S, SW, W, NW
- slope gradient measured in degrees with 1°accuracy
- elevation, taken from the forest stand map (1:10,000) with 10 m accuracy
- type of terrain, according to Zingg and Bachofen (1988), divided into five categories defined as follows:

1. flat ground: no slope or slope gradient up to 5°—plateau, tableland, terrace, base of valley;
2. summit, upper slope: convex form, water outflow prevails—hill, upper slope, ridge, rib, earth bank;
3. middle slope: water outflow and inflow are balanced—middle slope, steep incline, etc.;
4. foothill, depression: concave form, water inflow prevails—foothill, gutter duct, ditch;
5. indefinite: impossible to classify into previous four categories, terrain with frequent changes of slope and aspect.

Based on the data from RFI 98, mean values or proportions (for continuous and discrete variables, respectively) and their statistical errors were calculated in order to describe our area of interest. These parameters were calculated from the estimation models for mean values of a particular variable in case there was an unequal number of sample plots in a sampling group or an unequal number of trees on a sample plot (Cochran 1977; Šmelko 1997; Merganič 2001).

From the results, we can state that within the FSE region the eastern aspect and the terrain type described as "middle slope" prevail (Tables 1 and 2). The FSE can be characterised as an area with an average slope gradient 13.3 ± 0.9° (coefficient of variation 34.9%, relative mean standard error 6.8%). The greatest slope gradient was recorded on tracts 1, 16 and 25, situated in the borderland of the FSE. Towards the southeast, the slope gradient decreases. The lowest gradient was observed in the area next to Zvolen valley (around tracts 23 and 27). This corresponds well with the terrain type "flat ground" that was identified on tracts 18, 23 and 27. Relatively high slope gradients were also observed on tracts 14 and 20, in the eastern part of the locality "Sampor".

The average elevation of the FSE is 522.5 ± 31.4 m above sea level (coefficient of variation 30.6%, relative mean standard error 6.0%). The highest elevation is in the north, from where it decreases towards southeast

**Table 1** Proportion of aspects on the area of FSE determined from RFI 98 data (*n* = 120)

| Aspect | Proportion (%) | Standard error of proportion (%) | Relative standard error of proportion (%%) |
|---|---|---|---|
| N | 8.3 | 2.4 | 29.1 |
| NE | 15.0 | 5.2 | 34.5 |
| E | 21.7 | 6.6 | 30.3 |
| SE | 15.8 | 5.7 | 36.0 |
| S | 8.3 | 2.4 | 29.1 |
| SW | 6.7 | 2.9 | 43.6 |
| W | 17.5 | 5.1 | 28.9 |
| NW | 6.7 | 3.1 | 47.1 |

**Table 2** Proportion of existing terrain types over the area of FSE Zvolen estimated from RFI 98 data (*n* = 120)

| Type of terrain | | Proportion (%) | Standard error of proportion (%) | Relative standard error of proportion (%%) |
|---|---|---|---|---|
| Flat ground | 1 | 6.7 | 3.2 | 48.3 |
| Summit, upper slope | 2 | 32.5 | 6.6 | 20.2 |
| Middle slope | 3 | 39.2 | 6.3 | 16.0 |
| Foothill, depression | 4 | 17.5 | 4.1 | 23.3 |
| Indefinite | 5 | 4.2 | 2.4 | 58.7 |

into the central part of FSE. The lowest point of the FSE is situated at an elevation of 280 m above sea level near the village Jalná. The highest point is the peak Laurín at the end of the Sielnická valley (1,150 m above sea level).

### Quantification of tree species diversity

Tree species diversity was quantified by diversity indices. From the great number of existing indices, we selected those that are considered by the majority of authors (Ludwig and Reynolds 1988; Krebs 1989) to be the most suitable for numerical assessment of diversity in examined populations. These indices can be divided into three groups:

| | | |
|---|---|---|
| Indices of species richness describing community diversity on the base of number of species | | |
| $N0 = S$ | (Hill 1973) | (1) |
| $R1 = (S - 1)/\ln(N)$ | (Margalef 1958) | (2) |
| $R2 = S/\sqrt{N}$ | (Menhinick 1964) | (3) |
| Indices of species evenness | | |
| $E1 = H'/\ln(S)$ | (Pielou 1975, 1977) | (4) |
| $E3 = \left(e^{H'} - 1\right)/(S - 1)$ | (Heip 1974) | (5) |
| $E5 = ((1/\lambda) - 1)/\left(e^{H'} - 1\right)$ | (Hill 1973) | (6) |
| Indices of species heterogeneity combining species richness and evenness | | |
| $H' = -\sum_{i=1}^{S} (w_i \ln(w_i))$ | (Shannon and Weaver 1949) | (7) |
| $N1 = e^{H'}$ | (Hill 1973) | (8) |
| $N2 = 1/\lambda$ | (Hill 1973) | (9) |
| where $S$ number of species, $N$ number of individuals, $w_i$ relative abundance of tree species $i$ | | |
| $\lambda - \sum_{i=1}^{S} w_i^2$ | (Simpson 1949) | (10) |

Species heterogeneity and species evenness indices were calculated using the species proportion determined from the stand basal area in order to account for the tree size by means of diameter (Merganič 2001).

Before the final calculations of indices could be performed, the input variables had to be transformed since each primary sample unit is a sample plot composed of concentric circles with stand and tree characteristics $x_i$ representing a different area $a_i$. According to the latest findings (Šmelko 1997; Saborowski and Šmelko 1998; Šmelko and Saborowski 1999), these do not have the same probability of being selected. Therefore, it is necessary to recalculate all forest stand variables (e.g. number of trees, stand basal area, stand volume) on a per ha basis:

$$V_i\ \mathrm{ha}^{-1} = \frac{V_i}{a_i} \quad N_i\ \mathrm{ha}^{-1} = \frac{N_i}{a_i} \quad \text{generally} \quad X_i = \frac{x_i}{a_i} \tag{11}$$

In this way, the characteristics can be standardised, which consequently enables their simple comparison. The overall value of the particular stand characteristic $X$ on a sample plot is a sum of its values $X_i$ on all concentric circles. The tree characteristics can be calculated similarly, with the difference that one tree taken as a representative of a particular feature, e.g. defoliation percentage, represents a total of $1/a_i$ trees. At this point it is necessary to note that the above recalculation does not apply to the number of tree species. This characteristic is, contrary to other stand characteristics, e.g. stand volume, always related to the size of the sample unit on which it was recorded and cannot be standardised on a per hectare basis using the above method.

### Analysis of relationships between geomorphology features and tree species diversity

Mathematic-statistical analysis of the influence of the selected terrain characteristics on species diversity is based on correlation analysis and analysis of variance. Correlation analysis included the significance *F*-test of the coefficient of determination (multiple R squared for linear or quadratic form according to the regression) and was applied to continuous surveyed geomorphology features, i.e. elevation and slope. The relationship between discrete terrain characteristics, i.e. aspect and type of terrain, and species diversity indices was tested with the analysis of variance.

### Regionalisation of species diversity degree of the forest ecosystem tree layer

Each of the diversity indices cited above describes only one aspect of species diversity. An integrated method for species diversity quantification is implemented in the modelling program BIODIVERSS (Merganič 2001; Merganič and Šmelko, submitted). The model was constructed using the data from the forest stands of a relatively large size (2.98–6.48 ha), in which the position and the basic dendrometric characteristics of each tree are known. For the construction of the model, 865 optimally sized sample plots (the size of the plot depends on the forest stand density, whereby the criterion is that each plot should encompass approximately 20 trees) were used. The model is based on the assumption that if on a small plot high species diversity is observed, then the species diversity of the whole forest stand will be with a certain probability also high.

For the estimation of tree species diversity, the model uses a predictive discriminant procedure (Cooley and Lohnes 1962; Huberty 1994). The independent variables that enter the model as predictors are five diversity indices *R1, R2,* $\lambda$, H′and *E1*. Together, they represent all three groups of diversity, i.e. species richness, evenness and heterogeneity. Based on the values of the indices, the model BIODIVERSS classifies the examined stand into one of the four degrees of species diversity:

| | |
|---|---|
| 1 (Low) | Small number of occurring species (1–2) or medium number of species (3–4) with a very low balance in species composition and their uneven horizontal distribution over the forest stand area |
| 2 (Medium)<br>3 (High) | Transition degrees between 1 and 4 |
| 4 (Very high) | Large number of occurring species (above 4), whereby they are balanced in species composition and have even distribution over the forest stand area |

For geostatistical analysis, the species diversity degree was determined for one tract using the average values of indices calculated from all of its sample plots. This means that each tract was regarded as one spatial point characterised by one species diversity degree. In the next step, the geostatistical analysis of species diversity degrees was performed using the indicator kriging. This approach was first proposed by Journel (1983) and is performed on binary-transformed data. Thus, it can also be applied to categorical variables (Deutsch and Journel 1998; Marinoni 2002). In the first step of this analysis, the so-called indicator cutoff values are defined. In our case, these cutoff values are species diversity degrees, i.e. values 1, 2, 3 and 4. Secondly, a binary transformation of the interpolating categorical variable $z(x)$ is done for all indicator cutoff values separately, as follows:

$$i(x,\ \mathit{species\ diversity\ degree}) = \begin{cases} 1, & \mathit{if}\ z(x) = \mathit{species\ diversity\ degree} \\ 0, & \mathit{otherwise} \end{cases}$$

Spatial variability of a particular species diversity degree is then described by the indicator semi-variogram, which is constructed from the sample indicator semi-variogram calculated from the set of point data, in our case from the binary-transformed values of species diversity degree on individual tracts. The indicator semi-variogram is created by fitting a theoretical function to a sample indicator semi-variogram. For all species diversity degrees the circular semi-variogram fitted the data best. Its mathematical formulation is as follows (Pebesma 2000):

$$\gamma(h) = C_0 + C \cdot \left( \frac{2 \cdot h}{\pi \cdot b} \cdot \sqrt{1 - \left(\frac{h}{b}\right)^2} + \frac{2}{\pi} \cdot \mathit{arcsin}\left(\frac{h}{b}\right) \right) \tag{12}$$

where $h$ is the distance, $b$ the range, $C_0$ the nugget effect, and $C$ the sill.

As a result of the described procedure, we obtain as many variograms as cutoff values (categories of the interpolating variable), let us assume it to be a value $k$. Applying each of these variograms to indicator kriging analysis, we get $k$ layers of the interpolated probabilities for $k$ degrees of species diversity. The final kriging map and thus also the final values of the interpolating variable, i.e. species diversity degree for each interpolated point $[x,y]$ are estimated from the comparison of the $k$ probabilities from all obtained layers.

## Results

### Influence of selected geomorphology features on species diversity

The relation between the surveyed geomorphology factors and tree species diversity was tested with the correlation analysis and analysis of variance as described in a previous section (Analysis of relationships between geomorphology features and tree species diversity). Their usage is, in general, restricted to an assumption that the analysed data are not autocorrelated. Due to the applied sampling design, i.e. grouping sampling, we can assume that the values of species diversity indices will be spatially autocorrelated. Therefore, in the first step, the magnitude and the spatial range of this autocorrelation were assessed. The results showed that spatial autocorrelation in our data exists. Nevertheless, it is significant only up to a distance of 100 m. Moreover, the magnitude (power) of the autocorrelation is not large and fluctuates between $R = 0.07$ to 0.37.

Because of the detected spatial autocorrelation, each statistical analysis of the relationship between the selected geomorphology feature and species diversity started with the analysis of autocorrelation between the residuals by Wald's test (Meloun and Militký 1998). This test showed that the existing autocorrelation between the residuals is for most species diversity indices not significant. The significance of their autocorrelation was detected only for indices *R1* and *R2*, which was taken into account when interpreting the results.

The actual analysis of the influence of geomorphology on species diversity showed that elevation was the most important factor and seemed to affect all diversity indices. All of the nine coefficients of determination differed from zero with a statistical significance of $\alpha = 0.05$, eight coefficients even with $\alpha = 0.01$ (Table 3). Applying a Bonferroni-Holm multiple test procedure (Holm 1977, 1979, described in detail in Quednau 1992), we could reject the hypothesis of "$R^2 = 0$" for all of the nine indices, with a multiple error probability of $\alpha = 0.05$ and for three of them with $\alpha = 0.01$ (Table 3, right side). As shown in Fig. 2, the diversity first decreases with increasing elevation up to approximately 650 m above sea level, there it reaches its minimum, and then it rises again.

The factor "type of terrain" has a statistically significant influence only on the species richness indices, i.e. *N0*, *R1* and *R2*. The Bonferroni-Holm test procedure did not reveal a single significant difference at a multiple level. A more detailed analysis of the influence of this factor revealed that the highest species richness was found at locations characterised as plateau or sites with a very variable terrain type, classified as an indefinite

**Table 3** Linear-quadratic regression analysis of the influence of elevation on the values of species diversity indices, significance in Bonferroni-Holm procedure: + 0.05, + + 0.01 ($n = 120$)

| Diversity indices | $R$ | $R^2$ | $F$ | $rank(F)$ | $p(F)$ | 0.05/(10-rank) | 0.01/(10-rank) |
|---|---|---|---|---|---|---|---|
| $N_0$ | 0.34 | 0.12 | $7.67^b$ | (1) | 0.000 741 + + | 0.005 556 | 0.001 111 |
| $R1$ | 0.33 | 0.11 | $7.12^b$ | (3) | 0.001 208 + + | 0.007 143 | 0.001 429 |
| $R2$ | 0.23 | 0.05 | $3.24^a$ | (9) | 0.042 704 + | 0.050 000 | |
| $H'$ | 0.34 | 0.12 | $7.65^b$ | (2) | 0.000 754 + + | 0.006 250 | 0.001 250 |
| $N_1$ | 0.31 | 0.10 | $6.30^b$ | (4) | 0.002 520 + | 0.008 333 | 0.001 667 (!) |
| $N_2$ | 0.29 | 0.08 | $5.24^b$ | (6) | 0.006 615 + | 0.012 500 | |
| $E1$ | 0.30 | 0.09 | $5.86^b$ | (5) | 0.003 655 + | 0.010 000 | |
| $E3$ | 0.27 | 0.08 | $4.76^a$ | (8) | 0.010 293 + | 0.025 000 | |
| $E5$ | 0.28 | 0.08 | $5.16^a$ | (7) | 0.007 119 + | 0.016 667 | |

$^aF_{0.05(2,117)} = 3.072$
$^bF_{0.01(2,117)} = 4.791$

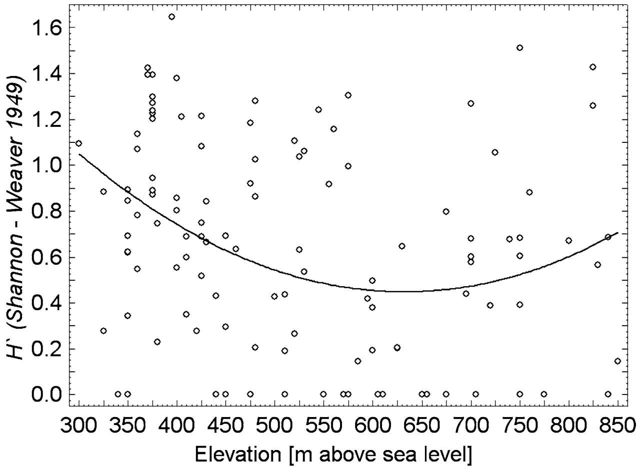


**Fig. 2** Relation between index $H'$ and elevation

type. In contrast, the lowest species richness was observed on upper and middle slopes.

Although the sample regression curves of the diversity indices versus the factor slope had a similar shape as for elevation, with the lowest diversity on the slopes with the gradient 17° (not shown), none of the relations were statistically significant. Similarly, aspect did not appear to be significantly related to any of the examined species diversity indices.

The behaviour of other indices over the range of the independent factor is not shown since their performance is very similar to those presented.

### Species diversity in the tree layer of forest ecosystem over the area of FSE Zvolen

In the second part of this paper, we present an overall evaluation of tree species diversity over the area of FSE Zvolen using the data collected within RFI 98. Figure 3 shows the characteristics of tree layer species diversity on individual tracts described by the selected species diversity indices *N0*, *R1*, *H′*and *E1*.

The degree of species diversity was estimated using the model BIODIVERSS that was constructed using the data from optimally sized sample plots. As can be seen from the data description, within RFI 98 a different type of sample units was applied, namely concentric circle

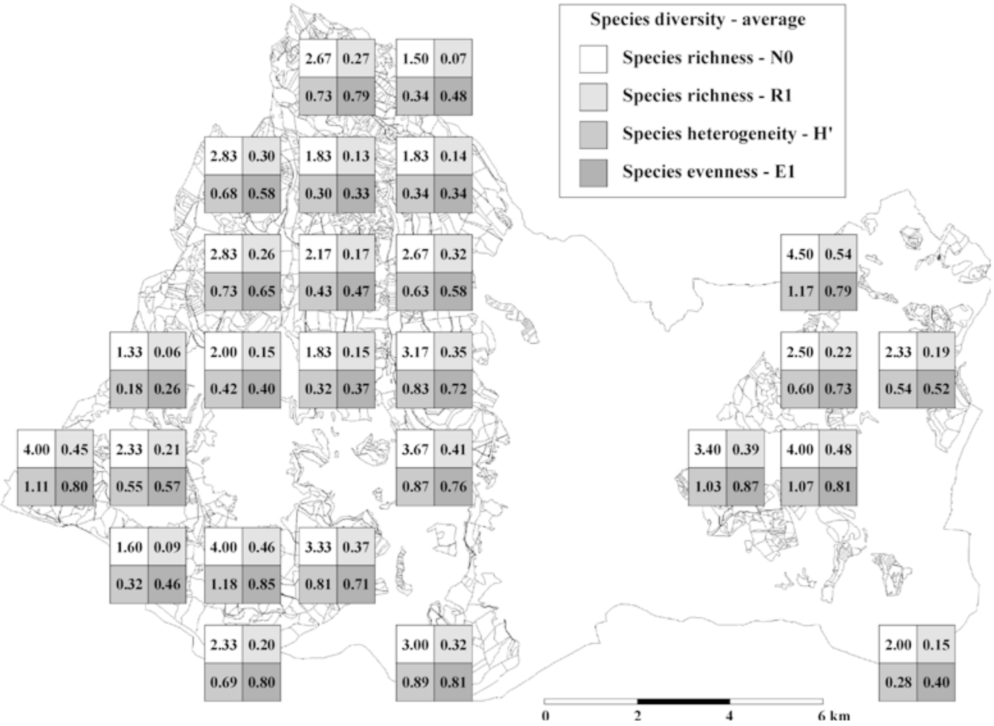


**Fig. 3** Characteristics of tree layer species diversity on individual tracts described by the species diversity indices *N0*, *R1*, *H′* and *E1*

sample plots. Although these concentric plots represent one type of optimally large sample plots because they react well to forest stand structure and at the same time optimise both inventory accuracy and costs, their suitability for diversity quantification of forest ecosystems is to a certain extent questionable, since in some forest stands of a particular structure the list of tree species recorded on such plots can be incomplete. The simulations on the parameterisation data set, however, showed that the differences in the estimated degree of species diversity between concentric and optimally sized sample plots are not significant. Moreover, when the degree of species diversity is estimated from the average values of species diversity indices from all sample plots within one tract, as done in this study, the existing differences are further reduced.

The spatial variability of the binary-transformed degrees of species diversity was assessed by the circle indicator semi-variograms. Figure 4 presents the shape of the applied indicator semi-variograms. From this figure we can see that only three semi-variograms were constructed, although model BIODIVERSS distinguishes four degrees of species diversity. This is due to the fact that within the simulated data no tract was assigned the degree 4. Therefore, it was not possible to create a semi-variogram for this degree. In all cases, modelled semi-variance is at the beginning very small and grows with the increasing distance between the pairs of sample points until the culmination point (sill) at the distance 2.275 km for species diversity degree 1, 2.6 km for degree 2 and 2.86 km for degree 3. If the distance between the sample points exceeds this value, the modelled semi-variance will not change, it remains constant at the sill value of 0.09, 0.256 and 0.216 for species diversity degrees 1, 2 and 3, respectively.

Using the three circle indicator semi-variograms within the kriging procedure, the species diversity degrees, estimated with the model BIODIVERSS, were interpolated over the whole area of FSE Zvolen as shown in Fig. 5. From this chart, as well as from Table 4, it can be concluded that the predominant part, i.e. 44.2 ± 4.8%, of the FSE area belongs to the second species diversity degree. Alternatively, the FSE can be characterised with an average species diversity degree of 2.23 ± 0.14. Moreover, from Fig. 5 it can also be seen that the areas with a high species diversity degree, i.e. degree 3, are situated in the national nature reserve Boky, at the edge of the "Sampor" location, around tract 16 (local name Trnavá Hora) as well as in the surroundings of tracts 8 and 13. On the other hand, the areas around tracts 5 and 10 have the lowest diversity degree of 1.

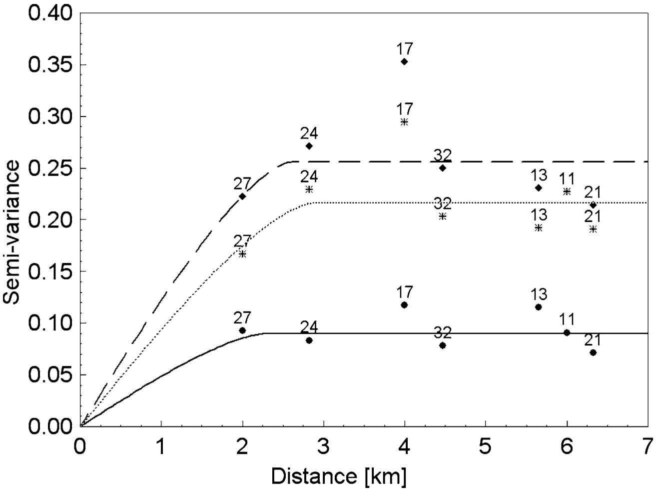


**Fig. 4** Indicator semi-variogram of species diversity degree. Real semi-variance: ● degree 1, ◆ degree 2, ⁎ degree 3; model semi-variance: — degree 1, – – degree 2, ······ degree 3

Figure 6 presents the probability of the estimated species diversity degree over the whole investigated area. As expected, the highest probability is observed next to the places with measurements. As the distance from these points increases, the probability decreases.

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## Discussion

The current spatial structure of tree species diversity on the FSE is the result of several factors and their interactions. Abiotic factors, such as elevation, slope, aspect, soil type etc., create together a unique complex of environmental conditions specifying forest communities (Spies and Turner 1999). The relation between elevation and species diversity is generally accepted and was documented by several authors, not only for tree species but also regarding the diversity of plants and animals (Rosenzweig 1995). In our analysis, elevation was found to have a significant influence on tree species diversity (Table 3) with the lowest values of tree species diversity at middle elevations. This performance can most probably be explained by the fact that in the Slovak Republic at about 600 m above sea level, beech has its optimum (i.e. climatic conditions at this elevation, such as temperature and precipitation, are the most suitable for this species),so at these altitudes beech is so vital and competitive that other species become rare.

Apart from environmental abiotic conditions, species diversity is also influenced by other factors, e.g. position of the forests within the landscape. As the kriging map (Fig. 5) shows, areas with the highest tree species diversity adjoin open agricultural land and can be hence characterised as relatively highly fragmented forests. This is partly in contradiction to Haila (1999), who presents an opposite reaction to forest fragmentation. However, the quoted work assessed species diversity using the number of species, which is known to be strongly correlated to the size of the evaluated area, whereas in our work we use standardised data always representing the same size. Moreover, it is also possible that the influence of elevation, and thus of improved climatic conditions, is stronger than the effect of fragmentation, since the area with high species diversity is located at lower altitudes.

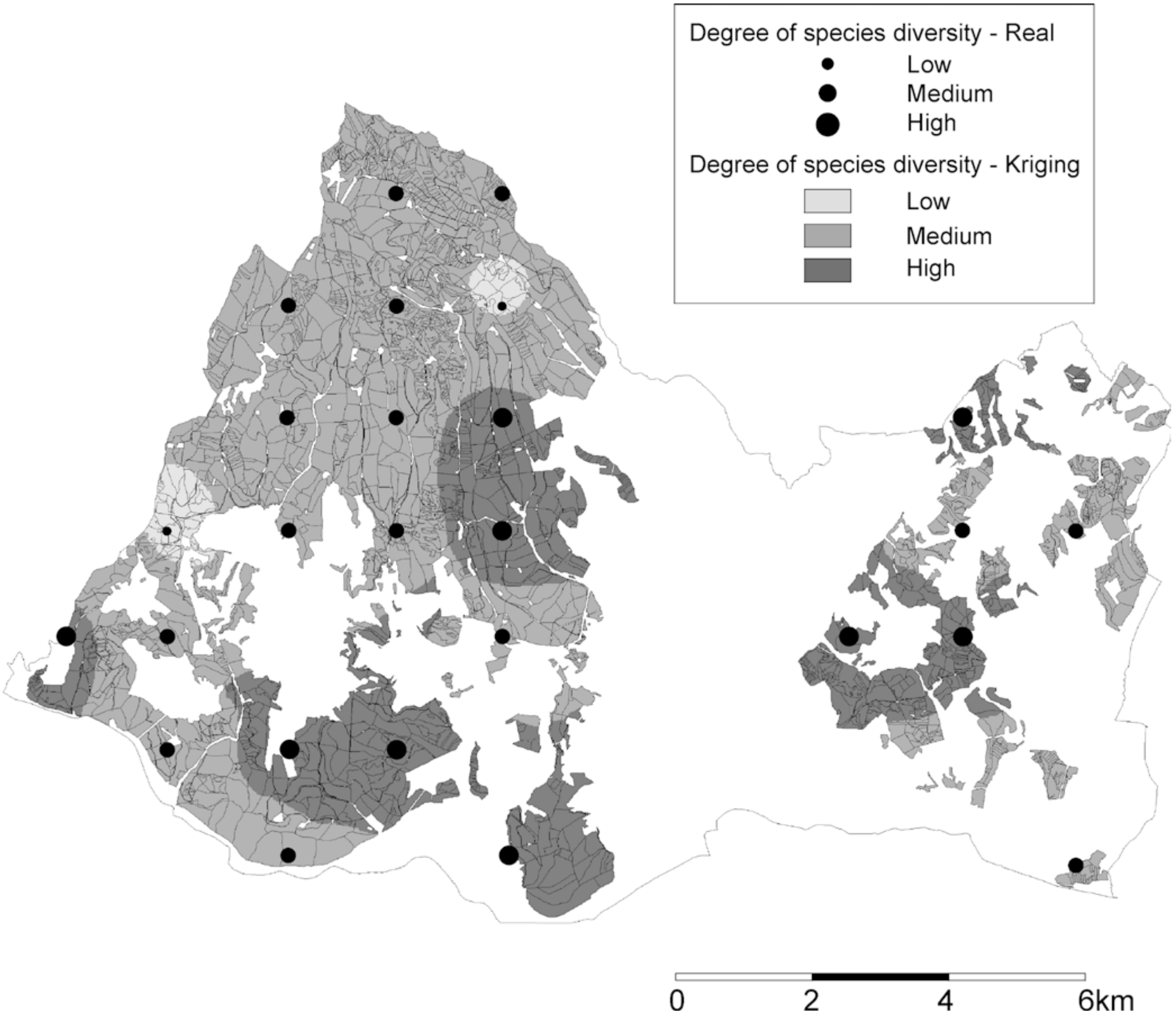

**Fig. 5** Species diversity of the tree layer over the area of FSE Zvolen

High species diversity is often connected to more complex vertical structure (Brokaw and Lent 1999). Although in our analysis we have not tested this fact explicitly, the results indicate the positive effect of vertical structure, which is usually more complex on the forest edges. Thus, the higher tree species diversity of the fragmented forests can also be explained by this factor.

Today, direct human impact can affect species composition of the forest stands to a great extent (Palik and Engstrom 1999). Forest management can either decrease species diversity, if the climax tree species are favoured, or increase diversity, for example, by introducing other tree species. Since within the surveyed region of the FSE protected forests comprise only 7% of the area, we also examined the possible relations between tree species diversity and forest management. However, the comparison of the final kriging map with the map of the current management status obtained from the forest

**Table 4** Proportion of degrees of species diversity over the area of FSE Zvolen estimated from RFI 98 data ($n = 120$)

| Degree of species diversity | | Proportion (%) | Standard error of proportion (%) | Relative standard error of proportion (%%) |
|---|---|---|---|---|
| Low | 1 | 20.8 | 5.0 | 23.8 |
| Medium | 2 | 44.2 | 4.8 | 10.9 |
| High | 3 | 28.3 | 4.4 | 15.5 |
| Very high | 4 | 6.7 | 2.2 | 32.8 |

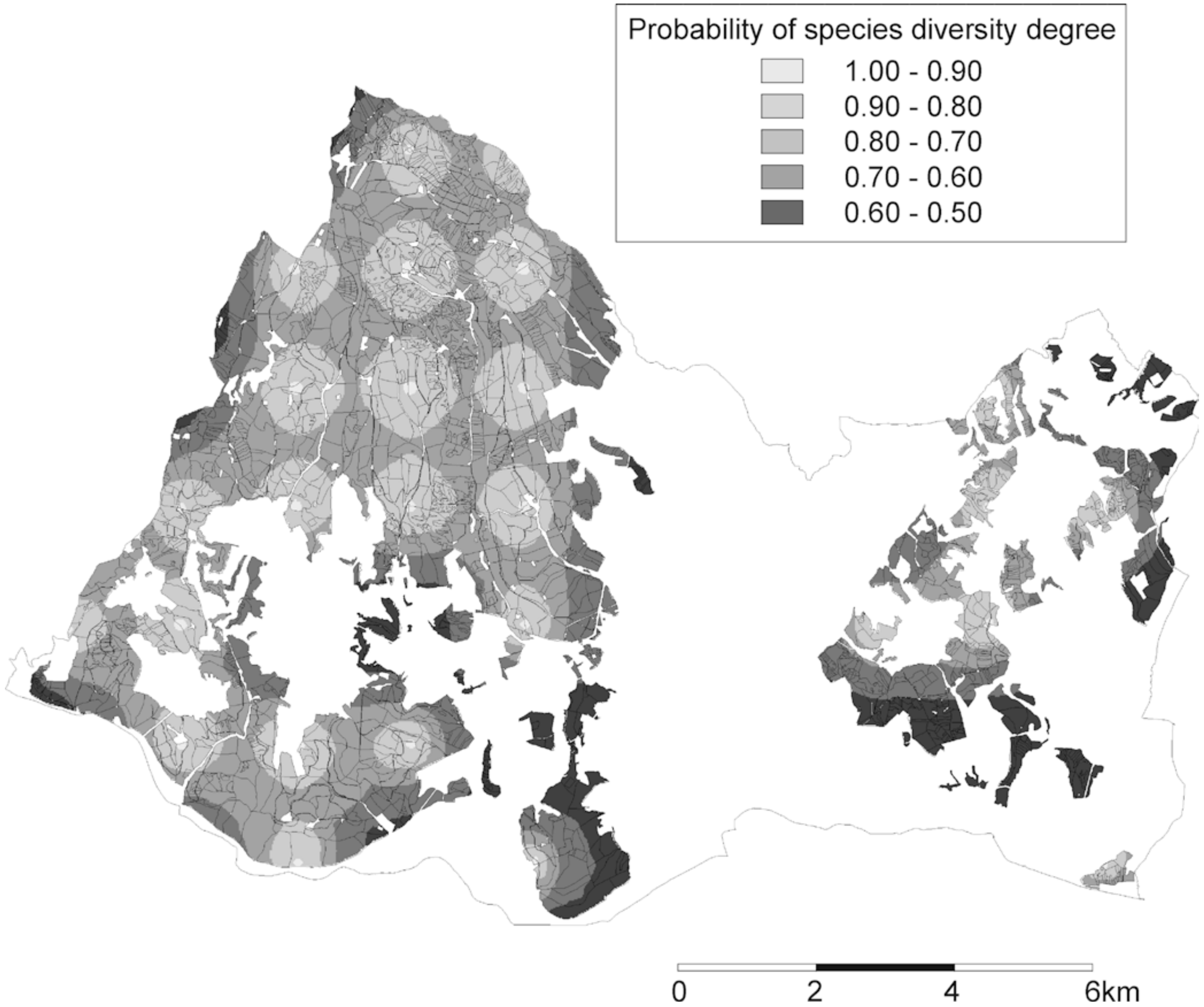

**Fig. 6** Probability of species diversity degree over the area of FSE Zvolen

information system LesHIS (Fabrika 2001) did not reveal any significant connection between species diversity and management.

## Conclusion

The analysis of the influence of the selected features of geomorphology (aspect, slope, elevation and type of terrain) on tree layer diversity quantified by nine species diversity indices (*N0, R1, R2, H′*, N1, N2, E1, E3, E5) showed that only elevation is significantly related to tree species diversity. Although the relations are very loose since the coefficients of correlation do not exceed 0.3–0.4, dependence of species diversity on elevation was also found in other publications and is generally accepted in ecological literature.

Spatial interpolation of species diversity degrees estimated with the model BIODIVERSS using geostatistical methods provides us with valuable output on a regional scale. As with all statistical methods, the model output represents an estimation of the actual state and does not necessarily have to coincide exactly with reality. Nevertheless, it allows us to make rapid spatial analyses, which in connection with GIS methods enables a user to test the influence of different factors on e.g. species diversity. Based on the mentioned instruments, we were able to assess the relation of tree species diversity to the type of management and forest fragmentation.

The presented work uses the data collected within the regional forest inventory. This kind of analysis could also have been performed using the data from forest management plans. Their utilisation would, however, cause several problems for the quantification of species diversity on a regional scale because:

- within forest taxation only dominant tree species are recorded, which means that information about rare tree species is missing,
- species diversity indices are strongly correlated to size (area) of the evaluated forest stand,
- classic species diversity indices are distance independent, i.e. they do not account for the spatial mixtures of species.

Therefore, in cases as presented in this study, it is advantageous to use data from the forest inventory, which enables the standardisation of evaluation methods. Moreover, quantification of species diversity degree with the model BIODIVERSS solves the problem of the different size of the evaluated object and takes the spatial mixture of species into account.

The map presenting the spatial interpolation of tree species diversity degree can be regarded as a simple tool for a forest manager, useful for the assessment of structural diversity, forest ecosystem stability, in public relations as well as for the implementation of the Convention on Biological Diversity.

**Acknowledgements** This work was performed within the framework of the project VEGA SR 1/7053/20 "Integrated system of sample methods for permanent monitoring of the status of forest ecosystems", the project ITM EU 1902, Implementing tree growth models as forest management tools, and the study visit of J. Merganič at the Technical University Munich financed by DAAD in autumn 2000. We thank two anonymous reviewers for their comments which helped to improve the quality of the presented paper.

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© Triveni Enterprises, Lucknow (India)
ISSN : 0254-8704
http : //www.geocities.com/j_environ_biol/

*J. Environ. Biol.*
**29(3),** 291-298 (2008)
j_environ_biol@yahoo.com

# Tree species diversity and its relationship to stand parameters and geomorphology features in the eastern Black sea region forests of turkey

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(Received: March 31, 2007 ; Revised received: August 08, 2007 ; Accepted: September 05, 2007)

***Abstract:*** *We studied the effects of stand parameters (crown closure, basal area, stand volume, age, mean stand diameter, number of trees, and heterogeneity index) and geomorphology features (elevation, aspect and slope) on tree species diversity in an example of untreated natural mixed forest stands in the eastern black sea region of Turkey. Tree species diversity and basal area heterogeneity in forest ecosystems are quantified using the Shannon-Weaver and Simpson indices. The relationship between tree species diversity, basal area heterogeneity, stand parameters and geomorphology features are examined using regression analysis. Our work revealed that the relationship between tree species diversity and stand parameters is loose with a correlation coefficient between 0.02 and 0.70. The correlation of basal area heterogeneity with stand parameters fluctuated between 0.004 and 0.77 ($R^2$). According to our results, stands with higher tree species diversity are characterised by higher mean stand diameter, number of diameter classes, basal area and lower homogeneity index value. Considering the effect of geomorphology features on tree species or basal area heterogeneity, we found that all investigated relationships are loose with $R^2 < 0.24$. A significant correlation was detected only between tree species diversity and aspect. Future work is required to verify the detected trends in behaviour of tree species diversity if it is to estimate from the usual forest stand parameters and topography characteristics.*

***Key words:*** *Tree species diversity, Mixed stands, Stand structure, Geomorphology*

*PDF of full length paper is available with author* ***(*ramazan@orman.sdu.edu.tr)***

## Introduction

Nature conservation priority was rated, using rarity, species richness, stratification, site age, and area of the habitats (Evrendilek, 2003). Biological diversity is a key issue of nature conservation, and species diversity is one of important components of the biological diversity (Ito, 1997). Forest lands extend over a great number of ecosystems, harboring a rich diversity of species and genes. Thus, within the biodiversity conservation debate top priority was given to forests. The diversity of tree species is fundamental to total forest biodiversity, because trees provide resources and habitats for almost all other forest species (Cannon *et al*., 1998; Pandeya *et al.,* 2007). To study tree species diversity of untreated, natural mixed stands is a key to conserving biodiversity of forest ecosystems.

Biodiversity assessment is often restricted in the red listing of threatened species and clarification of their habitat demands in forest practices and forest management plans. However, good data and appropriate indicators are necessary to assist policy making and monitoring to understand the causes of changes in biodiversity and to better implement protection strategies (Puumalainen *et al*., 2003).

Numerical quantification of biological diversity and/or its elements can be of great value because that kind of evaluation is objective and enables a comparison of current biodiversity status to be made between similar ecosystems. During the last century, a great number of different methods quantifying species diversity were developed (Ludwig and Reynolds, 1988; Patil and Taillie, 1982; Merganic and Smelko, 2004). However, while using any of the proposed measures one has to be aware of the fact that diversity changes in space and time as it is influenced by abiotic and biotic factors, and disturbances (Frelich *et al*., 1998; Nagaraja *et al*., 2005; Misir *et al.,* 2007; Ucler *et al.,* 2007). Parameters affecting plant growth and resource availability, *e.g.* climate, are regarded as primary influencing factors (Terradas *et al*., 2004), while the terrain characteristics, *e.g.* elevation, are considered indirect factors because they themselves have no direct impact on plant growth, but are correlated with the primary factors (Pausas *et al*., 2003; Bhattarai *et al*., 2004). The indirect factors are often used in the analysis when information about the primary factors is not available (Pausas and Saez, 2000). Most often, the relationship of diversity to elevation is investigated (Grytnes and Vetaas, 2002; Bhattarai and Vetaas, 2003; Bachman *et al*., 2004), while the effects of other topographic features are rarely examined (Johnson, 1986; Palmer *et al*., 2000). In addition, most of the published works analyse environmental factors only with regard to species richness, representing just one component of species diversity (Merganic *et al*., 2004).

This research aims to address tree species diversity (expressed by the Shannon-Weaver and Simpson indices) in untreated natural mixed stands of the eastern Black sea region forests of Turkey. This study also aims to clarify the relationship of the calculated tree species and basal area diversity with selected stand

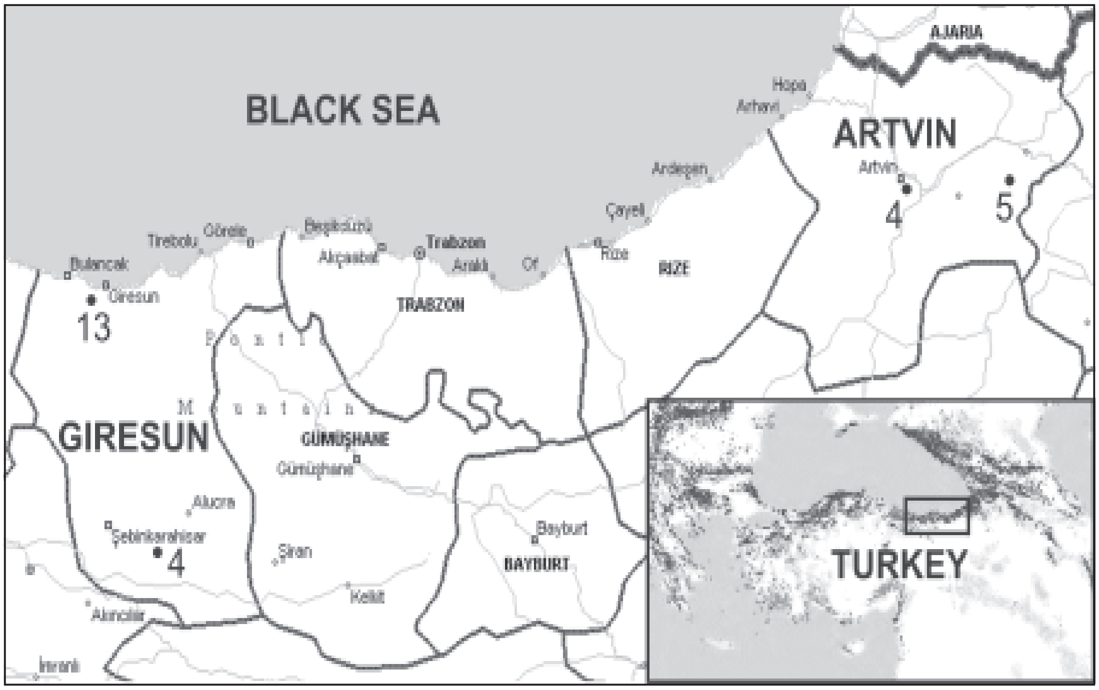

**Fig. 1:** Location of the sample plots in the region of interest. The number at the particular point represents the number of plots established in the area

parameters and three geomorphology features: elevation, aspect and slope. These relationships, if found to be significant, could be used for estimating the actual species diversity from usual forest stand parameters and/or topography characteristics. This could simplify the integration of biodiversity into forest management plans or models applied for the management of natural resources.

## Materials and Methods

In this study, the data collected on 26 sample plots taken by Kapucu (1988) in the eastern Black sea region were used (Fig. 1). The forests of this region were called "humid forests", because the region is the rainiest location of Turkey with an average annual precipitation of 1,500 mm. The average annual temperature is 14°C, with an average of 4°C in winter and 25°C in summer. The soil conditions of the eastern Black sea region are characterised by red podzolic soils and brown forest soils (Oakes, 1958).

All sample plots were located in mixed, natural, untreated stands. The forest stands are mainly composed of oriental spruce (*Picea orientalis* L.), nordmann's fir (*Abies nordmanniana* Link. Spach.), oriental beech (*Fagus orientalis* Lipsky.), scots pine (*Pinus sylvestris* L.), and of a small admixture of other broadleaf species (*Fraxinus, Alnus* and *Populus*). Sample plots were located at an elevation of 1,100 metres to 1,900 metres above sea level. The positions of the sample plots were selected randomly. The size of the plots varied (Table 1) due to the condition for their establishment as they had to encompass a minimum of 100 trees. The plots were of a rectangular shape. On each plot, breast height diameters were measured on trees with height of at least 1.30 m. For each tree, tree species, diameter at breast height determined in the field. Tree height was measured only on 3-5 dominant trees in every sample plot. Similarly, approximate stand age was determined from the increment cores taken from the dominant trees for each tree species. Stand volume in the sample plots was obtained from the local volume tables (Kapucu, 1988). Homogeneity index was calculated from the Lorenz curve which is a suitable tool for graphical representation and for comparison of stand structures. Heterogeneous stands have a low index of homogeneity, while homogeneous stands a higher one (Bachofen and Zingg, 2001).

The basic statistics of stand parameters and geomorphology features are given in Table 1.

**Quantification of tree species diversity:** To evaluate species diversity different indicators were formulated. Among them the most common methods are the "Shannon-Weaver Index (SW)" and the "Simpson Index (SI)" (Ludwig and Reynolds, 1988; Merganic and Smelko, 2004). It was stated that the Shannon-Weaver and Simpson indices are successfull tools for the evaluation and quantification of plant and animal diversity, and are easy and practical measures of area diversity (Dale *et al*., 1994). These indices are closely related and they can be derived from the same one-parameter family of diversity indices (Keylock, 2005). Gorelick (2006) stated that both Shannon's and Simpson's indices have stood the test of time and are still generally regarded as the premier measures of ecological diversity.

Shannon-Weaver index (SW)

$$SW = -\sum_{i=1}^{S} p_i \cdot \ln(p_i) \qquad (1)$$

**Table - 1:** Basic statistics and geomorphology features of sample plots

| | **Parameters** | **No. of plots** | **Min.** | **Max.** | **Mean** | **Standard deviation** |
|---|---|---|---|---|---|---|
| Stand parameters | Plot size [m$^2$] | 26 | 433 | 2,400 | 1.563.38 | 587.02 |
| | Number of trees (N/ha) [pcs/ha] | 26 | 496 | 2.398 | 1.179 | 519.47 |
| | Number of tree species (TS) [pcs] | 26 | 2 | 4 | 3 | 0.74 |
| | Basal area (BA) [m$^2$/ha] | 26 | 23.85 | 80.48 | 58.71 | 14.55 |
| | Number of diameter classes (DC) [pcs] | 26 | 7 | 22 | 13 | 4.46 |
| | Volume (V) [m$^3$/ha] | 26 | 205 | 1,033 | 619 | 206.52 |
| | Stand age (A) [yrs] | 26 | 45 | 150 | 89 | 26.52 |
| | Diameter (Ds) [cm] | 26 | 18.1 | 29.8 | 26.90 | 6.53 |
| | Homogeneity Index (HI) | 26 | 1.65 | 4.81 | 2.66 | 0.68 |
| | Tree species diversity (SW) | 26 | 0.339 | 1.096 | 0.7566 | 0.1925 |
| | Tree species diversity (SI) | 26 | 0.179 | 0.665 | 0.4849 | 0.1100 |
| | Basal area heterogeneity (SW) | 26 | 1.8613 | 2.8702 | 2.3755 | 0.3362 |
| | Basal area heterogeneity (SI) | 26 | 0.8314 | 0.9363 | 0.8910 | 0.0345 |
| Geomorphology features | Elevation (m) | 26 | 1,120.00 | 1,900.00 | 1,653.27 | 179.70 |
| | Slope (°) | 26 | 0.00 | 39.00 | 23.54 | 9.83 |
| | Aspect | 26 | | | Relative proportion (%) | |
| | N | 11 | | | 42.31 | |
| | SW | 5 | | | 19.23 | |
| | Plain | 2 | | | 7.69 | |
| | NW | 2 | | | 7.69 | |
| | NE | 2 | | | 7.69 | |
| | S | 1 | | | 3.85 | |
| | W | 1 | | | 3.85 | |
| | SE | 1 | | | 3.85 | |
| | E | 1 | | | 3.85 | |

**Note:** Shannon-Weaver index (SW), Simpson index (SI)

Simpson index (SI):

$$SI = 1 - \sum_{i=1}^{S} p_i^2 \qquad (2)$$

Where $p_{i,}$ is the proportion of species i on the sample plot calculated from number of trees N, basal area BA per hectare and S is the number of species.

Using the formulas (1) and (2) of the two species diversity indices SW and SI, tree species diversity values were calculated from two stand parameters: number of trees (N) and basal area (BA). The calculation of the heterogeneity indices using the different stand parameters was performed in order to examine and document the influence of tree dimensions on the index value. The calculation from the number of trees N neglects tree size, while the estimation of heterogeneity from BA accounts for the size of the tree (Merganic and Smelko, 2004).

**Basal area heterogeneity:** The heterogeneity of basal area was estimated by the indices SW and SI. To determine the basal area heterogeneity of the sample plots, trees on each sample plot were classified to diameter classes of 4 cm width. Basal area of one diameter class was obtained by multiplying the basal area of the mean tree in the diameter class with the number of trees in the particular diameter class. Total basal area of the sample plot was calculated as the sum of the basal areas of all diameter classes. The relative proportion of the basal area of the i$^{th}$ diameter class from the total basal area was taken as the input value $p_i$ for the calculation of the SW and SI indices in the formulas (1) and (2). Note that in this case the variable *S* in the formulas represents the number of filled diameter classes.

**The relationship between tree species diversity, basal area heterogeneity and stand parameters or geomorphology features:** To examine the relation between tree species diversity and basal area heterogeneity and stand parameters or geomorphology features, linear and non-linear (quadratic and logarithmic) models were used and tested using the SPSS packet (SPSS, 2004) and the Mathcad program (Mathsoft Inc, 2004). The significance of priority in the examined statistical relations was determined according to the coefficient of determinaton R$^2$, standard error of estimination SE and $\alpha$-values of treated linear and non-linear regression models. The type of the model was selected not only with regard to its significance but also by accounting for its logical behaviour.

**Table - 2:** Relationship between tree species diversity, detrended basal area heterogeneity and selected stand parameters according to the Shannon-Weaver and Simpson index

| Stand parameter | | Tree species diversity | | | | | | | | | | | | | Detrended basal area heterogeneity | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Shannon-Weaver index | | | | | | Simpson index | | | | | | | | | | | | |
| | Model | N | | | BA | | | N | | | BA | | | Model | Shannon-Weaver index | | | Simpson index | | |
| | | $R^2$ | SE | α | $R^2$ | SE | α | $R^2$ | SE | α | $R^2$ | SE | α | | $R^2$ | SE | α | $R^2$ | SE | α |
| CL | L | 0.177 | 0.173 | 0.032* | 0.019 | 0.189 | 0.506 | 0.182 | 0.106 | 0.030* | 0.007 | 0.098 | 0.676 | L | 0.005 | 0.128 | 0.741 | 0.004 | 0.036 | 0.752 |
| HI | L | 0.062 | 0.185 | 0.222 | 0.291 | 0.161 | 0.004** | 0.073 | 0.113 | 0.181 | 0.489 | 0.071 | 0.000** | L | 0.596 | 0.081 | 0.000** | 0.629 | 0.022 | 0.000** |
| TS | L | 0.217 | 0.169 | 0.017* | 0.438 | 0.143 | 0.000** | 0.049 | 0.115 | 0.278 | 0.158 | 0.091 | 0.044* | L | 0.059 | 0.124 | 0.232 | 0.027 | 0.035 | 0.424 |
| N/ha | Q | 0.068 | 0.177 | 0.210 | 0.132 | 0.171 | 0.074 | 0.053 | 0.110 | 0.270 | 0.105 | 0.090 | 0.113 | L | 0.321 | 0.106 | 0.003** | 0.282 | 0.030 | 0.005** |
| DC | Q | 0.174 | 0.166 | 0.038* | 0.424 | 0.139 | 0.000** | 0.099 | 0.107 | 0.126 | 0.374 | 0.075 | 0.001** | LN | 0.772 | 0.059 | 0.000** | 0.724 | 0.018 | 0.000** |
| V/ha | L | 0.028 | 0.188 | 0.412 | 0.117 | 0.180 | 0.088 | 0.008 | 0.117 | 0.662 | 0.108 | 0.093 | 0.102 | L | 0.538 | 0.087 | 0.000** | 0.508 | 0.025 | 0.000** |
| A | L | 0.004 | 0.190 | 0.747 | 0.067 | 0.185 | 0.203 | 0.016 | 0.117 | 0.542 | 0.084 | 0.095 | 0.152 | L | 0.195 | 0.115 | 0.02* | 0.216 | 0.032 | 0.017* |
| Ds | Q | 0.102 | 0.174 | 0.120 | 0.321 | 0.151 | 0.003** | 0.040 | 0.111 | 0.337 | 0.252 | 0.082 | 0.011* | LN | 0.699 | 0.068 | 0.00** | 0.633 | 0.021 | 0.000** |
| BA/ha | L | 0.076 | 0.183 | 0.172 | 0.222 | 0.169 | 0.015* | 0.025 | 0.116 | 0.440 | 0.202 | 0.088 | 0.021* | L | 0.407 | 0.099 | 0.000** | 0.385 | 0.028 | 0.001** |

**Note:** Closure (CL), Homogeneity index (HI), Number of tree species (TS) (pcs), Number of trees (N/ha) (pcs), Number of filled diameter classes (DC) (pcs), Volume (V) ($m^3$/ha), Age (A)(yrs), Diameter (Ds) (cm), Basal area (BA) ($m^2$/ha), Model: L = linear, Q = Quadratic, LN = Logarithmic, Statistics: $R^2$ = Coefficient of determination, SE = Standard error. Significance level: ** $\alpha < 0.01$ and * $\alpha < 0.05$

**Table - 3:** Relationship between tree species diversity, detrended basal area heterogeneity, and selected geomorphology features according to the Shannon-Weaver and Simpson Index

| Geomorph-ology feature | | Tree species diversity | | | | | | | | | | | | | Detrended basal area heterogeneity | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | Shannon-Weaver index | | | | | | Simpson index | | | | | | | | | | | | |
| | Model | N | | | BA | | | N | | | BA | | | Model | Shannon-Weaver index | | | Simpson index | | |
| | | $R^2$ | SE | α | $R^2$ | SE | α | $R^2$ | SE | α | $R^2$ | SE | α | | $R^2$ | SE | α | $R^2$ | SE | α |
| Elevation | L | 0.031 | 0.180 | 0.403 | 0.022 | 0.182 | 0.481 | 0.036 | 0.111 | 0.361 | 0.104 | 0.090 | 0.116 | Q | 0.116 | 0.116 | 0.096 | 0.141 | 0.032 | 0.065 |
| Aspect | L | 0.230 | 0.161 | 0.015* | 0.237 | 0.161 | 0.014* | 0.169 | 0.103 | 0.041* | 0.137 | 0.088 | 0.069 | Q | 0.144 | 0.114 | 0.061 | 0.141 | 0.032 | 0.065 |
| Slope | L | 0.148 | 0.169 | 0.058 | 0.058 | 0.178 | 0.245 | 0.181 | 0.102 | 0.034* | 0.046 | 0.093 | 0.303 | L | 0.042 | 0.125 | 0.315 | 0.055 | 0.035 | 0.249 |

Model: L = Linear, Q = Quadratic, Statistics: $R^2$ = Coefficient of determination, SE = Standard error, Significance level: ** $\alpha < 0.01$ and * $\alpha < 0.05$

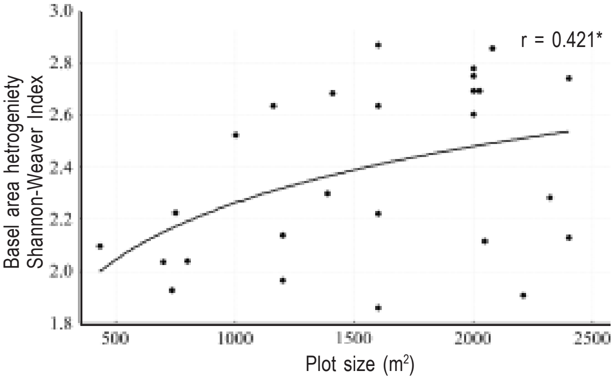

**Fig. 2:** Influence of plot size on basal area heterogeneity calculated from the Shannon-Weaver index

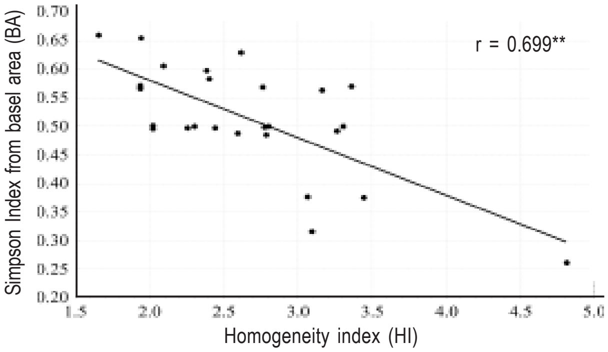

**Fig. 3:** Relationship between the homogeneity index and tree species heterogeneity derived from basal area according to the Simpson index

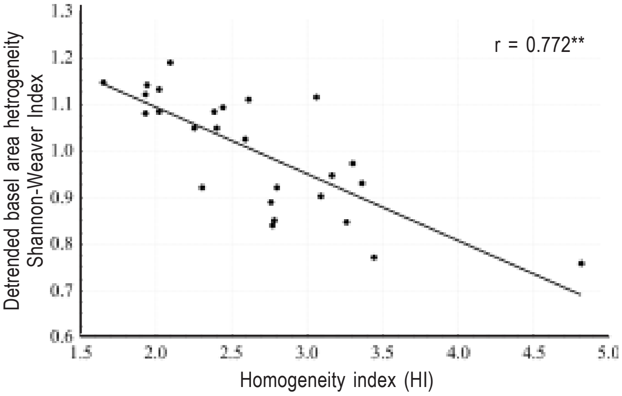

**Fig. 4:** Relationship between the homogeneity index and detrended basal area heterogeneity calculated from the Shannon-Weaver index. The detrended basal area heterogeneity was obtained by dividing the actual basal area heterogeneity with the value from the regression between the basal area heterogeneity and plot size

From the stand parameters the following parameters were included in the analysis: Crown closure CL, homogeneity index HI, number of tree species TS, number of filled diameter classess DC, and number of trees per hectare N/ha, basal area BA/ha, stand volume V/ha, stand age A and mean stand diameter Ds. When analysing their relationship to tree species diversity, in three cases (for N/ha, DC, Ds), a quadratic model was used, while for the other relationships a linear model was applied. Similarly, the relationship between the basal area heterogeneity values and selected stand parameters was also investigated. Linear regression was applied in the majority cases while the logarithmic model was used to describe the relation between basal area diversity and number of diameter classes DC and mean stand diameter Ds.

The examined geomorphology features were elevation, aspect, and slope. Also in this case, the relationship between them and tree species and basal area heterogeneity was tested using linear and non-linear regression analysis. To apply this kind of analysis to all cases, the aspect as a categorical variable was converted to degree values. The best results were obtained with the quadratic model for all of the examined relationships except the one between basal area heterogeneity and slope, where linear model explained more variability.

## Results and Discussion

**Tree species diversity and basal area variability:** The values of tree species diversity obtained from the SW index fluctuated between 0.339 and 1.096, SI ranged from 0.179 to 0.665 (Table 1). In both cases, tree species diversity was highest if the ratio values of all present tree species were equal or rather similar. Similar results were also obtained for basal area heterogeneity.

**How does plot size affect selected diversity quantifiers?:** Species diversity is highly dependent on the size of the analysed population. Due to the fact that the sample plots included in this analysis differ in their size (Table 1), it was of great importance to examine the influence of the plot size on the diversity quantifiers. Regression analysis revealed that all relationships between plot size and tree species diversity quantifiers ($SW_{N,BA}$ and $SI_{N,BA}$) are nonsignificant. This result suggests that the plots represent the minimum area, *i.e.* at the microsite level tree species heterogeneity will not increase if the plot is enlarged. Therefore, the data and the tree species diversity values derived from them can be used for further analyses without any modifications.

However, in the case of basal area heterogeneity, the analysis showed a significant correlation between basal area heterogeneity and plot size (Fig. 2). Using these results, the values of basal area heterogeneity were detrended by dividing the actual value with the value from the calculated regression. The following analyses were performed using these detrended values.

**Relationship between tree species diversity and stand parameters:** In general the relationships between tree species diversity and examined stand parameters are loose, since the correlation coefficient R fluctutaes between 0.02 and 0.70 (*i.e.* $R^2$ is from 0.00 to 0.49). Nevertheless, some relations were detected to be significant (Table 2). Regarding the use of the different stand variables (N and BA) in the calculation of tree species diversity, higher and significant correlations were obtained if diversity values were derived from the basal area rather than from the number of trees.

If the number of trees N was used as the basis for the calculation of tree species diversity, the analysis revealed three significant relations between the Shannon-Weaver Index and closure,

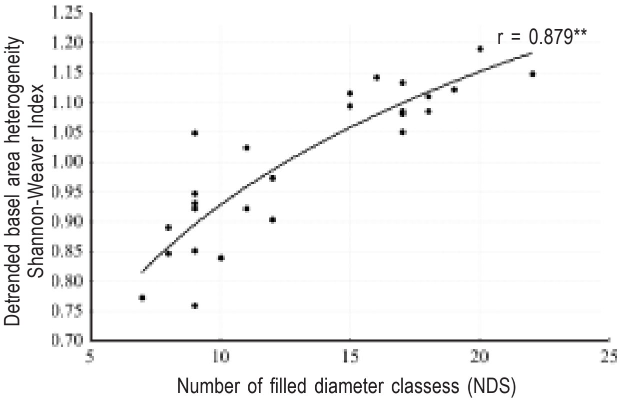

**Fig. 5:** Relationship between the number of filled diameter classes and detrended basal area heterogeneity calculated from the Shannon-Weaver index. The detrended basal area heterogeneity was obtained by dividing the actual basal area heterogeneity with the value from the regression between the basal area heterogeneity and plot size

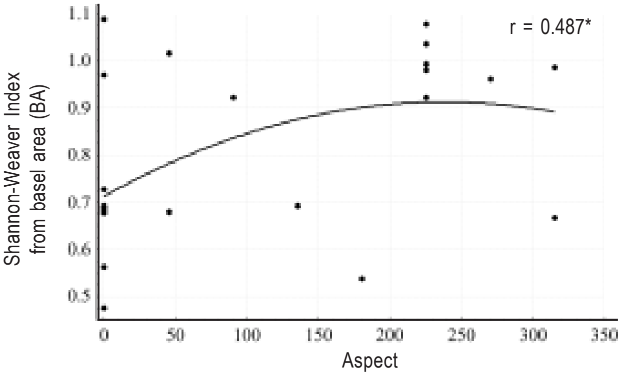

**Fig. 6:** Relationship between the aspect and tree species heterogeneity derived from basal area according to the Shannon-Weaver index

number of tree species, and number of filled diameter classes, and one significant relation between the Simpson index and closure, although in all cases the correlations are low ($R^2 < 0.22$; Table 2). When tree species diversity was calculated from BA, significant correlations with homogeneity index, number of tree species, number of diameter classes, mean stand diameter, and basal area were found for both indices (Table 2).

The results of the analysis indicate that tree species diversity increases in parallel with increasing stand closure, number of tree species, basal area, stand volume, and age, but decreases with an increasing homogeneity index (Fig. 3). Regarding the three relationships where quadratic regression was used for their description, it was found that tree species diversity first decreases with the increasing number of trees per hectare until it reaches its minimum at approximately 1,400 - 1,600 trees per ha, and then it begins to rise. Similar behaviour was observed for the relationship with the number of diameter classes and mean stand diameter.

**Relationship between basal area heterogeneity and stand parameters:** The statitical analysis of the relationship between basal area heterogeneity and selected stand parameters revealed that similar relationships between basal area heterogeneity values and all selected stand parameters were obtained by both diversity indices SW and SI. All examined relationships except the ones between basal area heterogeneity and number of tree species and closure, were significant. The highest $R^2$ value was obtained for the number of diameter classes followed by mean stand diameter, homogeneity index and stand volume (Table 2).

Generally, basal area heterogeneity increases in parallel with increasing age, basal area and stand volume, but decreases with an increasing number of trees per hectare and homogeneity index (Fig. 4). Increasing number of diameter classes DC and mean stand diameter Ds results in higher basal area heterogeneity, but this trend slows down when DC and Ds reach higher values (Fig. 5).

**Relationships between tree species diversity and basal area heterogeneity and geomorphology features:** The analysis revealed a significant correlation between calculated tree species diversity (SW) and aspect regardless of the stand parameter (N, BA) used for the calculation (Table 3). In the case of SI index, significant correlation with aspect was found only if SI was derived from N. Slope had a significant effect only on $SI_N$. However, all significant relationships are loose with $R^2 < 0.24$. Elevation did not appear to be significantly related to any of the examined tree species diversity indices (Table 3).

Regarding the performance of tree species diversity both indices, when calculated from N, first decrease with aspect. The lowest tree species heterogeneity is at the aspect of around 90° (*i.e.* east), after which the diversity begins to increase. If BA was used to calculate tree species diversity, this first increases with aspect, and at 230° (*i.e.* west) it reaches its maximum (Fig. 6). The same behaviour was observed for the relationship of SW and SI versus slope, whereby the minimum heterogeneity was at slopes of 10° and the maximum at around 23-30°. Similarly, with increasing elevation tree species diversity first increases up to approx. 1,800 m above sea level, where it reaches maximum and then decreases.

The relationship between the geomorphology features and basal area heterogeneity was analysed in the same manner. For elevation and aspect, a quadratic model was used, while for slope a linear regression model was applied. According to the results of the analysis, none of the geomorphology features have a significant effect on basal area heterogeneity ($R^2<0.5$ and $\alpha>0.05$, Table 3).

Since the analysis of the relationship between the plot size and tree species diversity did not detect any significant correlation, for a better understanding of these values we apply the verbal scale of species diversity suggested by Merganic *et al.* (2004), although he worked with optimally sized sample plots. The scale has four degrees of species diversity (low, medium, high, and very high) determined by the values of the particular index. According to this scale, the calculated values indicate that the evaluated forest stands have a high degree of tree species diversity.

From the examined stand parameters number of filled diameter classes has the highest correlation with tree species diversity (Table 2). Although in our analysis the quadratic model described this relationship best, in general our results correspond with the findings of Huang *et al*. (2003) who found a positive relation between the number of diameter classes and species diversity.

According to Pitkanen (1998) the significant stand variables for the classification of biodiversity are the number of tree species, and mean stand diameter, which was confirmed also in the presented analysis (Table 2). Generally, tree species diversity increases in parallel with increasing mean stand diameter (Denslow, 1995).

Homogeneity index is the only stand parameter with a negative relationship to tree species heterogeneity (Fig. 3) due to its character. Higher values of the homogeneity index indicate evenaged stand structure, whereas the values between 1.3 and 2.8 are characteristic for unevenaged stands (Kapucu, 1988) with a more complex vertical structure. In such stands, high species diversity can be expected (Brokaw and Lent, 1999). Our results support this hypothesis, as the highest tree species heterogeneity was observed on the plots with the lowest homogeneity index (Fig. 3).

Unlike in the number of works, that documented positive correlation between species diversity and stand density (Palmer *et al*., 2000; Steege *et al*., 2003), our results did not reveal a strong significant relationship with the number of trees per hectare. This is due to the different approach of quantifying species diversity: while the cited works dealt with species richness, in the presented work we examined species heterogeneity encompassing both species abundance and species evenness in a studied community (Bruciamacchie *et al*., 1995). Thus, both tested indices reacted not only to the number of species, but also to their equality in species composition, whereas the SW index is mainly sensitive to the level of evenness in species composition, but less to the numberof species. On the contrary, SI index reacts more to species abundance and less to species evenness (Hubalek, 2000; Liang *et al*., 2007).

Similarly, although our findings correspond with those of Fridley (2003) about the positive relationship between species richness and above-ground production (here represented by stand volume), in our case the correlation was very low and nonsignificant (Table 2).

The collected information of the influence of stand age on plant species diversity varied. Several studies showed a positive correlation (Kirby, 1988; Kiyono, 1990; Ohsawa and Nagaike, 2006), whereas Sykes *et al*., (1989) demonstrated a negative correlation with stand age. Nagaike *et al*. (2003) expressed that increasing stand age did not directly contribute to higher species diversity and richness, which was proved also in our analysis (Table 2).

Statistical analysis of the relationship between basal area heterogeneity and stand parameters revealed significant relations between basal area heterogeneity and the number of diameter classes, mean stand diameter, homogenity index and number of trees (Table 2). As expected, basal area heterogeneity is significantly correlated and increases with increasing number of diameter classes (Fig. 5), because this variable enters the calculation of basal area diversity. Nevertheless, to obtain the highest basal area heterogeneity, apart from the high number of diameter classes the ratio values of diameter classes should also be equal or similar, since the calculation of basal area heterogeneity using the SW and SI indices accounts for the number of diameter classes as well as for the level of evenness in the distribution of the trees in the diameter class.

Johnson (1986), who described topographic position by elevation, slope and aspect, also found its strong influence on species composition of the forests. Our analysis revealed that from these three geomorphology features aspect influences tree species diversity at most (Table 3). However, our data do not allow us to state at which aspect the lowest or highest diversity can be expected, since the experiment is unbalanced, *i.e.* the number of plots in individual aspect groups is unequal (Table 1, Fig. 6).

From the topographic characteristics, the effect of elevation on species diversity is most often examined in the scientific literature (Grytnes and Vetaas, 2002; Pausas *et al*., 2003; Bhattarai *et al*., 2004). Very often hump-shaped curves with maximum species diversity at mid-elevations were reported (Bhattarai and Vetaas, 2003; Bachman *et al*., 2004). A similar pattern was observed in our analysis, although the correlation was nonsignificant (Table 3). Such loose relationships between species diversity and elevation with $R^2$= 0.3 and 0.4 were reported in other studies (Merganic *et al*., 2004).

Biodiversity protection and maintenance is an important issue, which should be integrated into forest management plans or models applied to the management of natural resources. In this context, the main task is to quantify the biodiversity numerically. This study documented how this could be performed using simple index techniques. Another possibility could be to estimate the actual biodiversity status from the usual forest stand parameters and topography. Although our analysis did not reveal any strong correlations, we detected some trends in the behaviour that would require more thorough studies in the future.

## Acknowledgments

We thank Fikret Kapucu for his kind assistance during the fieldwork, and Robert Williams and Jana Šuchtová for their helpful review comments of the earlier drafts of the manuscript.

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**Beskydy**, 17 (2004): 65-72
Ediční středisko MZLU v Brně

# REAKCIA DRUHOVEJ BOHATOSTI A POKRYVNOSTI LESNÝCH FYTOCENÓZ NA ZMENU VLHKOSTNÝCH A TEPLOTNÝCH PODMIENOK V SMREČINÁCH 6. VEGETAČNÉHO STUPŇA STREDNÝCH BESKÝD

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## Abstract

Merganič, J., Ištoňa, J. (Forest Research Institute, Department of ecology and biodiversity of forest ecosystems, T. G. Masaryka 22, SK-960 92 Zvolen, Slovak Republic), Response of species richness and coverage of plant communities to changes in thermal and humidity conditions in spruce forests of 6th altitudinal vegetation zone in the Central Beskids, Beskydy, 2004 (17): 65-72

The presented work analyses the relationship between the change in species richness of plant communities in spruce forests situated in 6th altitudinal vegetation zone and the change in coverage of the plant species that react to thermal and humidity conditions over a period 29 years. For this analysis we used 14 sample plots established in 1972 within the frame of the national habitat type survey, which were re-measured in 2001. The plots represent three forest type groups: *Fagetum-abietino-piceosum, Abieto-Fagetum* a *Fageto-Abietum.* The results suggest that in the analysed forest type groups species richness is declining, which corresponds with the statistically significant increase in coverage of the plant species that are indifferent to thermal and humidity conditions.

**Keywords**: species richness, moisture, temperature, forest plant community, climate change

## 1. Úvod

Sotva sa v poslednom desaťročí trochu zlepšil zdravotný stav lesov, ktoré sa začali zotavovať z neúnosnej, kyslej imisnej záťaže a už nad nimi visí nová hrozba, hrozba z globálnej zmeny klímy i z narušenej ochrannej funkcie atmosféry. Očakáva sa, že zmena klímy bude nielen významná, ale tak rýchla, že časť jedincov v nich, ba aj niektoré druhy drevín, na väčšej časti terajších stanovíšť nebudú schopné sa na ňu adaptovať a ustúpia.

Lesné ekosystémy ako dôležitá zložka biosféry v nedávnej minulosti citlivo reagovali na neúnosnú imisnú záťaž. Ešte citlivejšiu reakciu očakávame pri zmene klímy, ktorá veľmi zúži existenciu najmä ihličnatých drevín, z nich hlavne smreka. Viac-menej podobné zmeny postihnú aj synúziu podrastu, ktorá je v silnej interakcii s inými biotickými zložkami ekosystému a mnohokrát sa využíva aj ako indikátor pre špecifické ciele.

Priebehy počasia za posledné dve decénia s častejšími výskytmi zosilňujúcich výkyvov extrémneho počasia nás presviedčajú, že sme svedkami už započatých klimatických zmien. Nakoniec i samotné klimatické merania potvrdzujú, že v tomto období priemerná teplota stúpla cca o jeden stupeň.

Problematikou zmien bylinnej zložky v lesných ekosystémoch sa venovalo viac autorov, napr. FALLKENGREN & GRERUP (1986, 1987, 1989), TYLER (1987), THIMONIER et al. (1992) a iní. Bylinnej synúzii ako indikátoru významných a dlhodobejších zmien v lesnom ekosystéme sa v našich podmienkach začali venovať až v 90-tych rokoch AMBROS & MÍCHAL

(1992) AMBROS et al. (1995), KRIŽOVÁ(1994, 1996), NIČ (1995, 1999), VOLOŠČUK (2001) a PAVLENDA & IŠTOŇA (2000).

Cieľom predkladanej práce je analýza zmeny druhovej bohatosti bylinnej synúzie lesných fytocenóz za 29-ročnú periódu. Zmena v druhovej bohatosti sa posudzuje jednak vzhľadom ku zmene výskytu teplomilných a vlhkomilných indikátorov a jednak vzhľadom na zmenu v pokryvnosti druhov indiferentných voči týmto ekologickým faktorom.

## 2. Empirický materiál a metodika

Predmetom analýzy je 14 výskumných plôch založených v roku 1972 za účelom národného typologického prieskumu. Plochy sa nachádzajú v nadmorskej výške 800 až 1250 m n. m. v oblasti Pilska a Paráča, ktorú orograficky zatrieďujeme do Stredných Beskýd. Podložie je možné charakterizovať ako nevápnitý flyšový pieskovec. Po typologickej stránke reprezentujú výskumné plochy v danej oblasti najrozšírenejšie lesné spoločenstvá 6. vegetačného stupňa. Z edaficko-trofickej (ekologickej) klasifikácie reprezentujú 3 plochy rad A so skupinou lesných typov (slt) *Fagetum-abietino-piceosum* (Fap), rad B so slt *Abieto-Fagetum* (AF) – 4 plochy a medzirad A/B zastupuje hlavne slt *Fageto-Abietum* (FA) – 7 plôch (ZLATNÍK 1956, 1976, HANČINSKÝ 1972).

V roku 2001 sa na uvedených plochách vykonali opakované kompletné fytocenologické zápisy aj s odberom pôdnych vzoriek. Forma fytozápisov je urobená podľa zaužívaných metodík v zmysle školy Prof. Zlatníka.

### Kvantifikácia druhovej bohatosti

Druhová bohatosť je najstaršie a najjednoduchšie poňatie druhovej diverzity a vyjadruje sa na základe počtu druhov. Diverzita je tým väčšia, čím viac druhov sa v danom spoločenstve nachádza. Vo všeobecnosti sa označuje indexom $N0$.

Z ďalších indexov, ktoré kvantifikujú druhovú bohatosť a z historického hľadiska patria k najznámejším, sú indexy $R1$ a $R2$. Autori, ktorí tieto indexy zaviedli, sa snažili zohľadniť vplyv veľkosti skúmanej populácie tak, že k nej „relativizovali“ počet druhov.

Matematická formulácia vybraných indexov druhovej bohatosti je nasledovná:

$$N0 = S \qquad \text{(HILL 1973)} \qquad [1]$$

$$R1 = (S-1)/\ln(P) \qquad \text{(MARGALEF 1958)} \qquad [2]$$

$$R2 = S/\sqrt{P} \qquad \text{(MENHINICK 1964)} \qquad [3]$$

kde: $S$ – počet druhov
$P$ – celková pokryvnosť bylinného krytu

### Kvantifikácia ekologickej indikácie faktorov prostredia bylinnými druhmi

Ekologická analýza vybraných spoločenstiev vychádzala z kvalitatívnych a kvantitatívnych znakov fytocenózy (druhová diverzita, abundancia a dominancia). Tie sú podľa ZLATNÍKA (1976) výsledkom pôsobenia faktorov prostredia a sú rozhodujúcim indikátorom jeho vlastností. Hodnotenie zmien prostredia pomocou ekologickej analýzy spočíva v kvantifikácii ekočísla pre každý rastlinný druh vzhľadom na ekologický faktor definovaný podľa ELLENBERGA et al. (1992). Štandardným postupom (KRIŽOVÁ & NIČ 1997) a automatizovaným spracovaním (JANKOVIČ et al. 1999) sa vypočítali priemerné ekohodnoty pre každý hodnotený faktor, pričom pre účely tejto práce sme použili výsledky viažuce sa k výskytu teplomilných a vlhkomilných indikátorov.

## 3. Výsledky a diskusia

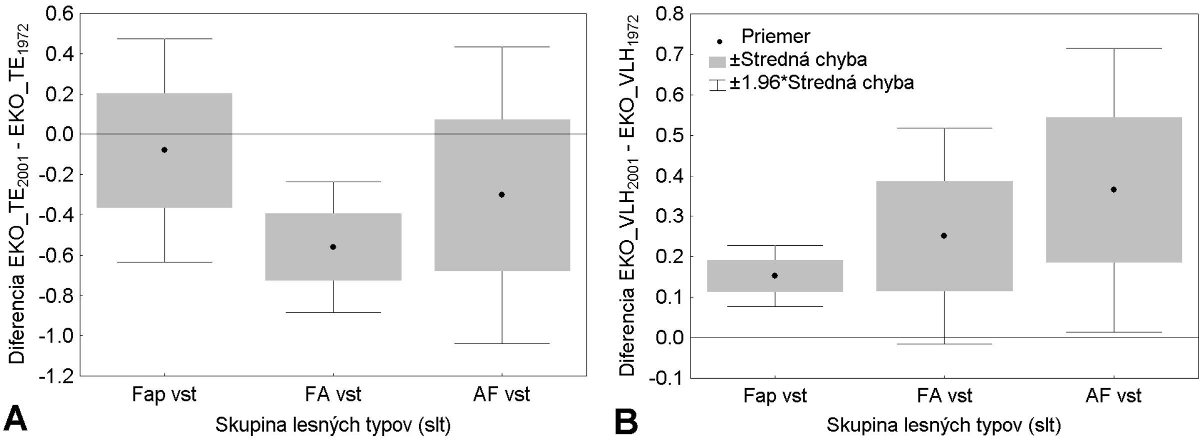
V prvom kroku sme analyzovali vplyv hustoty, veku porastu a celkovej pokryvnosti bylinného krytu na indexy druhovej bohatosti, ekočísla a pokryvnosť indiferentných druhov. Preukázanie vplyvu týchto faktorov je potrebné zohľadniť pri nasledovných analýzach. Je totiž známe, že zakmenenie významne koreluje s vyspelosťou lesného porastu, napr. s vekom a samozrejme ovplyvňuje celkovú pokryvnosť bylinného krytu. V súbore skúmaných dát predstavuje korelácia medzi zakmenením a pokryvnosťou bylinného krytu hodnotu –0.61, teda pokryvnosť stúpa so znižujúcim sa zakmenením. Okrem zakmenenia vplýva vek porastu na pokryvnosť bylinného krytu aj inou formou. Napr. pri rovnakom zakmenení sú v mladých a starých porastoch iné podmienky pre existenciu bylinného krytu spôsobené polohou korún. Viacnásobnou regresnou analýzou sme preto otestovali, či existuje štatisticky významný vplyv týchto faktorov na hodnotené indexy druhovej bohatosti, ekočísla a pokryvnosť druhov indiferentných na teplo a vlhkosť. Ani v jednom prípade sa tento vplyv nepreukázal a preto je možné jednoduchým spôsobom testovať rozdiely (Studentov t test významnosti diferencie) skúmaných veličín vyplývajúce z vývoja fytocenóz za 29-ročnú periódu.

### Analýza zmien vo fytocenologickej indikácii teplotných a vlhkostných podmienok prostredia

Analýza vývoja ekočísiel (ELLENBERG et al. 1992) naznačuje, že dochádza k miernym zmenám v bylinných spoločenstvách v dôsledku meniacich sa ekologických podmienok. Na jednej strane ubúda teplomilných druhov, t.j. pokles ekočísel popisujúcich faktor teplo (obrázok 1A), na strane druhej pribúdajú druhy obľubujúce humídne stanovištia (obrázok 1B). Tento trend je možné pozorovať vo všetkých troch hodnotených slt. Výsledky analýzy korešpondujú so všeobecne známymi poznatkami o tom, že s pribúdajúcou vlhkosťou klesá teplota. Pri druhoch indikujúcich teplo sa v slt FA vst tento výsledok potvrdil aj štatisticky s 95% spoľahlivosťou. Silnejšiu reakciu rastlinných druhov zaznamenávame pri hodnotení faktora vlhkosť, kde sme obdržali dva štatisticky signifikanktné výsledky a to v slt Fap vst a AF.

Obr. 1 Priebeh a testovanie zmeny v hodnotách ekočísel charakterizujúcich reakciu bylinných druhov na teplo – *EKO_TE* (A) a vlhkosť – *EKO_VLH* (B) v závislosti od skupiny lesných typov

Fig. 1 Performance and statistical test of the change in eco-numbers (ELLENBERG et al. 1992) characterising the response of the plant species to thermal – *EKO_TE* (A) and humidity conditions – *EKO_VLH* (B) in each examined forest type group

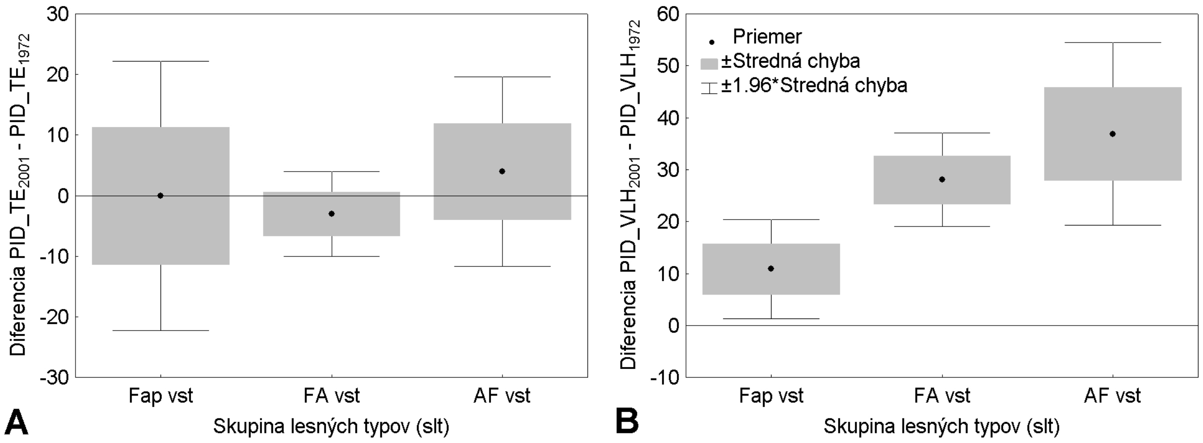


Zaujímavé poznatky priniesla aj ďalšia časť analýzy zameraná na zhodnotenie výskytu indiferentných druhov. Vzhľadom na teplo sa nezistila výrazná zmena pokryvnosti indiferentných druhov (obrázok 2A). V slt Fap vst sa pohybuje táto zmena okolo nuly, pričom je charakteristická veľmi vysokou variabilitou. V slt FA vst je náznak, že pokryvnosť indiferentných druhov v priemere ubudla a naopak v slt AF o niečo vzrástla. Ani jeden výsledok sa však nepotvrdil štatisticky.

Iná je situácia pri hodnotení zmeny v pokryvnosti indiferentných druhov s ohľadom na vlhkosť (obrázok 2B). Vo všetkých troch slt ich pokryvnosť výrazne stúpla, čo sa potvrdilo aj štatisticky na 95% spoľahlivosti. Rastúci trend v ich pokryvnosti môže indikovať to, že dochádza k pomerne častým výkyvom podmienok prostredia. V takýchto podmienkach ubúdajú druhy špecificky viazané na konkrétne podmienky, resp. klesá ich pokryvnosť, čím sa zákonite uvoľňuje priestor druhom so širokou ekologickou valenciou životaschopnosti. Nárast pokryvnosti takýchto druhov môže mať za dôsledok, že v budúcnosti bude problematické klasifikovať podmienky prostredia na základe fytoindikátorov.

Obr. 2 Priebeh a testovanie zmeny v pokryvnosti indiferentných druhov vzhľadom na teplo – *PID_TE* (A) a vlhkosť – *PID_VLH* (B) v závislosti od skupiny lesných typov

Fig. 2 Performance and statistical test of the change in coverage of the plant species that are indifferent to thermal – *PID_TE* (A) and humidity conditions – *PID_VLH* (B) in each examined forest type group



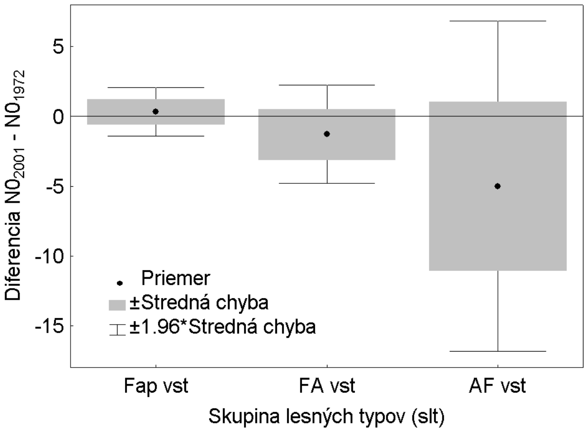
## Analýza zmeny v druhovej bohatosti lesných fytocenóz

Ako už bolo uvedené, druhová bohatosť je hodnotená troma indexami druhovej bohatosti *N0*, *R1* a *R2*. Z analýzy vyplýva, že k najväčšej zmene v počte druhov došlo za sledované obdobie 29 rokov v slt AF (obrázok 3). Táto zmena predstavuje v priemere o -5 druhov nižšiu početnosť v roku 2001 ako v roku 1972. V slt FA vst má táto zmena priemernú hodnotu -1.3 druha. Najnižšiu zmenu sme zaznamenali v treťom hodnotenom slt Fap vst s priemernou hodnotou +0.3 druhu. Jedine v tomto slt došlo k zvýšeniu počtu druhov za sledované obdobie. Vo všeobecnosti je potrebné poznamenať, že ani jedna diferencia sa však nepreukázala štatisticky významne.

Pri hodnotení druhovej bohatosti s ohľadom na rozsah pokryvnosti bylinného krytu (zohľadnenie veľkosti populácie), pričom podmienka rovnakej výmery fytocenologickej plochy je splnená, zisťujeme, že za sledované obdobie došlo k poklesu v druhovej diverzite vo všetkých hodnotených slt. Odrážajú to hodnoty indexov *R1* aj *R2* (obrázok 4A a 4B). Rozdiel v interpretácii týchto indexov spočíva v tom, že index *R1* silnejšie reaguje na plochy s nižšími celkovými pokryvnosťami ako index *R2* (Merganič 2001). Pri indexe *R1* je zmena v druhovej bohatosti v slt Fap vst dokonca na hranici štatistickej významnosti a pri indexe *R2* v tejto slt ide už o štatisticky signifikantnú zmenu potvrdenú na 95% spoľahlivosti.

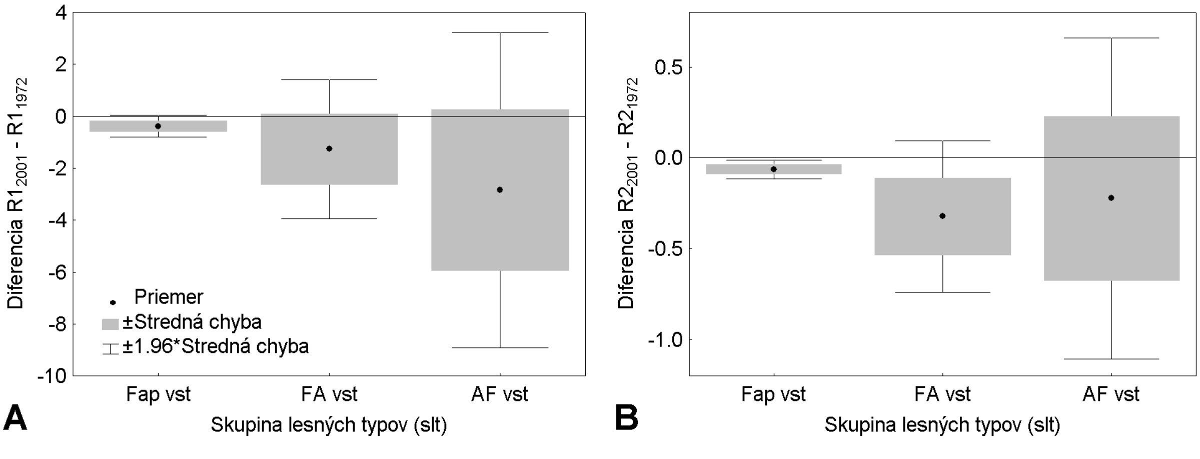
Obr. 3 Priebeh a testovanie zmeny druhovej bohatosti kvantifikovanej indexom *N0* v závislosti od skupiny lesných typov

Fig. 3 Performance and statistical test of the change in species richness characterised by the index *N0* in each examined forest type group



Obr. 4 Priebeh a testovanie zmeny v druhovej bohatosti kvantifikovanej indexom *R1* (A) a *R2* (B) v závislosti od skupiny lesných typov

Fig. 4 Performance and statistical test of the change in species richness characterised by the index *R1* (A) and *R2* (B) in each examined forest type group



Záverečná časť analýzy je zameraná na to, ako vyššie uvedené faktory prostredia indikované lesnou fytocenózou ovplyvňujú druhovú bohatosť. Je založená na jednoduchej korelačnej a viacnásobnej regresnej analýze. Z výsledkov uvedených v tabuľke 1 vyplýva, že medzi zmenou v druhovej bohatosti a zmenou v pokryvnosti druhov indiferentných na teplo a vlhkosť existuje štatisticky významný vzťah. Obrázok 5 prezentuje ukážku vzťahu medzi diferenciami v absolútnom počte druhov (index *N0*) a diferenciami v pokryvnosti druhov indiferentných na vlhkosť.

Dodatočnou viacrozmernou regresnou analýzou však zisťujeme, že zmenu v druhovej bohatosti štatisticky signifikantne ovplyvňuje iba zmena v pokryvnosti druhov indiferentných na vlhkosť, ktorá v sebe pravdepodobne integruje aj zmeny vyplývajúce z reakcie lesnej fytocenózy na zmeny v teplotných podmienkach.

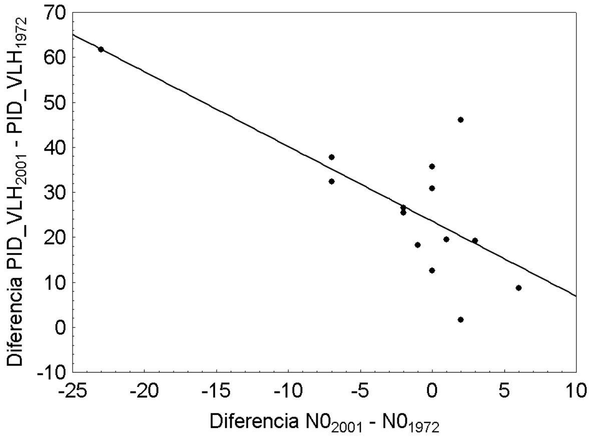
Tab. 1 Korelačná analýza vzťahu medzi zmenou druhovej bohatosti a zmenou pokryvnosti bylinného krytu indikujúceho teplotné a vlhkostné podmienky prostredia (*95% a **99% hladina spoľahlivosti)

Table 1 Correlation analysis of the relationship between the change in species richness and the change in the coverage of plant species indicating thermal and humidity site conditions (significant relationships at *95% and **99% significance level)

| Index | Ekologický faktor | | | |
|---|---|---|---|---|
| | Teplota | | Vlhkosť | |
| | Pokryvnosť indiferentných druhov | Ekohodnota | Pokryvnosť indiferentných druhov | Ekohodnota |
| | Korelačný koeficient (*Rxy*) | | | |
| *N0* | -0.6921 ** | 0.1699 | -0.7428 ** | -0.4084 |
| *R2* | -0.6311 * | 0.1845 | -0.5337 * | -0.4887 |
| *R1* | -0.7168 ** | 0.0311 | -0.6697 ** | -0.4248 |

Obr. 5 Vzťah medzi diferenciami v absolútnom počte druhov (index *N0*) a diferenciami v pokryvnosti indiferentných druhov na vlhkosť.

Fig. 5 Relationship between the differences in absolute number of species (index *N0*) and the differences in the coverage of the species indifferent to humidity



## 4. Záver

V predkladanej práci je analyzovaný vzťah medzi zmenou druhovej bohatosti lesných fytocenóz 6. lesného vegetačného stupňa a zmenou pokryvnosti bylinného krytu reagujúceho na teplotné a vlhkostné podmienky za 29-ročnú periódu. Empirický materiál predstavuje 14 výskumných plôch založených v roku 1972 za účelom celonárodného typologického prieskumu. Plochy reprezentujú tri skupiny lesných typov a to *Fagetum-abietino-piceosum, Abieto-Fagetum* a *Fageto-Abietum.*

Z dosiahnutých výsledkov vyplýva, že v uvedených skupinách lesných typov dochádza k poklesu druhovej bohatosti, čo korešponduje so štatisticky preukázateľným nárastom v pokryvnosti indiferentných druhov. Tento poznatok naznačuje, že globálne klimatické zmeny ovplyvňujú aj vývoj v lesných fytocenózach. Rastúci trend v pokryvnosti indiferentných druhoch môže indikovať to, že dochádza k pomerne častým výkyvom podmienok prostredia. V takýchto podmienkach ubúdajú druhy špecificky viazané na konkrétne podmienky, resp. klesá ich pokryvnosť, čím sa zákonite uvoľňuje priestor druhom so širokou ekologickou valenciou životaschopnosti. Nárast pokryvnosti takýchto druhov nesie so sebou aj riziko pre hospodársku úpravu lesov, pretože v budúcnosti bude problematické klasifikovať podmienky prostredia na základe fytoindikátorov. Tento predpoklad je však potrebné preveriť podrobnejšou analýzou zmien prítomnosti a zastúpenia diferenciálnych druhov, ktoré sú typickými predstaviteľmi konkrétnych súborov lesných typov.

## Poďakovanie

Práca vznikla v rámci riešenia projektu „Vplyv globálnej klimatickej zmeny na lesy Slovenska“.

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**Beskydy**, 18 (2005): 111-118
Ediční středisko MZLU v Brně

# REAKCIA DRUHOVEJ BOHATOSTI A POKRYVNOSTI LESNÝCH FYTOCENÓZ NA ZMENU EDAFICKO-KLIMATICKÝCH PODMIENOK V SMREČINÁCH 6. VEGETAČNÉHO STUPŇA STREDNÝCH BESKÝD

**J. Merganič – J. Ištoňa – K. Merganičová**

## Abstract

[1]Merganič, J., [2]Ištoňa, J., Merganičová, K. ([1]FORIM – Forest research, inventory and monitoring, kpt. Nálepku 277/11, SK-073 01 Sobrance, Slovak Republic, [2]Forest Research Institute, Department of ecology and biodiversity of forest ecosystems, T. G. Masaryka 22, SK-960 92 Zvolen, Slovak Republic), *Response of species richness and coverage of plant communities to changes of edaphic and climatic conditions in spruce forests of 6th altitudinal vegetation zone in the Central Beskids*, Beskydy, 2005 (18): 111-118

The presented work analyses the relationship between the changes in species richness of plant communities in spruce forests situated in 6th altitudinal vegetation zone over a period 29 years and the corresponding temporal change in the coverage of the plant species that are sensitive to edaphic and climatic conditions. For the analysis 14 sample plots were used. The sample plots were established in 1972, and re-measured in 2001. They represent three different forest type groups: *Fagetum-abietino-piceosum, Abieto-Fagetum* a *Fageto-Abietum.* The results show that in the analysed forest type groups species richness is declining. The environment acidification was found to be the main factor causing this decline. The other analysed factors, i.e. temperature, humidity and continentality, have a significant positive effect on the coverage of the species that are indifferent to these factors. The increase of the coverage of the indifferent species can be induced by frequent changes of environment conditions due to the global climatic changes. In future, the expansion of such species may cause problems in the classification of site conditions based on phytocenological surveys.

**Keywords**: species richness, moisture, temperature, light, continentality, soil reaction, nitrogen, forest plant community, climate change, Ellenberg ecovalues

## 1. Úvod

V posledných desaťročiach došlo pod vplyvom zložitého synergicky pôsobiaceho komplexu človekom podmienených faktorov k veľmi výraznému ovplyvneniu životného prostredia na celej Zemi. Táto situácia podmienila rozvoj metód kvantifikácie a analýzy týchto zmien, ich možných príčin a sa stala predmetom mnohých prác. V minulom ročníku „Beskydy 2004“ sme oboznámili vedeckú verejnosť s prácou, ktorej predmetom bola analýza reakcie druhovej bohatosti a pokryvnosti na zmenu vlhkostných a teplotných podmienok. V tomto ročníku si kladieme za povinnosť dokončiť uvedenú analýzu v širšom, ale komplexnejšom kontexte. Vzhľadom k tomu zaberajú v predkladanej práci kapitoly výsledky a diskusia väčší priestor, kým v úvodných kapitolách podávame len stručnú informáciu.

Cieľom práce je analýza zmeny druhovej bohatosti bylinnej synúzie lesných fytocenóz za 29-ročnú periódu. Zmena v druhovej bohatosti sa posudzuje vzhľadom ku zmene výskytu fytoindikátorov reagujúcich na pH, kontinentalitu, svetlo a dusík, ale aj vzhľadom na zmenu v pokryvnosti druhov indiferentných voči týmto ekologickým faktorom. Záverečným čiastkovým cieľom je súborná analýza reakcie zmeny druhovej bohatosti na vplyv komplexu 6 ekofaktorov, t.j. vlhkosť, teplota, pH, kontinentalita, svetlo a dusík definovaných podľa ELLENBERGA et al. (1992).

## 2. Empirický materiál a metodika

Predmetom analýzy je 14 výskumných plôch založených v roku 1972 reprezentujúcich tri trofické rady, t.j. rad A so skupinou lesných typov (slt) *Fagetum-abietino-piceosum* (Fap), rad B so slt *Abieto-Fagetum* (AF) a medzirad A/B zastupuje hlavne slt *Fageto-Abietum* (FA) (ZLATNÍK 1956, 1976, HANČINSKÝ 1972). Bližší popis je uvedený v MERGANIČ & IŠTOŇA (2004).

### 2.1 Kvantifikácia druhovej bohatosti

Druhová bohatosť bola kvantifikovaná tromi indexami, t.j. indexom *N0* (HILL 1973), *R1* (MARGALEF 1958) a *R2* (MENHINICK 1964). Bližší popis je uvedený v MERGANIČ & IŠTOŇA (2004).

### 2.2 Kvantifikácia ekologickej indikácie faktorov prostredia bylinnými druhmi

Hodnotenie zmien prostredia pomocou ekologickej analýzy spočíva v kvantifikácii ekočísla pre každý rastlinný druh vzhľadom na ekologický faktor definovaný podľa ELLENBERGA et al. (1992). Bližší popis je uvedený v MERGANIČ & IŠTOŇA (2004).

## 3. Výsledky a diskusia

### 3.1 Analýza zmien vo fytocenologickej indikácii podmienok prostredia – pH reakcia

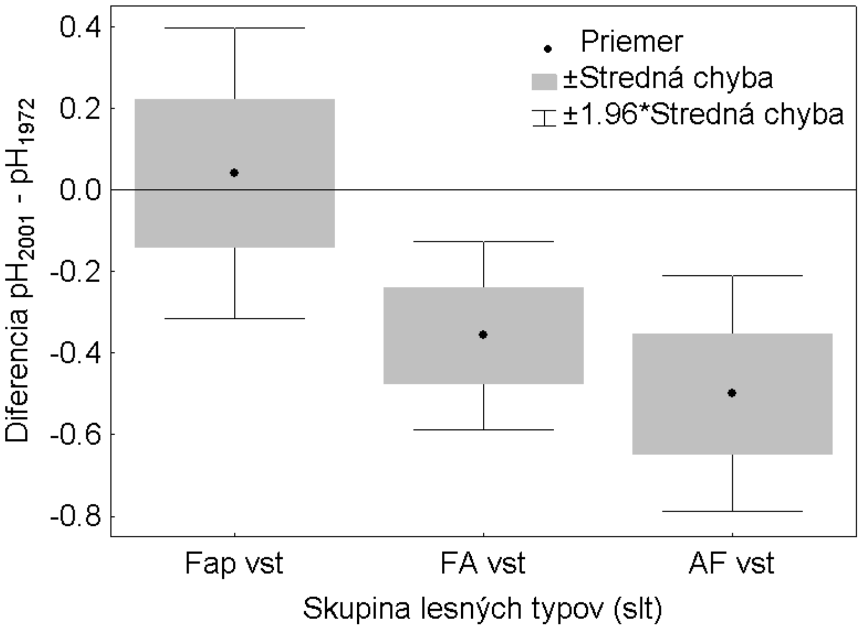
Úvodná analýza, ktorá čiastočne siaha nad rámec fytoindikačných analýz, je zameraná na zmenu v reakcii (pH) vrchnej vrstvy pôdy. Slúži pre porovnanie priebehu hodnôt charakterizujúcich reakciu fytoindikátorov (priemerná hodnota ekočísla) na zmenu veličiny, ktorá je predmetom fytoindikácie (priemerná hodnota pH). Z porovnania obrázkov 1 a 2A môžeme vidieť, že ich zmeny sú identické. Z obrázku 1 vyplýva, že za 29-ročnú periódu došlo v prechodnom trofickom rade (A/B) a v živnom rade (B) k štatisticky signifikantnému zakysleniu. Na túto skutočnosť reagovali aj fytoindikátory kyslosti a v FA vst a AF vst (obrázok 2A) zaznamenávame štatisticky preukázateľnú zmenu v priemerných hodnotách ekočísel. Z obrázku 2B, kde je znázornený priebeh priemerných diferencií v pokryvnosti indiferentných druhov, je vidieť takmer opačnú reakciu. Pokryvnosť týchto druhov v Fap vst klesá, kým v FA vst a AF vst stúpa. V dvoch prípadoch (Fap vst, AF vst) ide o výsledok štatisticky signifikantný na 68% spoľahlivosti. Štatisticky signifikantné zakyslenie v živnejších trofických radoch sa prejavilo zvýšenou pokryvnosťou druhov indiferentných k pH. V kyslom rade sa dominancia fytoindikátorov kyslosti ešte viac zvýraznila.

### 3.2 Kontinentalita

Toto ekočíslo vyjadruje reakciu rastlinných druhov na rozdiely v teplote a vzdušnej vlhkosti počas dňa a roka. Z obrázku 3A vyplýva, že v živnejších radoch došlo k štatisticky signifikantnému zvýšeniu priemeru ekohodnoty tohto faktora, t.j. zvýšil sa výskyt a pokryvnosť druhov znášajúcich kontinentálnejšiu klímu. Pri pohľade na pokryvnosť indiferentných druhov (obrázok 3B) k tomuto faktoru sme medzi hodnotenými obdobiami nezaznamenali výraznejšie rozdiely, čo je možné potvrdiť s 95% pravdepodobnosťou.

Obr. 1 Priebeh a testovanie zmeny priemerných pH hodnôt v závislosti od skupiny lesných typov

Fig. 1 Performance and statistical test of the change of average pH values in each examined forest type group



### 3.3 Svetlo

Ďalšie ekočíslo charakterizuje intenzitu osvetlenia, ktorá je pre rastliny počas vegetačného obdobia optimálna. Pri pohľade na zmenu hodnôt priemerných ekočísel (obrázok 4A) zisťujeme v živnejších radoch štatisticky silný (na 95% spoľahlivosti) kladný posun. V kyslom rade sa táto tendencia taktiež potvrdila, aj keď s nižšou pravdepodobnosťou (68%). Prezentované diferencie dokazujú zvýšenie prítomnosti druhov s vyššími nárokmi na svetlo. Jednou z možných analyzovaných príčin bola zmena v zakmenení (redukcia zápoja so zvyšujúcim sa vekom), ale vzťah k tejto zmene sa štatisticky nepotvrdil. Ide však o malý rozsah súboru, ktorý mohol spôsobiť nesignifikantnosť tohto výsledku.

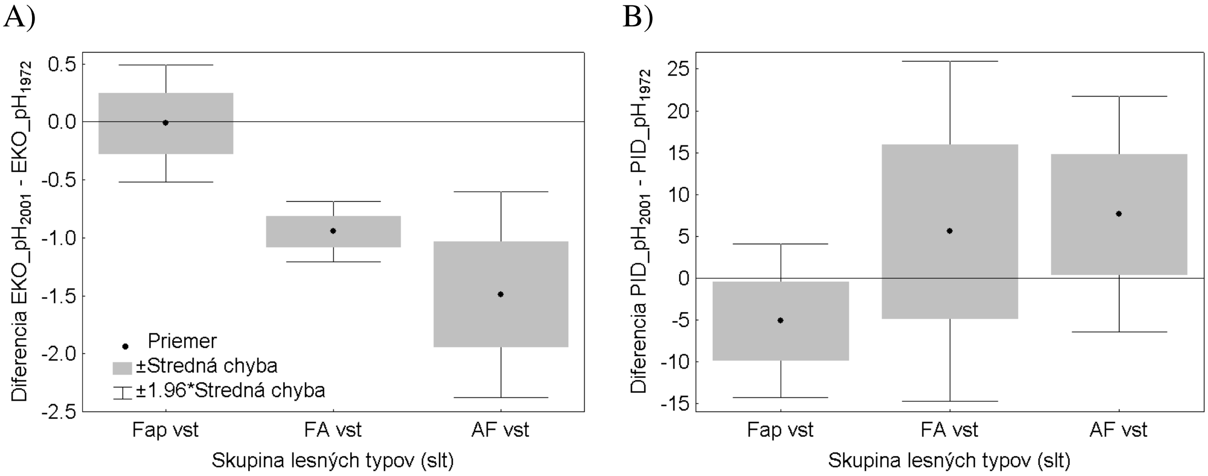
Ako vidieť na obrázku 4B, druhov indiferentných k svetlu naopak ubudlo, resp. ich pokryvnosť sa znížila. Tento jav sa potvrdil vo všetkých troch hodnotených slt štatisticky signifikantne, aj keď pri rozdielnej spoľahlivosti (v Fap vst, FA vst na 68% a v AF vst na 95%). Pokles pokryvnosti indiferentných druhov logicky súvisí so zvýšeným výskytom svetlomilných druhov, čo dokumentujú zmeny priemerných ekočísel (obrázok 4A).

### 3.4 Dusík

Ekočíslo charakterizuje nároky druhov na dusík a vyjadruje závislosť ich výskytu od zásoby minerálneho dusíka v pôde počas vegetačnej doby. Priebeh priemerných diferencií ekočísel (obrázok 5A) má veľmi podobnú tendenciu ako priebeh diferencií pri analýze zaoberajúcou sa pH reakciou. V živnejších radoch (A/B, B), teda v FA vst a AF vst došlo k výraznej zápornej štatisticky signifikantnej zmene. Súvisí to s poklesom pH, čím dochádza k spomaleniu humifikačných procesov a mineralizácie dusíka, a tým k obmedzeniu jeho prístupnosti pre rastliny. Správanie sa pokryvnosti indiferentných druhov ukazuje (obrázok 5B), že v slt Fap vst (rad A) sa vplyvom nezmenenej reakcie pH ich pokryvnosť veľmi nemení resp. je tu tendencia k ich poklesu. V prechodnom rade A/B (FA vst) a živnom rade B (AF vst) je naopak náznak k zvýšeniu ich pokryvnosti.

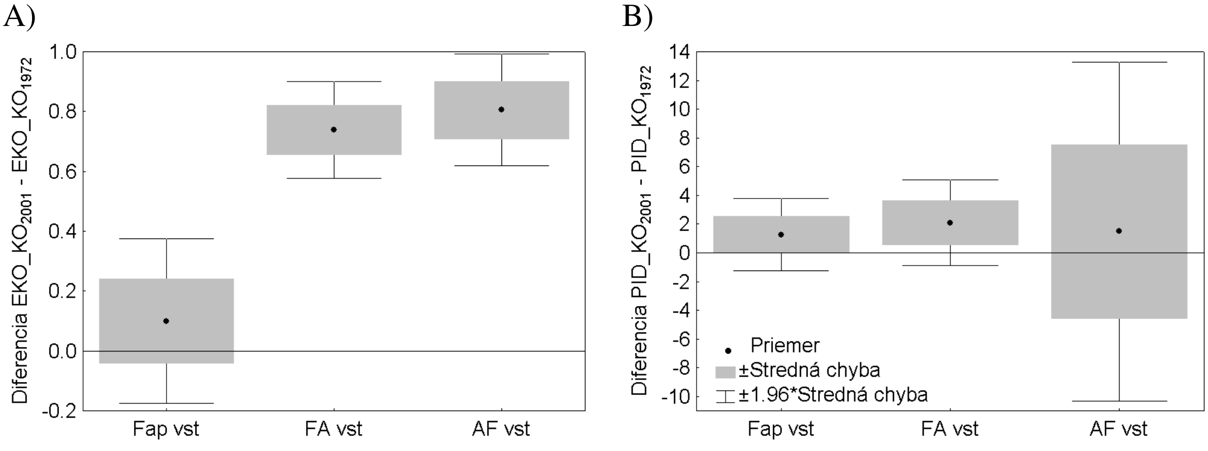
Obr. 2 Priebeh a testovanie zmeny v hodnotách ekočísel charakterizujúcich reakciu bylinných druhov na reakciu pH – *EKO_pH* (A) a priebeh a testovanie zmeny v pokryvnosti indiferentných druhov vzhľadom na reakciu pH – *PID_pH* (B) v závislosti od skupiny lesných typov

Fig. 2 Performance and statistical test of the change in the ecovalues (ELLENBERG et al. 1992) characterising the response of the plant species to pH – *EKO_pH* (A) and performance and statistical test of the change in coverage of the plant species that indifferent to pH – *PID_pH* (B) in each examined forest type group



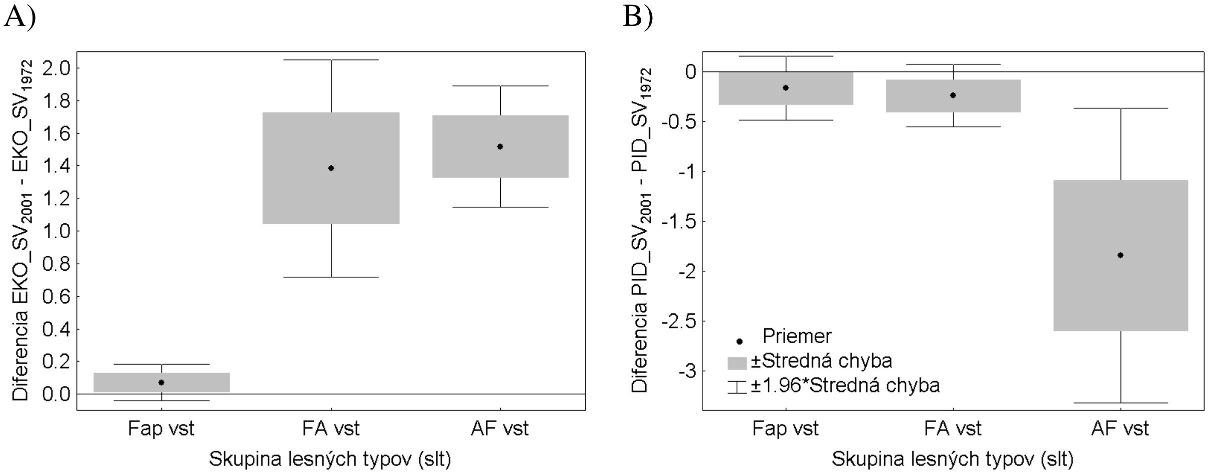
Obr. 3 Priebeh a testovanie zmeny v hodnotách ekočísel charakterizujúcich reakciu bylinných druhov na kontinentalitu – *EKO_KO* (A) a priebeh a testovanie zmeny v pokryvnosti indiferentných druhov vzhľadom na kontinentalitu – *PID_KO* (B) v závislosti od skupiny lesných typov

Fig. 3 Performance and statistical test of the change in the ecovalues (ELLENBERG et al. 1992) characterising the response of the plant species to continentality – *EKO_KO* (A) and performance and statistical test of the change in coverage of the plant species that are indifferent to continentality – *PID_KO* (B) in each examined forest type group



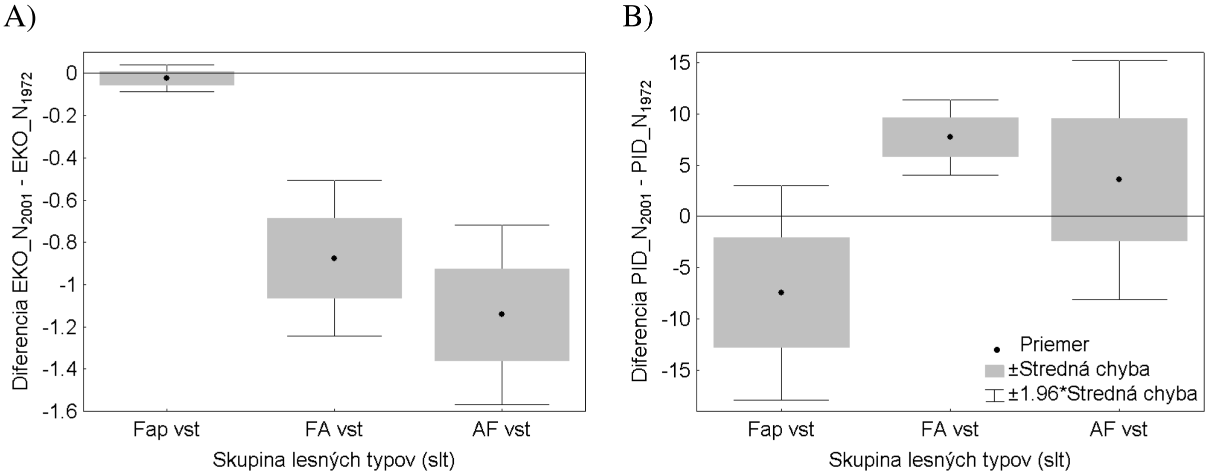
Obr. 4 Priebeh a testovanie zmeny v hodnotách ekočísel charakterizujúcich reakciu bylinných druhov na svetlo – *EKO_SV* (A) a priebeh a testovanie zmeny v pokryvnosti indiferentných druhov vzhľadom na svetlo – *PID_SV* (B) v závislosti od skupiny lesných typov

Fig. 4 Performance and statistical test of the change in the ecovalues (ELLENBERG et al. 1992) characterising the response of the plant species to light – *EKO_SV* (A) and performance and statistical test of the change in coverage of the plant species that are indifferent to light – *PID_SV* (B) in each examined forest type group



Obr. 5 Priebeh a testovanie zmeny v hodnotách ekočísel charakterizujúcich reakciu bylinných druhov na dusík – *EKO_N* (A) a priebeh a testovanie zmeny v pokryvnosti indiferentných druhov vzhľadom na dusík – *PID_N* (B) v závislosti od skupiny lesných typov

Fig. 5 Performance and statistical test of the change the in the ecovalues (ELLENBERG et al. 1992) characterising the response of the plant species to nitrogen – *EKO_N* (A) and performance and statistical test of the change in coverage of the plant species that are indifferent to nitrogen – *PID_N* (B) in each examined forest type group



Tab. 1 Viacnásobná regresná analýza vplyvu šiestich ekofaktorov (*pH* – reakcia pH, *SV* – svetlo, *N* – dusík, *TE* – teplota, *VLH* – vlhkosť, *KO* – kontinentalita) kvantifikovaná ekočíslami (*EKO*) podľa Ellenberga et al. (1992) na reakciu druhovej bohatosti – index *N0* (*95% a **99% hladina spoľahlivosti)

Table 1 Multiple regresion analysis of the influence of six ecofactors (pH value, light, nitrogen, temperature, humidity, continentality) quantified with ecovalues by Ellenberg et al. (1992) on species richness ([1]Ecofactor, [2]Standardised regression coefficient, [3]Partial correlation, [4]Regression coefficient, [5]Standard error of regression coefficient, [6]Student's *t* value, [7]Intercept, significant relationships at *95% and **99% significance level)

| [1]Ekofaktor | [2]Štandardizovaný regresný koeficient | [3]Parciálny korelačný koeficient | [4]Regresný koeficient (*b*) | [5]Stredná chyba *a, b* koeficienta | [6]Studentova *t* štatistika |
|---|---|---|---|---|---|
| EKO_pH | 1.081710 | 0.776185 | 10.04104 | 2.459268 | 4.082937 ** |
| EKO_SV | 0.593725 | 0.559869 | 4.79157 | 2.138112 | 2.241030 * |
| EKO_N | 0.164031 | 0.213583 | | | 0.691360 |
| EKO_TE | -0.110922 | -0.149162 | | | -0.477028 |
| EKO_VLH | -0.094270 | -0.113918 | | | -0.362601 |
| EKO_KO | -0.103992 | -0.100703 | | | -0.320077 |
| [7]Absolútny koeficient (*a*) | | | 1.57829 | 2.191272 | 0.720263 |

Tab. 2 Viacnásobná regresná analýza vplyvu šiestich ekofaktorov (*pH* – reakcia pH, *SV* – svetlo, *N* – dusík, *TE* – teplota, *VLH* – vlhkosť, *KO* – kontinentalita) kvantifikovaná pokryvnosťou indiferentných druhov (*PID*) na reakciu druhovej bohatosti – index *N0* (*95% a **99% hladina spoľahlivosti)

Table 2 Multiple regresion analysis of the influence of six ecofactors (pH value, light, nitrogen, temperature, humidity, continentality) quantified with the coverage of indifferent plant species ([1]Ecofactor, [2]Standardised regression coefficient, [3]Partial correlation, [4]Regression coefficient, [5]Standard error of regression coefficient, [6]Student's *t* value, [7]Intercept, significant relationships at *95% and **99% significance level)

| [1]Ekofaktor | [2]Štandardizovaný regresný koeficient | [3]Parciálny korelačný koeficient | [4]Regresný koeficient (*b*) | [5]Stredná chyba *a, b* koeficienta | [6]Studentova *t* štatistika |
|---|---|---|---|---|---|
| PID_pH | 0.411230 | 0.722187 | 0.138289 | 0.044150 | 3.13224 * |
| PID_TE | -0.426089 | -0.674071 | -0.228976 | 0.083639 | -2.73765 * |
| PID_KO | -0.374875 | -0.649277 | -0.404109 | 0.157789 | -2.56107 * |
| PID_VLH | -0.381895 | -0.606905 | -0.170640 | 0.074487 | -2.29086 * |
| PID_SV | 0.128597 | 0.315420 | | | 0.940135 |
| PID_N | -0.097409 | -0.180269 | | | -0.518369 |
| [7]Absolútny koeficient (*a*) | | | 2.667358 | 2.116314 | 1.26038 |

## 3.5 Integrovaný vplyv skúmaných faktorov na druhovú bohatosť

Súborný vplyv hodnotených ekofaktorov na druhovú bohatosť (*N0*, *R1*, *R2*) sme testovali viacnásobnou regresnou analýzou samostatne prostredníctvom hodnôt ekočísel a pokryvnosti indiferentných druhov. Tento vzťah bol analyzovaný na všetkých plochách spolu bez ohľadu na skupiny lesných typov z dôvodu malého rozsahu dát. V tabuľke 1 uvádzame jej výsledky pre hodnoty ekočísel a vzhľadom k indexu *N0*. Z nej vyplýva, že najtesnejší vzťah k zmene v počte druhov (index *N0*) má ekofaktor pH. Svedčí o tom najvyššia hodnota Studentovej *t* štatistiky ako aj hodnota parciálneho koeficienta, ktorý udáva tesnosť korelácie medzi hodnotenými veličinami pri vylúčení vplyvu ostatných ekofaktorov. Druhým ekofaktorom, ktorý výrazne vplýva na zmenu v druhovej bohatosti, je ekofaktor svetlo. Jeho významnosť sa potvrdila s 95% pravdepodobnosťou. Ostatné ekofaktory vyšli z analýzy ako nevýznamné. Výsledky tejto analýzy potvrdzujú doteraz známe tendencie v zakysľovaní prostredia vplyvom jeho značného znečistenia.

Pri sledovaní vplyvu ekofaktorov na zmenu v druhovej bohatosti kvantifikovanú indexami *R1* a *R2* sme dospeli k obdobnému výsledku s tým rozdielom, že najvýznamnejším a jediným štatisticky vplyvným ekofaktorom bola reakcia pH.

Výsledky druhej časti súhrnnej analýzy zameranej na preukázanie vplyvu hodnotených ekofaktorov na druhovú bohatosť (index *N0*) prostredníctvom fytoindikácie zmien v pokryvnosti indiferentných druhov sú uvedené v tabuľke 2. Z nej vyplýva, že pokles druhovej bohatosti je spôsobený zvyšovaním pokryvnosti resp. výskytom druhov indiferentných voči štyrom ekofaktorom a to pH reakcii, teplote, kontinentalite a vlhkosti. Z nich najsilnejší vplyv má pokryvnosť druhov indiferentných voči pH reakcii.

Pri sledovaní vplyvu týchto ekofaktorov na reakciu druhovej bohatosti kvantifikovanú indexami *R1* a *R2* sa štatisticky signifikantne preukázal iba jediný ekofaktor a to teplota.

## 4. Záver

V predkladanej práci je analyzovaný vzťah medzi zmenou druhovej bohatosti lesných fytocenóz 6. lesného vegetačného stupňa a zmenou ekologických podmienok. Druhová bohatosť je kvantifikovaná troma indexami *N0* (HILL 1973), *R1* (MARGALEF 1958) a *R2* (MENHINICK 1964). Zmena ekologických podmienok je stanovená fytoindikáciou druhov voči šiestim ekologickým faktorom (vlhkosť, teplota, pH reakcia, kontinentalita, svetlo a dusík) definovaným podľa ELLENBERGA et al. (1992) za 29-ročnú periódu. Empirický materiál predstavuje 14 výskumných plôch z oblasti Stredných Beskýd založených v roku 1972 za účelom celonárodného typologického prieskumu. Plochy reprezentujú tri skupiny lesných typov a to *Fagetum-abietino-piceosum, Abieto-Fagetum* a *Fageto-Abietum.*

Z dosiahnutých výsledkov vyplýva, že v uvedených skupinách lesných typov došlo za uvedenú periódu k poklesu druhovej bohatosti. Za najvážnejší faktor môžeme považovať zakysľovanie prostredia pravdepodobne spôsobené znečistením ovzdušia. Ďalšími významnými faktormi, ktoré ovplyvňujú druhovú bohatosť hodnotených lokalít, sú teplota, vlhkosť a kontinentalita. Uvedené ekofaktory štatisticky preukázateľne spôsobujú nárast v pokryvnosti druhov indiferentných k daným ekofaktorom. Rastúci trend v pokryvnosti indiferentných druhov môže indikovať to, že dochádza k pomerne častým výkyvom podmienok prostredia, čo úzko súvisí v súčasnosti s veľmi frekventovaným javom globálnych klimatických zmien. V takýchto podmienkach ubúdajú druhy špecificky viazané na konkrétne podmienky, resp. klesá ich pokryvnosť, čím sa zákonite uvoľňuje priestor druhom so širokou ekologickou valenciou. Výskyt a rozširovanie sa takýchto druhov môže v budúcnosti spôsobiť problémy typológii pri klasifikácii podmienok prostredia, avšak túto hypotézu je potrebné preveriť podrobnejšou analýzou na rozsiahlejšom empirickom materiáli.

## Poďakovanie

Táto práca bola podporovaná Agentúrou na podporu vedy a techniky na základe Zmluvy č. APVT- 27-009304.
This work was supported by Science and Technology Assistance Agency under the contract No. APVT-27-009304.

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Beskydy, 2008, 1 (2): 155–162
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ISSN: 1803-2451

# Vzťah medzi fytocenózou a vývojovým štádiom smrekového prírodného lesa v supramontannom stupni NPR Babia Hora

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**Abstract:** Merganič, J., Merganičová, K., Vorčák, J., Ištoňa, J., 2008: Relationship between plant communities and developmental stage of natural spruce forest in the subalpine forest belt of National Nature Reserve of Babia hora. – *Beskydy*, 1 (2): 155–162

The work presents the results from the survey of plant communities in forest stands of national nature reserve Babia hora. The relevés were collected on 57 sample plots situated at an elevation ranging from 1,222 m to 1,503 m above sea level. The plots are equally divided between the three developmental stages of virgin forests. The analysis revealed that in the studied area no plant species can be accounted for the differentiation between the developmental stages. Our results indicate that in the conditions of Babia hora two types of cyclical changes of plant species occur: a small and a large cycle of plant communities. The analysis of the influence of developmental stage on the proportion coverage of plant species with the same Ellenberg´s indicator value showed that environmental ecological conditions are not affected by the developmental stage of virgin forest. The status of forest stands (permanently released canopy) and specific climate conditions of Babia hora are preconditions for spatially homogenous plant communities.

**Keywords:** virgin forest, developmental stage, plant community, Ellenberg´s indicator value, covariance analysis

## Úvod a problematika

Pralesy predstavujú v Strednej Európe klimaxové štádium ekosystémov (Križová et al. 1992). Základnou črtou klimaxu je dynamická rovnováha medzi prijatou, t.j. fixovanou energiou a energiou spotrebovanou (Križová et al. 1992), čo z produkčného hľadiska znamená „nulový“ prírastok (Vacek 2003). Podľa Korpeľa (1995) sa v stredoeurópskych podmienkach takáto rovnováha dá dosiahnuť na ploche 30 až 60 ha v závislosti od geografického územia a lesného typu. Na menších plochách je možné pozorovať prirodzenú dynamiku vývoja lesného ekosystému. Vývoj stredoeurópskych pralesov prebieha cez tzv. malý vývojový cyklus (Zukrigl et al. 1963, Mayer et al. 1972, Průša 1990, Korpeľ 1995 atď.), ktorý autori charakterizujú striedaním sa niekoľkých vývojových štádií. Korpeľ (1995) vylíšil tri štádiá vývojového cyklu: štádium dorastania, optima a rozpadu, pričom každé z nich definoval na základe vlastností drevinovej vrstvy lesného ekosystému. Cieľom predkladaného príspevku je na príklade smrekového prírodného lesa v NPR Babia Hora zistiť, či sa jednotlivé vývojové štádiá odlišujú aj v druhovom zložení fytocenózy. V rámci tejto analýzy chceme tiež preveriť, či existuje indikačný druh, ktorý by mohol slúžiť ako pomocný ukazovateľ pri vylišovaní vývojových štádií prírodného lesa v danej oblasti. Zároveň si kladieme za cieľ zhodnotiť vplyv vývojového štádia na ekologické podmienky indikované fytocenózou v danej oblasti.

## Materiál a metodika

Údaje použité v tejto práci pochádzajú z inventarizácie NPR Babia hora vykonanej v roku 2002 (Merganič et al. 2003). NPR Babia hora patrí orograficky do sústavy vonkajších Západných Karpát, časti Oravských Beskýd, do komplexu horského masívu Babej hory. Národná prírodná rezervácia zaberá celkovo 503.94 ha a je umiestnená na západných, južných a juhozápadných svahoch Babej hory v nadmorskej výške 1100 až 1725 m n. m. (Korpeľ 1989). Geologické podložie je tvorené zo súvrství nevápnitých flyšových pieskovcov. Hlavnými pôdnymi predstaviteľmi sú podzol a kambizem, len v malej miere litozem. Priemerné ročné teploty dosahujú vo vrcholových polohách 2 °C a 4 °C v nižších polohách a priemerný ročný úhrn zrážok je 1400 mm.

Lesné porasty sú tvorené prevažne smrekom obyčajným (*Picea abies* /L./ Karst.), vtrúsene sa vyskytujú jarabina vtáčia (*Sorbus aucuparia* L.), jedľa biela (*Abies alba* Mill.) a buk lesný (*Fagus sylvatica* L.). Les vystupuje približne do nadmorskej výšky 1500 m n. m. Nad touto hranicou lesa nastupuje pásmo kosodreviny vystriedané v najvyšších partiách NPR alpinskými lúkami.

V rámci inventarizácie zalesneného územia NPR Babia hora bolo založených 57 kruhových skusných plôch o rozlohe 500 $m^2$ tak, aby boli plochy rovnomerne rozdelené medzi tri vývojové štádiá (dorastanie–optimum–rozpad podľa Korpeľa (1989)) a štyri výškové kategórie, t.j. do 1260 m, 1260–1360 m, 1360–1460 m a nad 1460 m n. m. V prvých troch výškových kategóriách sa v každom vývojovom štádiu založilo po 5 skusných plôch a v poslednej, najvyššej výškovej kategórii po 4 skusných plochách, pretože táto kategória tvorí plošne nepatrnú časť záujmovej oblasti.

Súčasťou terénneho šetrenia boli aj fytocenologické zápisy bylinnej a krovinnej vrstvy. Plocha fytocenologického zápisu bola lokalizovaná v strede skusnej plochy a zodpovedala požiadavke stanovištnej a floristickej homogenity. Tvar plochy bol štvorcový a jej výmera mala 25 $m^2$, čo v daných lokalitách zodpovedá výmere minimálneho areálu a spadá do intervalov odporúčaných výmer fytocenologického zápisu podľa Mueller, Domboisa, Ellenberga (in Krížová 1995). Pokryvnosť druhov sa odhadovala v %. Na 57 skusných plochách bolo vykonaných 581 databázových záznamov a súhrnne sme zachytili 39 taxónov (*Adenostyles alliariae* (Gouan) A. Kern., *Athyrium distentifolium* Tausch ex Opiz, *Avenella flexuosa* (L.) Parl., *Bistorta major* Gray, *Calamagrostis arundinacea* (L.) Roth, *Calamagrostis villosa* (Chaix ex Vill.) J. F. Gmel, *Cardamine amara* L., *Dicranum scoparium* Hedw., *Dryopteris dilatata* (Hoffm.) A. Gray, *Eupatorium cannabinum* L., *Gentiana asclepiadea* L., *Homogyne alpina* (L.) Cass., *Hypnum cupressiforme* Hedw., *Galeobdolon luteum* Huds. emend. Holub, *Luzula sylvatica* (Huds.) Gaudin, *Lycopodium clavatum* L., *Maianthemum bifolium* (L.) F. W. Schmidt, *Plagiomnium affine* (Blandow ex Funck) T. J. Kop., *Moneses uniflora* (L.) A. Gray, *Myosotis sylvatica* subsp. *sylvatica*, *Nardus stricta* L., *Oxalis acetosella* L., *Phyteuma spicatum* L., *Pleurozium schreberi* (Brid.) Mitt., *Polytrichum formosum* Hedw., *Prenanthes purpurea* L., *Primula elatior* (L.) L., *Ranunculus platanifolius* L., *Rhytidiadelphus triquetrus* (Hedw.) Warnst., *Rubus idaeus* L., *Acetosa arifolia* (All.) Schur, *Rumex alpinus* L. 1759 non L. 1753, *Senecio germanicus* Wallr., *Sphagnum girgensohnii* Russow, *Stellaria nemorum* L., *Vaccinium myrtillus* L., *Vaccinium vitis-idaea* L., *Veratrum album* subsp. *lobelianum* (Bernh.) Arcang., *Viola biflora* L.). Názvy druhov sú uvedené podľa Marholda et al. (1998).

Hodnotenie podmienok prostredia je založené na bioindikačných vlastnostiach rastlinných druhov definovaných podľa Ellenberga et al. (1992). Každému rastlinnému druhu vo fytocenologickom zázname sa priradia indikačné čísla (ekočísla) 6 ekologických faktorov (svetlo, teplota, vlhkosť, kontinentalita, dusík, reakcia na pH, Ellenberg et al. 1992). Pre každý ekologický faktor sa vypočítajú percentuálne podiely pokryvnosti indikačných čísel, t.j. podiely súčtovej pokryvnosti druhov s rovnakou indikačnou hodnotou.

Pre analýzu vzťahov medzi vývojovým štádiom a fytocenózou sme použili viacrozmernú regresnú analýzu a zmiešaný model analýzy kovariancie. Pre účely regresnej analýzy boli kategorické premenné (vývojové štádium) prevedené na „dummy“ premenné.

## Výsledky a diskusia

### *Vplyv vývojového štádia prírodného lesa na pokryvnosť druhov fytocenózy*

Výsledky parciálnych korelačných koeficientov viacrozmernej lineárnej regresie medzi vývojovým štádiom a pokryvnosťou konkrétneho druhu po eliminácii vplyvu nadmorskej výšky vo všeobecnosti naznačujú veľmi slabú závislosť pokryvnosti konkrétneho taxónu od vývojového štádia. V štádiu dorastania sa hodnoty parciálnych korelačných koeficientov pohybujú v rozmedzí hodnôt –0.25 do 0.30, v štádiu optima v rozpätí –0.21 do 0.31 a v štádiu rozpadu od –0.23 do 0.19. Najsilnejší a signifikantný vzťah sme zistili medzi pokryvnosťou *Polytrichum formosum* Hedw. a štádiom optima, kde dosiahol parciálny korelačný koeficient hodnotu 0.31, čo znamená, že tento druh má v štádiu optima vyššiu pokryvnosť ako v ostatných dvoch štádiách. Veľmi podobnú tesnosť vzťahu sme zistili aj medzi *Dicranum scoparium* Hedw. a štádiom dorastania s hodnotou parciálneho korelačného koeficienta 0.30. Tieto výsledky však nedávajú odpoveď na otázku, či je tieto dva druhy možné považovať za indikačné druhy vývojových štádií v danej oblasti. Pre tento účel sme použili dvojfaktorovú (vývojové štádium, druh) analýzu kovariancie, ktorou sme komplexnejšie testovali interakciu vplyvu faktorov na pokryvnosť druhov fytocenózy. Kovariančnou premenou bola nadmorská výška, ktorá má nielen v danej oblasti ale i vo všeobecnosti výrazný vplyv na viacero ekologických charakteristík. Z výsledkov analýzy vyplýva, že interakcia vplyvu skúmaných faktorov je štatisticky nesignifikantná (tab. 1), t.j. že sa vo fytocenóze v danej oblasti nenachádza druh, ktorého zvýšená alebo znížená pokryvnosť by poukazovala na príslušnosť k vývojovému štádiu prírodného smrekového lesa.

K podobným záverom dospeli v jedľovo-bukových pralesoch Ujházy et al. (2005) a Šamonil a Vrška (2007), ktorí zistili len veľmi slabý náznak zmeny druhov fytocenózy v závislosti od vývojového štádia. Kým nesignifikanté rozdiely vo fytocenóze medzi štádiami jedľovo-bukového pralesa Šamonil a Vrška (2007) vysvetľujú výskytom malých medzier v poraste a teda nedostatočným priamym osvetlením podrastu, smrekové prírodné lesy v oblasti Babej hory sa naopak vyznačujú trvalo rozpojeným zápojom (Merganič et al. 2003). V oboch prípadoch sa však jedná o vyrovnané svetlostné a mikroklimatické podmienky, ktoré dávajú predpoklad, že počas vývojových štádií týchto ekosystémov dochádza vo fytocenóze k minimálnym a štatisticky nesignifikantným zmenám.

Napriek štatisticky nesignifikantným výsledkom sme pri podrobnejšej analýze mohli pri jednotlivých druhoch podobne ako Ujházy et al. (2005) a Šamonil a Vrška (2007) pozorovať určitý náznak cyklických zmien. V podmienkach Babej hory sú tieto zmeny dvojakého charakteru. Zmeny v pokryvnosti *Polytrichum formosum* Hedw. a *Dicranum scoparium* Hedw. môžeme charakterizovať „veľkým cyklom“ (obr. 1, 2). Ich pokryvnosť je vysoká len v určitom vývojovom štádiu, kým v ďalších dvoch štádiách je výrazne nižšia ako vo vrcholovom štádiu, avšak v oboch štádiách zhruba na rovnakej úrovni. Zmeny prebiehajúce v „malom cykle“, pri ktorom sa pokryvnosť druhu kontinuálne mení, sme zistili v pokryvnosti taxónov *Avenella flexuosa* (L.) Parl., *Dryopteris dilatata* (Hoffm.) A. Gray, *Gentiana asclepiadea* L., *Hypnum cupressiforme* Hedw., *Oxalis acetosella* L., *Rubus idaeus* L., *Senecio germanicus* Wallr. a *Vaccinium myrtillus* L.. Napr. pokryvnosť taxónu *Oxalis acetosella* L. narastá od štádia dorastania k rozpadu (obr. 1). Naopak, taxón *Dryopteris dilatata* (Hoffm.) A. Gray (obr. 2) má najvyššiu po-

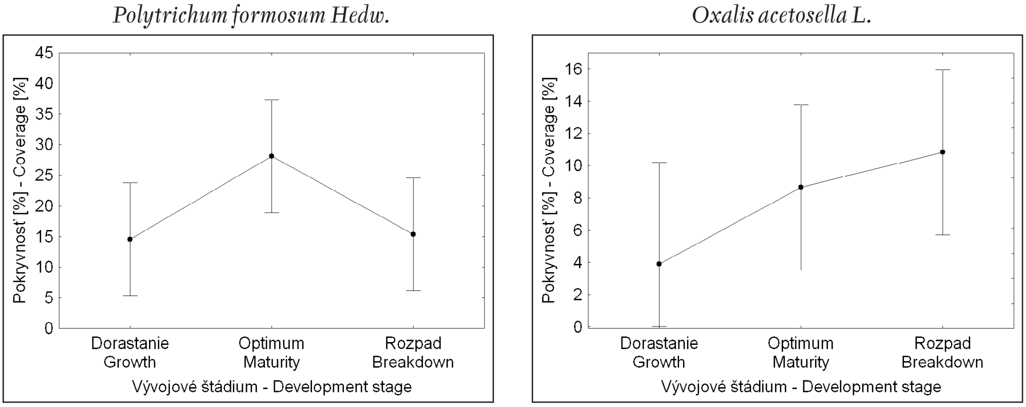
*Tab. 1: Analýza kovariancie vplyvu vývojového štádia a druhu na pokryvnosť druhov fytocenózy.*
*Tab. 1: Covariance analysis examining the influence of developmental stage and plant species on plant species coverage.*

| Faktor[1] | Povaha faktora[2] | Počet stupňov voľnosti[3] | Priemer štvorca odchýlok[4] | Počet stupňov voľnosti pre reziduál♣ | Priemer štvorca odchýlok reziduálu[5] | F | Hladina spoľahlivosti[6] |
|---|---|---|---|---|---|---|---|
| Štádium×Druh[7] | Náhodný[8] | 76 | 74.29 | 2105 | 70.75 | 1.05 | 0.36 |

[1] factor, [2] nature of a factor, [3] degrees of freedom, [4] mean square error, [5] mean square error of residual, [6] significance level, [7] developmental stage×plant species, [8] random

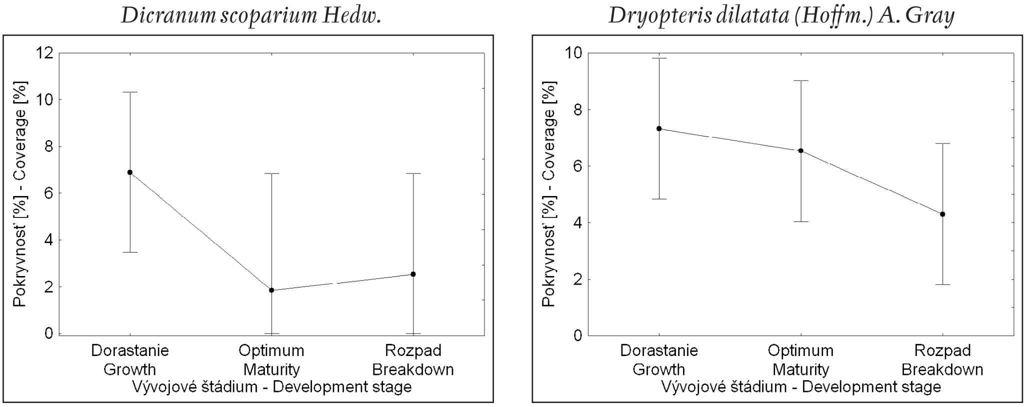
Poznámka: ♣ – počet stupňov voľnosti pre reziduál je počítaný použitím Satterthwait metódy; hladina spoľahlivosti – *95%, **99%

Note: ♣ – degrees of freedom for residual calculated by Satterthwait method; significance level – *95%, **99%



*Obr. 1: Pokryvnosť taxónov Polytrichum formosum Hedw. („veľký cyklus“) a Oxalis acetosella L. („malý cyklus“) vo vývojových štádiách prírodného smrekového lesa v NPR Babia hora (● – aritmetický priemer, ⌶ ± 1.96 · stredná chyba (IS 95 %). Výsledky analýzy sú štandardizované na priemer nadmorskej výšky ako kovariančnej premennej 1352.7 m n.m.)*

*Fig. 1: Coverage of two species Polytrichum formosum Hedw. („large cycle“) and Oxalis acetosella L. („small cycle“) in the developmental stages of natural spruce forests in National Nature Reserve of Babia hora (● – average, ⌶ ± 1.96 · mean error (95 % confidence interval). The results of the analysis are standardised for an average of covariate variable elevation equal to 1,352.7 m above sea level)*



*Obr. 2: Pokryvnosť taxónov Dicranum scoparium Hedw. („veľký cyklus“) a Dryopteris dilatata (Hoffm.) A. Gray („malý cyklus“) vo vývojových štádiách prírodného smrekového lesa v NPR Babia hora (● – aritmetický priemer, ⌶ ± 1.96 · stredná chyba (IS 95 %). Výsledky analýzy sú štandardizované na priemer nadmorskej výšky ako kovariančnej premennej 1352.7 m n.m.)*

*Fig. 2: Coverage of Dicranum scoparium Hedw. („large cycle“) and Dryopteris dilatata (Hoffm.) A. Gray („small cycle“) in the developmental stages of natural spruce forests in National Nature Reserve of Babia hora (● – average, ⌶ ± 1.96 · mean error (95 % confidence interval). The results of the analysis are standardised for an average of covariate variable elevation equal to 1352.7 m above sea level)*

kryvnosť v štádiu dorastania, ktorá postupne k štádiu rozpadu klesá. Taxón *Rubus idaeus* L. sa vyznačuje najnižšou pokryvnosťou v štádiu optima a najvyššou v štádiu rozpadu.

### *Vplyv vývojového štádia prírodného lesa na podiel druhov fytocenózy s rovnakou indikačnou hodnotou podmienok prostredia podľa Ellenberga et al. (1992)*

Obdobným spôsobom ako v predošlom prípade sme pomocou viacrozmernej lineárnej regresie otestovali vplyv vývojového štádia na podiel druhov s rovnakou indikačnou hodnotou. Hodnoty parciálnych korelačných koefi-

*Tab. 2: Analýza kovariancie vplyvu vývojového štádia, ekologického faktora a Ellenbergovho indikačného čísla na podiel pokryvnosti druhov s rovnakou indikačnou hodnotou.*
*Tab. 2: Covariance analysis examining the influence of developmental stage, ecological factor and Ellenberg´s indicator value on the coverage proportion of plant species with the same indicator value.*

| Faktor[1] | Povaha faktora[2] | Počet stupňov voľnosti[3] | Priemer štvorca odchýlok[4] | Počet stupňov voľnosti pre reziduál♣ | Priemer štvorca odchýlok reziduálu[5] | F | Hladina spoľahlivosti[6] |
|---|---|---|---|---|---|---|---|
| Štádium×Ekofaktor×Ekočíslo[7] | Náhodný[8] | 90 | 0.01 | 3401 | 0.01 | 0.88 | 0.78 |

[1] factor, [2] nature of a factor, [3] degrees of freedom, [4] mean square error, [5] mean square error of residual, [6] significance level, [7] developmental stage×ecological factor×Ellenberg´s indicator value, [8] random

Poznámka: ♣ – počet stupňov voľnosti pre reziduál je počítaný použitím Satterthwait metódy; hladina spoľahlivosti – *95%, **99%

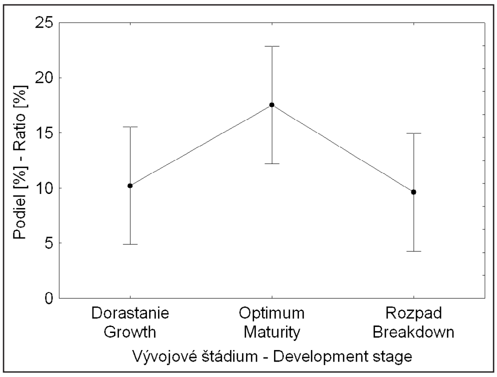
Note: ♣ – degrees of freedom for residual calculated by Satterthwait method; significance level – *95%, **99%

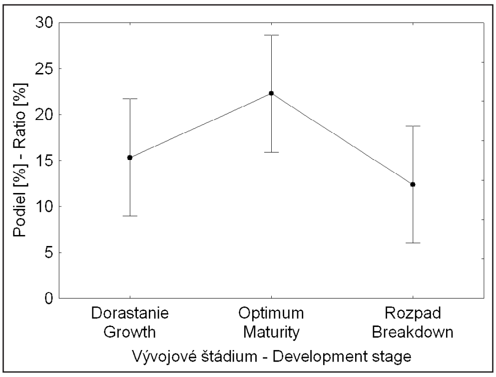
Faktor: Teplota
Factor: Temperature

Ellenbergove indikačné číslo: 2
Ellenberg´s indicator value: 2
indikátory zimy až chladu (alpínske druhy)
coldness indicators (alpine species)

Faktor: Svetlo
Factor: Light

Ellenbergove indikačné číslo: 4
Ellenberg´s indicator value: 4
tieňomilné až polotieňomilné druhy
sciophilous species





*Obr. 3: Podiel pokryvnosti druhov s rovnakou indikačnou hodnotou vo vývojových štádiách prírodného smrekového lesa v NPR Babia hora (● – aritmetický priemer, ⌶ ± 1,96 · stredná chyba (IS 95 %). Výsledky analýzy sú štandardizované na priemer nadmorskej výšky ako kovariančnej premennej 1352,7 m n.m.)*
*Fig. 3: Coverage proportion of plant species with the same Ellenberg´s indicator value in developmental stages of natural spruce forests in National Nature Reserve of Babia hora (● – average, ⌶ ± 1.96 · mean error (95 % confidence interval). The results of the analysis are standardised for an average of covariate variable elevation equal to 1.352.7 m above sea level)*

cientov sa medzi podielom indikačného čísla a vývojovým štádiom pohybovali v štádiu dorastania od −0.24 do 0.27, v štádiu optima od −0.23 do 0.31 a v štádiu rozpadu v rozpätí hodnôt od −0.21 do 0.21. Signifikantné vzťahy sme zistili medzi podielom pokryvnosti indikátorov zimy až chladu (alpínske druhy) a štádiom optima (s hodnotou parciálneho korelačného koeficienta 0.31), podielom pokryvnosti tieňomilných až polotieňomilných druhov opäť so štádiom optima (0.29) a podielom pokryvnosti druhov subkontinentálnych a štádia dorastania s hodnotou parciálneho korelačného koeficienta 0.27. Trojfaktorovou analýzou kovariancie sme komplexne otestovali vplyv faktorov: vývojové štádium, ekologický faktor a indikačné číslo na podiel pokryvnosti druhov s rovnakou indikačnou hodnotou. Z analýzy vyplýva, že v skúmanej oblasti sa ekologické podmienky indikované fytocenózou v závislosti od vývojového štádia prírodného lesa štatisticky signifikantne nemenia (tab. 2, obr. 3).

Podobnú analýzu vykonali v jedľovo-bukových pralesoch Salajka a Razula aj Šamonil a Vrška (2007), ktorí medzi vývojovými štádiami zistili štatisticky signifikantné rozdiely v pôdnej reakcii pH a pôdnom dusíku indikovaných fytocenózou.

## Záver

V predkladanej práci analyzujeme vplyv vývojového štádia prírodného lesa na fytocenózu. Analýza bola zameraná na preukázanie existencie diferenciačných druhov viažucich sa na vývojové štádiá ako aj skupiny druhov s rovnakou indikačnou hodnotou podmienok prostredia v zmysle Ellenberga et al. (1992). Z výsledkov analýzy vyplýva, že sa vo fytocenóze v danej oblasti nenachádza druh, ktorého zvýšená alebo znížená pokryvnosť by poukazovala na príslušnosť k vývojovému štádiu prírodného smrekového lesa. Najsilnejší a signifikantný vzťah sme zistili medzi pokryvnosťou *Polytrichum formosum* Hedw. a štádiom optima a medzi *Dicranum scoparium* Hedw. a štádiom dorastania. Zmeny v pokryvnosti týchto druhov môžeme charakterizovať „veľkým cyklom". Ich pokryvnosť je vysoká len v určitom vývojovom štádiu, kým v ďalších dvoch štádiách je výrazne nižšia, avšak v oboch štádiách zhruba na rovnakej úrovni. Určitý, aj keď štatisticky nesignifikantný náznak cyklických zmien („malý cyklus") sme zistili aj v pokryvnosti taxónov *Avenella flexuosa* (L.) Parl., *Dryopteris dilatata* (Hoffm.) A. Gray, *Gentiana asclepiadea* L., *Hypnum cupressiforme* Hedw., *Oxalis acetosella* L., *Rubus idaeus* L., *Senecio germanicus* Wallr. a *Vaccinium myrtillus* L., pri ktorých sa pokryvnosť kontinuálne mení.

Z analýzy vplyvu vývojového štádia prírodného lesa na podiel druhov fytocenózy s rovnakou indikačnou hodnotou podmienok prostredia podľa Ellenberga et al. (1992) vyplýva, že v skúmanej oblasti sa ekologické podmienky indikované fytocenózou v závislosti od vývojového štádia prírodného lesa štatisticky signifikantne nemenia. Stav porastov v danej oblasti (trvalo rozpojený zápoj) a špecifické klimatické podmienky vytvárajú predpoklady priestorovo homogénnej fytocenózy, teda stavu, ktorý je podobný klimaxu.

## Poďakovanie

Ďakujeme Ministerstvu životného prostredia Slovenskej republiky za finančnú podporu poskytnutú pre riešenie tejto práce.

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## Summary

The presented paper analyses the effect of the developmental stages of virgin forests on their plant communities. For the analysis, we used the data from the survey of plant communities in the forest stands of national nature reserve Babia hora performed in 2002 as a part of forest inventory. The relevés were collected on 57 sample plots situated at an elevation ranging from 1,222 m to 1,503 m above sea level. The plots are equally divided between the three developmental stages of virgin forests: stage of growth, maturity, and breakdown as defined by Korpeľ (1989).

The analysis was aimed at examining if any of the recorded plant species or a group of species with the same indication value of environmental conditions according to Ellenberg et al. (1992) can be used for differentiating between the developmental stages of virgin forests.

The relationship between plant communities and developmental stages was examined using multiple regression analysis and a mixed model of covariance analysis, while for the purposes of regression analysis the categorical variables (developmental stage) were transformed into dummy variables.

The results revealed that in the studied area no plant species can be accounted for differentiating between the developmental stages. The significant relationships were found between the coverage proportion of *Polytrichum formosum* Hedw. and stage of maturity, and between *Dicranum scoparium* Hedw. and stage of growth. The changes in the coverage proportion of these two plant species can be characterised by a „large cycle“. Their coverage is high in one developmental stage, while in the other two stages their coverage is markedly lower, though in both stages at a similar level. A trend toward cyclical changes, which was however not found significant, was also observed in the coverage of *Avenella flexuosa* (L.) Parl., *Dryopteris dilatata* (Hoffm.) A. Gray, *Gentiana asclepiadea* L., *Hypnum cupressiforme* Hedw., *Oxalis acetosella* L., *Rubus idaeus* L., *Senecio germanicus* Wallr. a *Vaccinium myrtillus* L. Unlike for the two above mentioned species, the coverage of these species is changing continually characterising a „small cycle“.

The environmental conditions were assessed indirectly using Ellenberg´s indicator values of six ecological factors: light, temperature, humidity, continentality, nitrogen, and soil reaction (Ellenberg et al. 1992). For each ecological factor, precentage coverage proportions of indicator values, i.e. the proportions of cumulative coverage of species with the same indicator value, were calculated. The analysis of the influence of developmental stages on the proportion coverage of plant species with the same Ellenberg´s indicator value showed that environmental ecological conditions are not affected by the developmental stage of virgin forest. The status of forest stands (permanently released canopy) and specific climate conditions of Babia hora are preconditions for spatially homogenous plant communities.

Ekológia (Bratislava) Vol. 28, No. 4, p. 424–437, 2009
doi:10.4149/ekol_2009_04_424

# ASSESSMENT OF THE IMPACT OF DEER ON THE DIVERSITY OF YOUNG TREES IN FOREST ECOSYSTEMS IN SELECTED LOCALITIES OF THE CZECH REPUBLIC

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Abstract

Merganič J., Russ R., Beranová J., Merganičová K.: Assessment of the impact of deer on the diversity of young trees in forest ecosystems in selected localities of the Czech Republic. Ekológia (Bratislava), Vol. 28, No. 4, p. 424–437, 2009.

The presented paper analyses deer impact on tree species and height diversity of young trees in three localities of the Czech Republic. The selected localities consisted of one fenced part, where deer has been excluded for a long time; and an unfenced part with free access to deer. The data were collected within the framework of the statistical forest inventory based on systematic sampling of the inventoried area. Tree species and height diversity were quantified using ten diversity indices. The analysis revealed that excessive deer densities pose a threat both to tree species and height diversity of young trees. Higher negative deer impact on tree species diversity can be expected on acidic sites, while height diversity is more sensitive to deer influence on fertile sites. However, if deer densities are low and do not reach the carrying capacity of the site, deer presence has no effect on tree species and height diversity of young trees.

*Key words*: deer impact, tree species diversity, height diversity, diversity indices, fencing

## Introduction

The impact of deer on forest ecosystems has long been considered (Putman, 1986; Gill, 1992a, b; Gill, Beardall, 2001; Rooney, 2001; Côté et al., 2004). Deer affects not only vegetation, but also other animal groups, invertebrates, soil, nutrient cycling, etc., while the effects may

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be both direct and indirect (Putman, 1986; Rooney, 2001; Rooney, Waller, 2003), as well as positive and negative (Putman, 1986; Reimoser et al., 1999; Gill, Beardall, 2001; Crête et al., 2001; White et al., 2004). To state if and how deer presence influences a particular element of woodland biodiversity depends on deer density, species-specific and site-specific factors, and on the relationship between deer and the examined element (e.g. competition, predation) (Putman, 1986; Stewart, 2001; White et al., 2004). However, in general excessive deer densities usually exert an adverse overall effect on biodiversity (Putman, 1986; Rooney, 2001; Côté et al., 2004; White et al., 2004; Carson et al., 2005), although some plants, invertebrates and animals may benefit from it (White et al., 2004).

Forest regeneration representing the future forest stand is most vulnerable to damage caused by deer when considering the tree layer of the ecosystem (Potvin et al., 2003). In general, three major types of direct damage can occur: browsing, bark stripping, and fraying trees with antlers (Gill, 1992a, b; Motta, 1996). Although all these effects are classified as damage to a particular tree, considering the forest stand as a unit they do not always have to have a negative influence on its tree species diversity. The overall impact is related to many different factors, e.g. to timing and intensity of damage, to tree species composition of the understorey, to what species is affected by deer, to the susceptibility of the tree species to damage etc. (Putman, 1986; Gill, 1992a; Reimoser et al., 1999; Côté et al., 2004). If e.g. deer reduces the proportion of the most abundant tree species in understorey/regeneration, this can increase tree species diversity (Helle, Aspi, 1983; Gill, 1992b), while damaging rare and vulnerable tree species can cause species loss from the ecosystem (Martin, Daufresne, 1999).

Furthermore, damage by deer has an influence on the forest structure (Putman, 1986; Gill, Beardall, 2001; Rooney, 2001; Rooney, Waller, 2003; Côté et al., 2004). When deer densities are sufficiently great, the vertical habitat complexity of forest ecosystems may be reduced (Rooney, 2001). On the other hand, below a certain threshold of deer density no damage occurs and very little effect either on species composition or on woodland structure is apparent (Gill, 1992a).

As Rooney (2001) pointed out on an example in North America, in the pre-settlement period deer densities were low and regulated by weather, predators, and forest structure and composition. Human induced changes to natural woodlands brought imbalances in these relationships. Hence, it is now difficult to know what level of deer density can be expected in a sustainable forest ecosystem for any particular conditions. To obtain such information, long-term exclosure experiments have been established to study the response of forest ecosystems to deer exclusion (e.g. Eiberle, 1967; Leibundgut, 1974; Ertl, 1989; König, Baumann, 1990; Pollanschütz, 1992; König, 1997; Nomiya et al., 2003; Von Oheimb et al., 2003; Stone et al., 2004; etc.). Although such studies do not provide us with the information about the desired status of the forest because deer naturally belong to forest ecosystems, they can support objective judgements of deer impact (Reimoser et al., 1999).

In the presented paper we examined an indirect influence of deer presence on young trees in the forest understorey using the data from long-term exclosures. The analysis consisted of two partial goals:

(1) to evaluate how deer influences tree species and height diversity of young trees, which is in accordance with Reimoser et al. (1999), who suggested that total tree density, spe-

cies composition, and height structure are the main indicators of deer impact on forest regeneration,

(2) and to assess how site fertility affects deer impact on diversity of young trees, if at all.

Tree species and height diversity of young trees was quantified using ten most common diversity indices in order to analyse and present the reaction of each of them, which can in future help a scientist to select the most appropriate index for a particular task.

## Material and methods

### *Site description*

Within the presented work the data from three different forest regions of the Czech Republic were used: Brdská vrchovina (locality Svätá Anna), Středočeská pahorkatina (locality Libeř), and Jihomoravské úvaly (locality Ranšpurk and Cahnov). All localities consist of one fenced (control) and one unfenced part with free access to animals.

The fenced part of Svätá Anna has a size of 47.46 ha (fenced since 1970's) while the unfenced part covers 25.13 ha. Regarding the site characteristics, 60% of the experiment area is represented by acidic edaphic series (acidic oak beech forests) (UHÚL, 2001). The locality is situated at an elevation of 491 m a.s.l. , has an average annual temperature of 7.7° and a mean annual precipitation of 571 mm. Coniferous tree species, namely Norway spruce (*Picea abies* Karst.), Scots pine (*Pinus sylvestris* L.), silver fir (*Abies alba* Mill.), and European larch (*Larix decidua* Mill.), make up approximately 70% of species composition. The amount of broadleaved species, from which the most common are oak (*Quercus* sp.) and hornbeam (*Carpinus betulus* L.), is in the fenced part 10% higher than outside the exclosure. There are several deer species present in the locality: red deer (*Cervus elaphus* L.), fallow deer (*Dama dama* L.), white

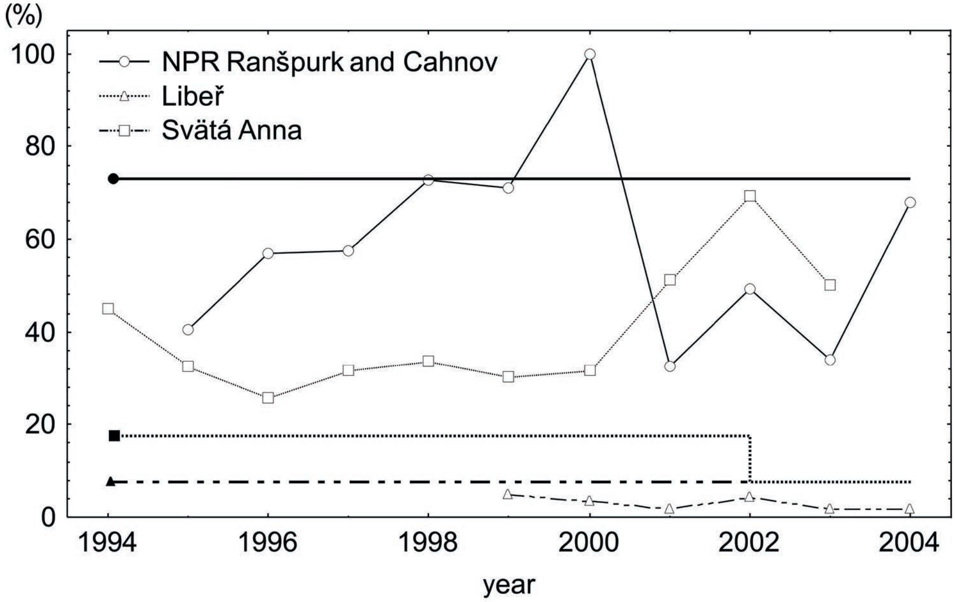

Fig. 1. Relative values of deer density (thin line) standardised in the units of roe deer (*Cervus elaphus*) density per 1,000ha and their carrying capacity (heavy line) in the examined localities.

tailed deer (*Odocoileus virginianus* Bodd.), and roe deer (*Capreolus capreolus* L.). Deer densities in the unfenced part have consistently exceeded the carrying capacity (by a factor of 3) for over a long time (Fig. 1).

The exclosure of the locality Libeř covers 28.74 ha (fenced since 1970's). The unfenced part has an area of 18.64 ha. Acidic edaphic series (acidic oak beech forest) prevail in this locality (UHÚL, 2001). The locality is situated at an elevation of 408 m a.s.l., has an average annual temperature of 8.1° and a mean annual precipitation of 614 mm. The main tree species are Norway spruce, and Scots pine. Oak and hornbeam are the most common broadleaves. In the exclosure, the proportion of broadleaved species is higher by about 10% than in the unfenced part. Within the locality, only roe deer occurs from deer species, whereby in the long term its densities have never reached the carrying capacity of the locality (Fig. 1).

The last locality consists of the two state nature reserves Ranšpurk (fenced since 1983) and Cahnov (represents unfenced control site). Both reserves are hardwood floodplains (elm hardwood floodplain) (UHÚL, 1999). The locality is situated at an elevation of 165 m a.s.l., and has a mean annual temperature of 9.7° and an average annual precipitation of 527 mm. There are only broadleaved species present in the reserves, whereby field maple (*Acer campestre* L.), narrow-leaved ash (*Fraxinus angustifolia* Wahlenb.), hornbeam, and oak are the dominant species there. The locality is characterised by the presence of roe deer, fallow deer, and red deer. In the long term, deer densities have fluctuated around the carrying capacity of the locality, usually being below it (Fig. 1).

## *Material*

In all three localities, the data were collected within the framework of the statistical forest inventory. As a sampling design systematic sampling was applied, i.e. the sample plots each of size 500 m$^2$ were established in a regular net over the inventoried area. In the locality Svätá Anna in total 69 sample plots were established, out of which 39 plots were located in the fenced area and 32 plots in the unfenced part. In Libeř 25 plots were situated in the fenced part and 24 plots in the unfenced part. In the last evaluated locality 23 and 22 sample plots were established in the reserves Ranšpurk and Cahnov, respectively.

From the total of 50 variables assessed during the inventory, we used the information about site, parent stand, species composition, frequency and height configuration of young trees in the understorey, i.e. of the trees with a minimum height of 10 cm and a maximum diameter at breast height of 6.9 cm. Within the sample plots, the young trees were assessed by stratified sub-sampling on three circle sub-plots, each with a radius of 2m (i.e. area 12.57 m$^2$).

## *Quantification of tree species and height diversity of young trees*

Tree species and height diversity of young trees in the understorey was quantified using diversity indices. From the great number of existing indices we selected those that are considered by the majority of authors (e.g. Ludwig, Reynolds, 1988; Krebs, 1989) as the most suitable for numerical assessment of diversity in examined populations. These indices can be divided into three groups:

- indices of species richness describing community diversity on the base of number of species

$$N0 = S$$ (Hill, 1973) [1]

$$R1 = (S-1)/\ln(N)$$ (Margalef, 1958) [2]

$$R2 = S/\sqrt{N}$$ (Menhinick, 1964) [3]

- indices of species heterogeneity combining species richness and evenness

$$\lambda = 1 - \sum_{i=1}^{S} w_i^2$$ (Simpson, 1949) [4]

$$H' = -\sum_{i=1}^{S} (w_i \ln(w_i))$$ (Shannon, Weaver, 1949) [5]

$N1 = e^{H'}$ (Hill, 1973) [6]

$N2 = 1/\lambda$ (Hill, 1973) [7]

- indices of species evenness that quantify the equitability of species in community

$E1 = H`/ln(S)$ (Pielou, 1975, 1977) [8]

$E3 = (e^{H`}-1)/(S-1)$ (Heip, 1974) [9]

$E5 = ((1/\lambda)-1)/(e^{H`}-1)$ (Hill, 1973) [10]

where *S* is the number of tree species or height classes; *N* is the number of individuals; and $w_i$ is the relative abundance of tree species or height class *i*.

Species heterogeneity and species evenness indices were calculated using the species or height class proportion determined from the number of individuals. Considering the interpretation of the above-mentioned indices, higher values always indicate higher diversity.

### *Statistical analysis of deer impact on diversity of young trees in understorey*

The analysis of deer impact on diversity of young trees is based on simple or hierarchical analysis of variance. The examined factors were deer presence (fenced and unfenced part) and site fertility described by integrated edaphic series (acidic versus fertile sites). The Tukey test was used to test the significance of the differences between the levels of the particular factor.

For correct interpretation of the obtained results the homogeneity of the experiment must be ensured. However, in long-term experiments such as the one presented here, it is difficult to meet this requirement, because of the problems to maintain the conditions at the starting level (Gill, Beardall, 2001), and hence, to exclude the influence of side factors. Therefore, prior to analysis the homogeneity of the conditions inside and outside the exclosures was tested using the principles of contingency tables, analysis of variance, and regression analysis.

## Results

### *Examining homogeneity of site and stand conditions inside and outside exclosures*

The homogeneity test examined seven potential factors that can affect the interpretation of the results: slope, type of terrain, aspect, edaphic series as a parameter integrating site conditions, age of the parent stand, stand closure, and the proportion of the side light. According to the results of the homogeneity test, no significant differences in site and stand conditions between the fenced and the unfenced part of the locality Svätá Anna were detected.

In Libeř significant differences between the fenced and the unfenced part were found in the edaphic series and stand closure, while the exclosure consists of more fertile sites, and its forest stands are characteristic of lower closure. Due to these facts, more detailed analyses were made to examine the relationship between stand closure and the values of tree species and height diversity for the fenced and the unfenced part and the acidic and fertile edaphic series separately. In the acidic edaphic series outside the exclosure significant relationship was detected between closure and the indices *S, R1, H´,* $\lambda$*, N1* and *N2*. The situation was similar in the case of height diversity indices, where the homogeneity test detected significant relationship between closure and all indices except *R2*. In the next step, the values of the significant indices were corrected for the influence of closure, i.e. new values of the indices were obtained from the regression for the average value of stand closure in the fenced part. Further analyses were performed with these corrected values. To eliminate the differences between the edaphic series, indices were compared only within one edaphic series.

The test of the last locality comprising the Nature Reserves Ranšpurk and Cahnov revealed the identity between them. Overall, after the correction of the data in Libeř we can state that the homogeneity between the compared parts of the localities is assured.

### *Deer impact on tree species diversity of young trees*

In the first examined locality Svätá Anna, the ratio between the broadleaved and coniferous tree species in species composition of young trees is balanced, whereby in the exclosure broadleaved species and in the unfenced part coniferous species slightly prevail. Young trees in the fenced part are mainly silver fir, Norway spruce, oak, and hornbeam, but there is also a considerable amount (approx. 10%) of fast-growing broadleaves. In the unfenced part, spruce, and hornbeam are the most common species among the young trees. The factor "fence" was detected as a significant factor influencing tree species diversity of young trees, while the factor "edaphic series" was not proven to have a significant influence in this locality (Table 1). It means that significant differences were found between the fenced and the unfenced part, but not between the acidic and fertile edaphic series within a certain part (Fig. 2). Significant deer impact was found for nine out of ten diversity indices (Table 1). As shown in Fig. 2, higher values of indices were obtained for the fenced part in all three components of diversity (richness, heterogeneity, evenness). In the acidic edaphic series, significant differences between the fenced and the unfenced part were detected in all examined cases, while in the fertile edaphic series this was found only for $H'$ and $\lambda$. Considering the sensitivity of indices to deer impact, *R1*, $\lambda$ and *E5* were the most sensitive indices from the groups of species richness, heterogeneity, and evenness, respectively.

In the case of Libeř, among young trees broadleaved species prevail both in the fenced and the unfenced part, although in the exclosure their proportion is by about 10% higher. Species composition of young trees consists mainly of Norway spruce, oak, hornbeam and fast-growing broadleaves (silver birch – *Betula verrucosa* Ehrh., wild cherry – *Cerasus avium* (L.) Moench., European aspen – *Populus tremula* L., rowan – *Sorbus aucuparia* L., black alder – *Alnus glutinosa* Gaertn.).

Regarding tree species diversity, it was necessary to evaluate deer impact separately for each edaphic series using the corrected values as described in section 3.1. Analysis of variance did not detect a significant influence of the factor "fence", and hence of deer presence, on any of the examined diversity indices. Fig. 2 presents that in the acidic edaphic series higher tree species diversity was observed in the unfenced part. In contrast, the results for the fertile edaphic series show an opposite trend, since all indices except *S* and *R1* indicate

Table 1. Hierarchic (Svätá Anna) and simple (Ranšpurk and Cahnov) analysis of variance examining the influence of the factor “fence” (i.e. deer impact) and the factor “integrated edaphic series” on the values of tree species diversity.

| Diversity index | Factor | Locality | | | |
|---|---|---|---|---|---|
| | | Svätá Anna<br>*p* | | State Nature Reserves Ranšpurk and Cahnov<br>*p* | |
| *S* | fence | 0.0006 | *** | 0.0731 | |
| | edaphic series*fence | 0.6178 | | | |
| *R1* | fence | 0.0002 | *** | 0.7652 | |
| | edaphic series*fence | 0.7281 | | | |
| *R2* | fence | 0.1002 | | 0.0054 | ** |
| | edaphic series*fence | 0.1703 | | | |
| *H´* | fence | 0.0000 | *** | 0.9197 | |
| | edaphic series*fence | 0.2971 | | | |
| | fence | 0.0000 | *** | 0.9030 | |
| | edaphic series*fence | 0.1693 | | | |
| *N1* | fence | 0.0001 | *** | 0.8232 | |
| | edaphic series*fence | 0.5533 | | | |
| *N2* | fence | 0.0001 | *** | 0.5518 | |
| | edaphic series*fence | 0.5699 | | | |
| *E1* | fence | 0.0000 | *** | 0.5285 | |
| | edaphic series*fence | 0.1524 | | | |
| *E3* | fence | 0.0001 | *** | 0.2974 | |
| | edaphic series*fence | 0.1684 | | | |
| *E5* | fence | 0.0000 | *** | 0.3230 | |
| | edaphic series*fence | 0.1488 | | | |

Note: Significance level: ***$p < 0.001$=99.9%, **$p < 0.01$=99% and *$p < 0.05$=95%; grey part indicates fields where the analysis was not possible

higher diversity in the exclosure. The most sensitive indices were *E3* and *N2* regardless of the edaphic series.

In Ranšpurk and Cahnov species composition of the young trees is very similar to that of the parent stand, while field maple highly prevails. Narrow-leaved ash, and hornbeam are also frequent among the young trees. This last locality is characteristic of only one edaphic

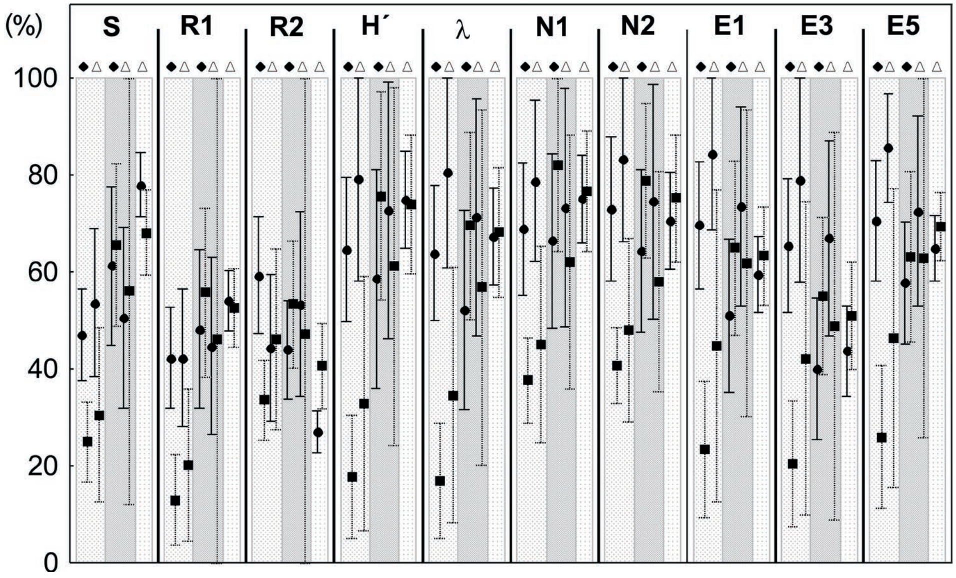

Fig. 2. Relative values of tree species diversity of young trees in the fenced and the unfenced part of each examined locality described by diversity indices *S* - *E5*.
(Legend: Arithmetic mean: ● – fenced part, ■ – unfenced part; Interval, inside which the real mean should occur with probability 95% ($t_{\alpha/2}$ standard error of mean): ⊥ – fenced part, ⊥ – unfenced part; Integrated edaphic series: ◆ – acidic, △ – fertile; Locality: ░ – Svätá Anna, ▒ – Libeř, ⬚ – State Nature Reserves Ranšpurk and Cahnov)

series present in both reserves Ranšpurk and Cahnov. Hence, within the analysis only the influence of the factor "fence" was tested. As shown in Table 1, the only significant difference between the fenced and the unfenced part was found for *R2*. In seven out of ten cases, higher values of diversity indices were calculated for the unfenced part (Fig. 2). The most sensitive indices to deer impact were *R2* and *S*.

### *Deer impact on height diversity of young trees*

In Svätá Anna, a significant influence of deer was detected for all indices except *R2* (Table 2). The factor "edaphic series" had a significant effect only on *N1*, although *S*, *R1*, *H´* and *N2* were close to critical value $p = 0.05$ (Table 2). As presented in Fig. 3, higher average values of the majority of indices were observed in the fenced part regardless of the edaphic series. The analysis revealed significant differences between the fenced and the unfenced part only in the fertile edaphic series in the case of *S*, *R1*, *H´*, λ, *N1* and *N2*. The most sensitive indices were *S*, *H´* and *E1* representing richness, heterogeneity, and evenness, respectively.

T a b l e 2. Hierarchic (Svätá Anna) and simple (Ranšpurk and Cahnov) analysis of variance examining the influence of the factor “fence” (i.e. deer impact) and the factor “integrated edaphic series” on the values of height diversity of young trees.

| Diversity index | Factor | Locality | | | |
|---|---|---|---|---|---|
| | | Svätá Anna<br>*p* | | State Nature Reserves Ranšpurk a Cahnov<br>*p* | |
| *S* | fence | 0.0053 | ** | 0.0000 | *** |
| | edaphic series*fence | 0.0641 | | ▨ | ▨ |
| *R1* | fence | 0.0121 | * | 0.0000 | *** |
| | edaphic series*fence | 0.0740 | | ▨ | ▨ |
| *R2* | fence | 0.9750 | | 0.0804 | |
| | edaphic series*fence | 0.3149 | | ▨ | ▨ |
| *H´* | fence | 0.0043 | ** | 0.0000 | *** |
| | edaphic series*fence | 0.0551 | | ▨ | ▨ |
| | fence | 0.0072 | ** | 0.0000 | *** |
| | edaphic series*fence | 0.1216 | | ▨ | ▨ |
| *N1* | fence | 0.0110 | * | 0.0000 | *** |
| | edaphic series*fence | 0.0355 | * | ▨ | ▨ |
| *N2* | fence | 0.0321 | * | 0.0000 | *** |
| | edaphic series*fence | 0.0514 | | ▨ | ▨ |
| *E1* | fence | 0.0139 | * | 0.0000 | *** |
| | edaphic series*fence | 0.2829 | | ▨ | ▨ |
| *E3* | fence | 0.0283 | * | 0.0003 | *** |
| | edaphic series*fence | 0.3487 | | ▨ | ▨ |
| *E5* | fence | 0.0252 | * | 0.0000 | *** |
| | edaphic series*fence | 0.3029 | | ▨ | ▨ |

Note: Significance level: ***$p < 0.001$=99.9%, **$p < 0.01$=99% and *$p < 0.05$=95%; grey part indicates fields where the analysis was not possible

In Libeř, the analysis did not detect any significant impact of deer on height diversity of young trees. According to the results shown in Fig. 3, in the acidic edaphic series higher height diversity in the exclosure was obtained only from two indices *S* and *R1*. When examining the fertile edaphic series, the values of *R2, H´,* λ*, N1, N2* and *E3* suggest that the fenced part has higher height diversity of young trees (Fig. 3). The most sensitive indices in the acidic and fertile part were *S* and *N2*, respectively.

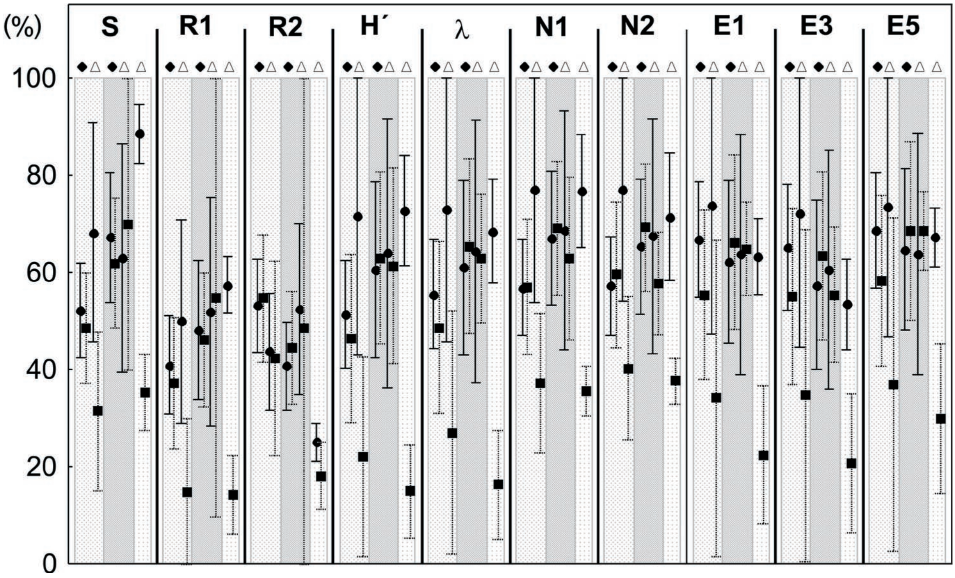

Fig. 3. Relative values of height diversity of young trees in the fenced and the unfenced part of each examined locality described by diversity indices *S* - *E5*.
(Legend: Arithmetic mean: ● – fenced part, ■ – unfenced part; Interval, inside which the real mean should occur with probability 95% ($t_{\alpha/2}$ standard error of mean): ⊥ – fenced part, ⊥ – unfenced part; Integrated edaphic series: ◆ – acidic, △ – fertile; Locality: ░ – Svätá Anna, ▒ – Libeř, ⬚ – State Nature Reserves Ranšpurk and Cahnov)

The analysis of the last locality consisting of the two reserves Ranšpurk and Cahnov revealed a significant deer impact on height diversity of young trees in all cases except *R2*, which is close to the critical value (Table 2). Fig. 3 documents that all indices exhibited higher values of height diversity in the fenced part Ranšpurk. Regarding the sensitivity of indices, the results coincided with those obtained for Svätá Anna, i.e. the most sensitive indices were *S*, *H´* and *E1*.

## Discussion

From the three examined localities, locality Svätá Anna was found to be influenced by deer the most. This locality has been under intense pressure due to the excessive deer density for the last ten years (Fig. 1), which has affected both the tree species and height diversity of young trees. The influence of deer was apparent in all components of tree species diversity, i.e. in richness, heterogeneity, and evenness, while the values of all the calculated indices were higher in the exclosure (Table 1, Fig. 2).

The analysis showed that due to high deer pressure the risk of the loss in tree species diversity is higher on the acidic sites, whereas the height diversity of young trees is more threatened on the fertile sites (Fig. 2). Considering tree species diversity, fertile sites provide suitable conditions for more tree species (Barnes et al., 1998), and hence, deer selection does not have such profound effects on tree species richness as on acidic sites (Fig. 2). Côté et al. (2004) also reported that deer impact is larger in low-productivity habitats.

On the other hand, fertile sites are characteristic by denser and more diverse ground vegetation (Barnes et al., 1998). Therefore, such sites are more attractive to deer, also because of the higher nutrient content of the plants from the fertile sites (Gill, 1992a). If deer density is too high, as it is in Svätá Anna, where the average deer density during the last ten years was 3 times higher than its carrying capacity, the height diversity of young trees is more affected on fertile sites (Fig. 3).

In addition, excessive deer pressure usually leads to a reduction in palatable plant species and the spread of unpalatable ones or those tolerant to deer damage, mainly ferns and graminoids (Kirby, 2001; Rooney, 2001; Carson et al., 2005). This shift in plant composition may inhibit tree regeneration (Kirby, 1990; Kirby, 2001; Côté et al., 2004). Selective feeding of deer is also documented by the values of *E1-E5* that are significantly lower in the unfenced part regardless of site fertility (Fig. 2). Similar findings were observed in the evaluation of the height diversity of young trees (Fig. 3), because most browsing by deer usually occurs at an intermediate level between ground and full reach (Gill, 1992a; Ellenberg et al., 2001; Côté et al., 2004).

Locality Libeř is the opposite example to Svätá Anna, as here deer densities have always been far below the carrying capacity (Fig. 1). Therefore, the analysis of the deer impact on forest ecosystem could hardly detect any significant influence of deer. In several cases (e.g. index *S*), a higher diversity was found in the unfenced part. Similar results were reported also for ground vegetation by Rambo and Faeth (1999), Von Oheimb et al. (2003), Stone et al. (2004), etc. Helle and Aspi (1983) showed that moderately grazed areas by reindeer had the highest species richness, since this grazing pressure created space for the plants, which would be in ungrazed areas out-competed (Putman, 1986). Higher diversity of the unfenced part may also be caused by management practices, as e.g. Reimoser and Gossow (1996) documented that ungulates impact depends on the applied silvicultural system. Although the management analysis detected that the same management prescription was to be applied in both fenced and unfenced parts of the locality Libeř, according to the information obtained from the forest enterprise the forest stands inside the exclosure were in reality left to natural development until the 1990s, while in the unfenced part clear-cutting, shelterwood and selection systems were applied. This fact can explain the obtained results, because the usual management practices are aimed at maintaining a certain species composition, which does not have to coincide with the natural status. Hence, tree species diversity can be higher in the unfenced managed part than inside the exclosure, because intermediate frequencies of disturbance increase species diversity when compared with no or very low disturbance frequencies (Barnes et al., 1998). In addition, clear-cutting can also lead to increased tree species diversity through promoting pioneer tree species (Barnes et al., 1998; Brokaw, Lent, 1999).

The State Nature Reserves Ranšpurk and Cahnov represent the locality in which the silvicultural human influence can be excluded. Nonetheless, the only significant difference in tree species diversity between the fenced part Ranšpurk and the unfenced part Cahnov was detected for the index *R2* (Table 1, Fig. 2). In addition, although in general Ranšpurk has a higher species richness than Cahnov, *R2* indicates the opposite. This is due to the character of the index, as it accounts for the size of population and is sensitive to a wide range of individuals' frequencies. It was the frequency of young trees that caused the opposite reaction of *R2* as expected, since in the unfenced part Cahnov the frequency of young trees is only half their frequency in Ranšpurk. Interestingly, a higher tree species heterogeneity and evenness was also observed in the unfenced Cahnov (Fig. 2). This may be explained by the fact that in Ranšpurk, where higher species richness was detected, new tree species can occur thanks to deer exclusion, but their proportion in species composition is small.

Considering height diversity of young trees, the results of the analysis revealed significantly higher height diversity in the exclosure Ranšpurk than in Cahnov (Table 2, Fig. 3). This corresponds with other works, which reported a simplified vertical structure of forest stands as a result of deer impact (e.g. Gill, 2000; Rooney, 2001).

## Conclusion

While assessing deer impact on forest ecosystems, it should not be forgotten that deer have always been a considerable component of their natural biodiversity. Therefore, within the sustainable forest management the efforts should focus on keeping deer densities at sensible levels, which depend on site-specific and species-specific conditions. The results of our analysis support this statement, since only excessive deer densities pose a threat to tree species and height diversity of young trees. Higher negative deer impact on tree species diversity can be expected on acidic sites, while height diversity is more sensitive to deer influence on fertile sites. With low deer densities that do not reach the carrying capacity of the site, no significant effect of deer presence on tree species and height diversity of young trees has been detected.

*Translated by the authors*

*Acknowledgements*

This work was performed within the framework of the project RTD funded by Czech Ministry of Environment. We are grateful to R. Williams for his helpful review comments of the manuscript.

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Environmental Management (2010) 46:908–919
DOI 10.1007/s00267-010-9506-2

# Forest Naturalness: Criterion for Decision Support in Designation and Management of Protected Forest Areas

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Received: 29 January 2009 / Accepted: 14 May 2010 / Published online: 20 June 2010

**Abstract** The article analyses the possibilities of developing an integrated indicator and a model of the assessment of forests naturalness using the data from the database of mountainous spruce forests situated in the Western Carpathians of Slovakia. The article presents two variants of such a model, one based on discriminant analysis, while the second one using an additive approach. The analysis of the data from mountainous spruce forests revealed significant indicators of forest naturalness degree: the arithmetic mean of the ratio between crown length and tree height, the deadwood volume, the coverage of grasses, the coverage of mosses and lichens, and the aggregation index. In addition, the coefficient of variation of tree diameters was included in the final model, since its presence in the model had a positive influence on the correctness of the classification of the forest naturalness degree. The correctness of the classification of the proposed discriminant model was 74.5%. For the additive model, the ranges of the values of the integrated indicator were defined for every degree of forest naturalness by taking into account the error ranges of the arithmetic mean values and the percentiles of the values in individual degrees of forest naturalness. The overall correctness of the classification with the additive model was 63.4%. In the second step, the scheme how to apply the classification model of the forest naturalness degree in the decision-making process of designating as a forest protected areas was proposed. In this scheme, the degree of forest naturalness is considered as a basic criterion for the determination of nature-conservation value of forest ecosystems. As further decision-making criteria we identified the possibility to restore, or the possibility to improve the naturalness of less natural forest ecosystems, which are designated as protected; the occurrence of the endangered species; and the occurrence of other natural values.

**Keywords** Indicators and models of forest naturalness · Evaluation of forest naturalness · Forest protected area · Decision-making support in forest utilisation

## Introduction

Forest naturalness, or more precisely a degree of forest naturalness, is a significant indicator of the intensity of human interventions in forest ecosystems, i.e., it specifies the extent of human influence (Cluzeau and Hamza 2007). There exist a number of less or more detailed classifications of forest naturalness with a common feature that the scale begins with the forests in the original state representing the highest degree of forest naturalness and ends up with man-made forests (Welzholz and Bürger-Arndt 2004).

Naturalness is a pan-European indicator of sustainable forest management (SFM) belonging to the set of criteria and indicators for sustainable forest management (No. 4.3) proposed within the framework of the Ministerial Conference on the Protection of Forests in Europe (MCPFE (Ministerial Conference on the Protection of Forests in Europe) 2002). In this context, forests are divided into

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forests *undisturbed by man*, which encompass forests with least human interventions; *modified natural forests*, *semi-natural forests* and *plantations* (*productive* and *protective*), which cover man-made (artificial) forests.

According to the Global Forest Resources Assessment 2010 (FAO 2007), forests are distinguished into *primary forests* defined as naturally regenerated forests of native tree species with no clearly visible indications of human activities and with not significantly disturbed ecological processes; *other naturally regenerated forests* which are also regenerated naturally but the indications of human activities are clearly visible; and *planted forests*, where the trees established through planting or seeding prevail.

The degree of forest naturalness is assessed through various indicators, mainly: nativeness of species and genotypes, differentiation of stand structure (e.g. diameter frequency distribution, vertical and age structure, occurrence of deadwood, natural regeneration of forests and coverage of ground vegetation), as well as the existence and extent of human influence in particular forest ecosystems (e.g. occurrence of timber felling and forest re-establishment and the applied methods, soil scarification, existence of forest roads, recreational activities, grazing, forest damage). (e.g. McComb and Lindenmayer 1999; Müller-Starck 1996; Peterken 1996; Scherzinger 1996; Frank 2000).

Some European countries assess forest naturalness at a sample plot level within the framework of their national forest inventories. However, such an assessment provides summary information on individual degrees of forest naturalness only at national or regional levels.

Since the assessment of forest naturalness is very demanding from the points of methodology, applied techniques and funding, its realisation is reasonable if this indicator is an essential element in a specific decision-making process. In forestry, forest naturalness is of the greatest significance in the decision-makings that deal with the designation of forests as protected areas, and in the second step that determine the need and the urgency of their management (cultivation, tending) in such a way, which will secure the protection of their biological diversity and/or of other natural values. For these purposes, it is required to perform detailed surveys of forest naturalness focusing particularly on such forest ecosystems which are the subject of decision-making processes whether they are to be declared protected areas or not.

In the case of protected forest areas, at least of those with the highest status of protection, one expects them to be very close-to-nature, with almost no human influence. It is assumed that only in such forests the natural developmental cycle, the adequate tree species composition, the age structure, and other components of forest naturalness have been preserved or have recovered. The maintenance and enhancement of these features should be the primary goal of nature conservation (Welzholz and Bürger-Arndt 2004; Bartha and others 2006).

From the perspective of nature conservation, the forests *undisturbed by man* are of great value, particularly if they are large compact forest areas. Such forests can serve as reference areas, where natural ecological processes can be studied; and can also contribute to the development of close-to-nature forest management methods (MCPFE 2007). In the conditions of Central Europe, such forests occur only scarcely and hence, are very precious. Owing to functioning natural ecological processes, they should be left to self-regulating processes without any human interventions.

Because of the above-mentioned reasons it is required to know the actual degree of forest naturalness in protected areas, and in the forest ecosystems, which have the potential for being protected, since it can be taken as an objective criterion for decision-making about forest use and consequently about forest management (Hoerr 1993; Schmidt 1997). This is a generally applicable requirement and a need for achieving the optimal and the most effective use of forestland.

Hence, our goal was to prepare and propose a generally applicable method for the derivation of an integrated indicator and a model of forest naturalness degree. Our requirement was to obtain unit values of the indicator and the variability of such a magnitude, that the differences between the individual degrees of forest naturalness would be significant. In order to examine the practical applicability of the proposed method, it was developed for a specific case of forest ecosystems located in a spruce altitudinal vegetation zone (hereinafter called SVZ).

We selected this type of ecosystems due to two reasons. First, this forest community is very valuable with significant ecological and social functions including nature-conservation functions (Korpeľ 1989). Secondly, extensive national and international scientific activities (e.g. S4C Initiative, Mountain Research Initiative, International Scientific Committee on Research in the Alps ISCAR) have been carried out in the forests of SVZ. During the scientific works performed in the years 1999–2002, a large amount of data suitable for solving our task was gathered, since the need for the research of naturalness of forest ecosystems was respected already in the process of data acquisition.

## Materials and Methods

### Database

The SVZ is the highest altitudinal vegetation zone (VZ) with forest and tree-like vegetation in Slovakia (Fig. 1). It

**Fig. 1** Distribution of spruce vegetation zone over the area of Slovakia. Similar forest types occurred in other European mountain ranges (e.g. Alpine and Carpathian regions in Romania, Ukraine, Poland, Austria, Germany, Switzerland, France and Czech Republic); *Legend*: Boundary of the Slovak Republic, Forest stand area, Spruce vegetation zone

is situated above spruce-beech-fir VZ, in which typical well-grown management forests occur. Forests in SVZ reach smaller heights (at an upper zone line the trees are dwarfed), and have a character of protective forests. At higher altitudes, SVZ merges into mountain pine VZ. The SVZ is located at an elevation from 1,250 to 1,550 (1,600) m above sea level. Total annual precipitation is 1,100–1,600 mm per year. Vegetation season lasts from 70 to 100 days, and mean annual temperature is approximately 2–4°C.

Apart from Norway spruce (*Picea abies* (L.) K.), other tree species, namely rowan (*Sorbus aucuparia* L.), European larch (*Larix decidua* Mill.), Arolla pine (*Pinus cembra* L.), Sycamore (*Acer pseudoplatanus* L.), European beech (*Fagus sylvatica* L.), Silesian willow (*Salix silesiaca* Willd.), Carpathian birch (*Betula pubescens ssp. carpatica* Willd.), Great sallow (*Salix caprea* L.), and Dwarf mountain pine (*Pinus mugo ssp. mugo* Turra), also occur in these stands with a proportion of up to 30%, approximately. The age, diameter, and height structure of the forests in SVZ should be highly (horizontally and vertically) diversified to ensure the fulfilment of important ecological functions.

Almost identical forest types spread over the whole Alpine and Carpathian region, less frequently they can be found also in other European mountain ranges (Palearctic habitat 42.21: Alpine and Carpathian subalpine spruce forests).

Empirical material was collected in permanent research plots (PRP) by preferential and non-random sampling. This sampling is common in ecological studies, which are mainly aimed at ecological gradients, because it covers a broader range of vegetation variability (Rolecek and others 2007). The PRPs were established as circle plots of a size of 100–1,000 $m^2$ in order to meet the prerequisite that a minimum of 25 trees occurr within each plot. The PRPs were localised using the global positioning system (GPS). The methodological intention was to establish PRPs in such a manner that detailed information about the natural and stand conditions (inclusive of forest naturalness) of forests in SVZ could be obtained (Table 1). In the process of the methodology preparation, indicators suitable for the description of the state of structurally differentiated forests that were assumed to be related to the forest naturalness degree were identified and proposed.

To assess and to characterise natural and stand conditions of PRPs, a lot of information was collected: forest type, soil type and crown closure were determined; basic mensurational parameters were measured; and the developmental phase and the degree of forest naturalness of the stand inside the PRP were assessed. Tree crown length was calculated as the difference between tree height and height to crown base. For age analysis, a required number of core bores were collected. The coverage of ground vegetation (grasses, herbs, mosses, lichens, subshrubs, and shrubs), the conditions for natural regeneration of spruce, and existing natural regeneration were also assessed.

In total, 122 PRPs were established. Table 1 presents the summary information about the data structure with regard to natural and stand conditions of the PRPs, in which they were established (forest eco-region, group of forest types, degree of forest naturalness, and elevation). As shown in Table 1, the category Natural forests having 94 sample plots is the most abundant, while the category of Man-made forests has the lowest frequency.

### *Classification of Forest Naturalness for Parametrisation of Classification Model*

The classification of forest stands into degrees of forest naturalness was based on the categorisation of Zlatník (1976) (Table 2). The assessed degree of forest naturalness resulted from the detailed, though subjective evaluation of the forest status. Naturalness was assessed as a rate of human influence on a forest on the base of visual features that indicate human interventions (inclusive of forest management), which affect tree species, spatial and age structure (Fleischer 1999) of forests in SVZ. Each PRP was assigned one degree of forest naturalness from the scale A to G (Zlatník 1976), which were further aggregated into three degrees: Primeval forest, Natural forest, Man-made forest (Moravčík and others 2003; Moravčík and others 2005; Moravčík 2007a, b) prior to data processing due to the insufficient number of plots in the degrees of the finer scale from A to G. For further processing and evaluation of information from 122 PRPs, the database system "Mountainous forests" was created in the environment MS Access 2000.

**Table 1** Data structure with regard to natural and stand conditions

| Degree of forest naturalness, *n*/% | | | | | | | |
|---|---|---|---|---|---|---|---|
| Primeval forest | | | Natural forest | | | Man-made forest | |
| 17/13.9 | | | 94/77.1 | | | 11/9.0 | |
| Of it the stage of | | | Of it the stage of | | | Of it the phase of | |
| Growth | Optimum | Disintegration | Growth | Optimum | Disintegration | Tending | Regeneration |
| 2 | 9 | 6 | 32 | 36 | 26 | 2 | 9 |

| Forest eco-region, *n*/% | | | |
|---|---|---|---|
| Veľká Fatra | Poľana | Nízke Tatry | Vysoké Tatry |
| 7/5.7 | 12/9.8 | 85/69.7 | 18/14.8 |

| Group of forest side types, *n*/% | | | |
|---|---|---|---|
| SP, LP sup | AcP sup | FP sup | CP |
| 84/68.9 | 22/18.0 | 9/7.4 | 7/5.7 |

| Elevation (meters above sea level), *n*/% | | | | | |
|---|---|---|---|---|---|
| Up to 1,350 | 1,351–1,400 | 1,401–1,450 | 1,451–1,500 | 1,501–1,550 | 1,551 and above |
| 14/11.5 | 21/17.2 | 29/23.8 | 32/26.2 | 19/15.6 | 7/5.7 |

**Table 2** Criteria for the classification of stands by the naturalness classes

| NC | Name | Signs of anthropic effect; signs of stand structure |
|---|---|---|
| A | Primeval forest | Without any effect of human activity |
| B | Natural forest | Appearance of primeval forest without obvious signs of anthropic activity, possible selective felling in past, natural forests affected by natural disasters left to natural development are included as well |
| C | Semi-natural forest | Natural tree species composition, altered spatial structure due to extensive human activity |
| D | Predominantly natural forest | Natural signs predominate over anthropic signs |
| E | Slightly altered forest | Forest with natural as well as anthropic signs, the latter ones prevail |
| F | Markedly altered forest | Forests only with anthropic signs but of natural appearance |
| G | Completely altered forest | Forest stand only with anthropic signs of its origin or formation |

### *Proposal of Indicators of Forest Naturalness for Classification Model*

Considering the structure and the type of data stored in the database system "Mountainous forests", a number of indicators that were assumed to be related to a degree of forest naturalness were proposed.In total, 25 different indicators of naturalness of forest ecosystems in SVZ were quantified, while tree species diversity was represented with 10 indicators, and structural diversity with 15 indicators (Table 3).

Tree species diversity was quantified with five indices of species richness, two indices of species heterogeneity, and three indices of species evenness (Table 3). The indices of species heterogeneity were calculated from the proportion of basal area of particular tree species from the total basal area in a sample plot.

The indicators of structural diversity reflect the diversity of structural elements of a forest ecosystem in horizontal and vertical directions. From 15 proposed structural indicators, two characterise vertical diversity (number of tree layers determined on the base of the sociological position of trees, and "Arten Profil" (species profile) index (Pretzsch 1996), while horizontal diversity is quantified by an aggregation index (Clark and Evans 1954). The remaining structural indicators are relatively simple and easy to be quantified, and are also related to static stability, stand density, and site quality. The average ratio of crown length to tree height, and the average ratio of tree height to tree diameter were calculated from the trees ranked in 1st to 3rd sociological layers. The indicators describing the coverage of herbs, grasses, mosses and lichens, shrubs and subshrubs; the coverage of phases describing the conditions for natural regeneration (juvenile, optimal, senile); the

**Table 3** Calculated indicators of naturalness of forest ecosystems

| Tree species diversity | | | | | Structural diversity | | | |
|---|---|---|---|---|---|---|---|---|
| Category | Indicator | Formula | Units | Reference | Indicator | Formula | Units | Reference |
| Species richness | Index N0—living trees | $N0 = S$ | DIM | Hill (1973) | Number of tree layers ($Z$) | $Z = j$ | DIM | |
| | Index N0—mosses and lichens | $N0 = S$ | DIM | Hill (1973) | Arten profil index ($A$) | $A = \sum_{i=1}^{S}\sum_{j=1}^{Z} P_{ij} \ln P_{ij}$ | DIM | Pretzsch (1996) |
| | Index N0—shrubs and subshrubs | $N0 = S$ | DIM | Hill (1973) | Aggregation index ($R$) | $R = \frac{\frac{1}{M}\cdot\sum_{i=1}^{M} r_i}{0.5\cdot\sqrt{\frac{M}{A}}}$ | DIM | Clark and Evans (1954) |
| | Index R1 | $R1 = (S - 1)/\ln(M)$ | DIM | Margalef (1958) | Coefficient of variation of tree diameter ($CV\_D1.3$) | $CV\_D1.3 = \frac{\bar{d}}{SD_d}$ | % | Šmelko (2000) |
| | Index R2 | $R2 = S/\sqrt{M}$ | DIM | Menhinick (1964) | Average ratio of crown length to tree height ($AM\_K$) | $AM\_K = \frac{\sum_{i=1}^{M}\frac{cl_i}{h_i}}{M}$ | % | Šmelko (2000) |
| Species heterogeneity | Index $\lambda$ | $\lambda = 1 - \sum_{i=1}^{S} p_i^2$ | DIM | Simpson (1949) | Average height/diameter ($h/d$) ratio ($AM\_HDR$) | $AM\_HDR = \frac{\sum_{i=1}^{M}\frac{h_i}{d_i}}{M}$ | DIM | Šmelko (2000) |
| | Index H′ | $H' = -\sum_{i=1}^{S} p_i \cdot \ln(p_i)$ | DIM | Shannon and Weaver (1949) | Coverage of grasses ($PK\_T$) | $PK\_T = p_i$ | % | |
| Species evenness | Index E1 | $E1 = H^{'}/\ln(S)$ | DIM | Pielou (1975) and (1977) | Coverage of herbs ($PK\_B$) | $PK\_B = p_i$ | % | |
| | Index E3 | $E3 = (e^{H'} - 1)/(S - 1)$ | DIM | Heip (1974) | Coverage of mosses and lichens ($PK\_M$) | $PK\_M = p_i$ | % | |
| | Index E5 | $E5 = ((1/\lambda) - 1)/(e^{H'} - 1)$ | DIM | Hill (1973) | Coverage of shrubs and subshrubs ($PK\_K$) | $PK\_K = p_i$ | % | |
| | | | | | Coverage of juvenile regeneration stage ($PK\_JS$) | $PK\_JS = p_i$ | % | |
| | | | | | Coverage of optimum regeneration stage ($PK\_OS$) | $PK\_OS = p_i$ | % | |
| | | | | | Coverage of senile regeneration stage ($PK\_SS$) | $PK\_SS = p_i$ | % | |
| | | | | | Coverage of natural regeneration ($PK\_NR$) | $PK\_NR = p_i$ | % | |
| | | | | | Deadwood volume ($MOD$) | $MOD = \frac{\sum_{i=1}^{m} v_i}{A/10000}$ | $M^3$/ha | |

$S$ number of species; $M$ number of individuals, number of living trees in a sample plot; $m$ number of deadwood individuals (stumps, lying deadwoood); $p_i$ probability, proportion of $i$th species or category in a sample plot; $p_{ij}$ proportion of trees of $i$th tree species in $j$th stand layer; $Z$ number of layers—stories of the stand; $r_i$ distance between $i$th tree and its closest neighbour (m); $A$ area of a sample plot ($m^2$); $d$ tree diameter; $SD_d$ standard deviation of tree diameters in a sample plot; $cl$ crown length; $h$ tree height; $v$ volume

coverage of natural regeneration were visually estimated in the field and are given in relative values (%) (Moravčík and others 2005).

### *Classification Model of a Degree of Forest Naturalness*

Two variants of the classification model of forest naturalness were proposed, one based on the principles of discriminant analysis, while the second one uses an additive approach to derive the integrated indicator of the degree of forest naturalness.

The discriminant model is derived as an application of multivariate statistical analysis, so-called predictive discriminant analysis (Cooley and Lohnes 1971; Huberty 1994; StatSoft 1996; Merganič and Šmelko 2004). Its role is to classify the sampling unit on the base of several quantitative variables into one of the pre-defined qualitative classes, in our case into one of the three degrees of forest naturalness. Using the data from the database, three discriminant equations were derived, each for one degree of forest naturalness. These discriminant equations serve for the classification of an evaluated forest stand into one of the three degrees of forest naturalness.

Secondly, we proposed an integrated indicator of forest naturalness degree. This indicator belongs to complex indicators that combine several diversity components into a single value. The indicator is based on an additive approach, while the partial components are given in real measurement units. Mathematical formula of the integrated indicator of the degree of forest naturalness (*IISP*) is as follows:

$$IISP = ID_1 + ID_i + \ldots ID_n \tag{1}$$

where *ID* partial indicator of the degree of forest naturalness

### *Data Adjustment to Meet the Needs for the Derivation of the Classification Model of the Degree of Forest Naturalness*

The relation between a diversity indicator and an area, for which the indicator was assessed, is known from a number of theoretical and practical studies. Due to the varying area of our sample units, we tested the relationship between the values of the partial indicators of forest naturalness and the area of the sample plot. The analysis revealed that 9 indicators (R1, R2, the average ratio of crown length to tree height, the average ratio of tree height to tree diameter, coverage of herbs and grasses, coverage of juvenile and senile phases and deadwood volume per hectar) had a significant relationship with the plot area ($p < 0.05$). This result is logical and is mainly coupled with the effect of the developmental stages. The significant influence of the developmental stage on the indicators of forest naturalness was found in 16 out of 25 cases. Since the plots were distributed among the developmental stages, the varying area of the sample plots should not have a negative influence on subsequent analyses and on the creation of the classification model of the degree of forest naturalness. On the contrary, the estimates of the average values and the variation of the indicators derived from tree data (the average ratio of crown length to tree height, aggregation index etc.) are even more representative, since they always represent a similar group of trees (approx. 25 trees).

As can be seen in Table 1, the numbers of the plots (PRP) in individual degrees of forest naturalness, as well as the numbers of the plots in individual developmental stages (growth, optimum, disintegration) within the naturalness degrees are imbalanced. Due to this and the above-stated facts, it was required to equalise the number of the sampling units in individual developmental stages and in individual degrees of forest naturalness. The missing plots were added by random replication of the existing sample plots using bootstrap technique (Chernick 2008; Yu 2003) until the number of the plots in the most abundant developmental stages was reached in other stages, too. In this way, the numbers of the plots in less abundant developmental stages and 1st, 2nd, and 3rd degree of naturalness were set to 9, 36, and 9 plots, respectively.

## Results

Two different variants of the integrated complex indicator and the model of the degree of forest naturalness were proposed, one as a discriminant model, while the other one as an additive model.

### Discriminant Model

From a great number of the examined combinations of the indicators (see Methods for the list), the best results of the correct classification of the degree of forest naturalness were obtained using the combination of the following six indicators: the arithmetic mean of the ratio between crown length and tree height (*AM_K*), the deadwood volume (*MOD*), the coverage of grasses (*PK_T*), the coverage of mosses and lichens (*PK_M*), the aggregation index (*R*), and the coefficient of variation of tree diameters (*CV_D*1.3). The general formula of the final discriminant model looks as follows:

$$\begin{aligned}\text{Discriminant score}_{\,j} = AM\_K \cdot b_{j1} + MOD \cdot b_{j2} + PK\_T \\ \cdot b_{j3} + PK\_M \cdot b_{j4} + R \cdot b_{j5} \\ + CV\_D1.3 \cdot b_{j6} + b_{j7}\end{aligned} \tag{2}$$

where: *J* 1st to 3rd degree of forest naturalness

The values of the regression coefficients in individual discriminant equations are given in Table 4.

The classification of the degree of forest naturalness is performed in several steps. First, the discriminant score of each naturalness degree (1–3) is calculated from the particular discriminant equation using the real values of the partial indicators. An evaluated location, a stand, or in our case a sample plot, is assigned such a degree of forest naturalness, for which the calculated discriminant score is a maximum.

The results of the classification matrix of the parameterisation data set are presented in Table 5. As can be seen in this table, the overall correctness of the classification of the degree of forest naturalness using the proposed discriminant model is 74.5%. The highest probability of correct classification is in marginal classes (degrees 1 and 3),

**Table 4** Values of regression coefficients of the derived discriminant model

| Partial indicator of a degree of forest naturalness | Regression coefficient | Degree of forest naturalness | | |
|---|---|---|---|---|
| | | 1 | 2 | 3 |
| Arithmetic mean of crown length/tree height ratio (*AM_K*) [%] | $b_1$ | 1.2521 | 1.1154 | 0.9108 |
| Deadwood volume (*MOD*) [$\mathrm{m}^3$/ha] | $b_2$ | 0.0306 | 0.0139 | 0.0058 |
| Coverage of grasses (*PK_T*) [%] | $b_3$ | −0.0290 | 0.0070 | 0.0059 |
| Coverage of mosses and lichens (*PK_M*) [%] | $b_4$ | 0.1708 | 0.1132 | 0.0693 |
| Aggregation index (*R*) | $b_5$ | 36.7123 | 33.5378 | 30.8615 |
| Coefficient of variation of tree diameter (*CV_D*1.3) [%] | $b_6$ | −0.0348 | −0.0615 | −0.0723 |
| Absolute coefficient | $b_7$ | −73.8718 | −57.0578 | −40.3190 |

while the lowest probability is in the middle class (degree 2, 68.5%).

Following Table 6 presents the statistical characteristics of the model. According to the values of Fischer *F* and Wilks‘ Lambda statistics we can, with 99.9% probability, say that the proposed discriminant model is highly significant. The Willks‘ Lambda can be interpreted in the following manner: if its value is close to 0, the model is appropriate; if, on the other hand, the value approaches 1, the model is not suitable. The partial Lambda values given in the third column of Table 5 provide us with the information about the contribution of each independent variable to the discrimination of the dependent variable. Five out of six selected indicators are significant, which means that their contribution to the discrimination of the degree of forest naturalness is significant. Although the sixth indicator, the coefficient of variation of tree diameters, was insignificant, its presence in the model improved the classification. The indicators *AM_K* and *MOD* have the largest influence on the discrimination of the degree of forest naturalness.

In order to explain the classification graphically, the canonical analysis was applied to the data set. Figure 2

**Table 5** Classification matrix of the discriminant model

| Degree of forest naturalness | Correct classification in% | Degree of forest naturalness according to the model | | | |
|---|---|---|---|---|---|
| | | 1<br>Number of plots | 2 | 3 | Total |
| 1 | 85.2 | 23* | 4 | 0 | 27 |
| 2 | 68.5 | 15 | 74[a] | 19 | 108 |
| 3 | 94.4 | 0 | 1 | 17* | 18 |
| Total | 74.5 | 38 | 79 | 36 | 153 |

[a] Indicates the cases with correctly classified degree of forest naturalness

**Table 6** Statistic characteristics of the discriminant model

| Discriminant model | | | |
|---|---|---|---|
| Number of variables: 6 | | | Number of groups: 3 |
| Wilks’ Lambda: 0.43676 | | | $F_{(12,290)} = 12.401$*** |
| Input variables | | | |
| Indicator | Wilks’ Lambda | Partial Lambda | $F_{(3,935)}$ 95%**, 99.9%*** |
| Arithmetic mean of crown length/tree height ratio (*AM_K*) [%] | 0.587 | 0.744 | 24.944*** |
| Deadwood volume (*MOD*) [$\mathrm{m}^3$/ha] | 0.491 | 0.889 | 9.062*** |
| Coverage of grasses (*PK_T*) [%] | 0.469 | 0.932 | 5.314** |
| Coverage of mosses and lichens (*PK_M*) [%] | 0.465 | 0.940 | 4.608** |
| Aggregation index (*R*) | 0.458 | 0.953 | 3.580** |
| Coefficient of variation of tree diameter (*CV_D*1.3) [%] | 0.442 | 0.988 | 0.862 |

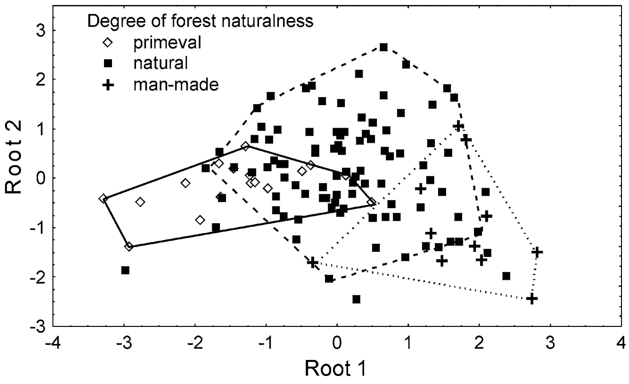

**Fig. 2** Graphical interpretation of the classification of forest naturalness degree with the discriminant model using canonical analysis; *Legend*: Degree of forest naturalness: primeval, natural, man-ma

shows the position of the groups of the sample plots with the same degree of forest naturalness and their approximate borders. From this figure it is obvious that the marginal categories of naturalness degrees have the highest probability of correct classification because their overlap with the neighbouring class is the smallest.

### Additive Model

The partial indicators in the additive model are the same as in the discriminant model, i.e. the arithmetic mean of the ratio between crown length and tree height (*AM_K*), the deadwood volume (*MOD*), the coverage of grasses (*PK_T*), the coverage of mosses and lichens (*PK_M*), the aggregation index (*R*), and the coefficient of variation of tree diameters (*CV_D*1.3).

The significance of the model was tested by single-factor analysis of variance. The analysis revealed significant differences between the average values of *IISP* of the degrees of forest naturalness (the whole model $F_{(2,\ 150)} = 21.849^{***}$, Tukey test). Figure 3 presents the graphical interpretation of the model. The range of *IISP* values was divided between the degrees of forest naturalness using the weighted approach, taking into account the error ranges of the average values of *IISP* and the percentiles of the values in every degree of forest naturalness. The objects, e.g. the stands, with the *IISP* values exceeding the value of 267 represent primeval forests; the *IISP* values in the range from 182 to 267 indicate that the forests are natural, while the values of *IISP* below 182 classify the objects as man-made forests.

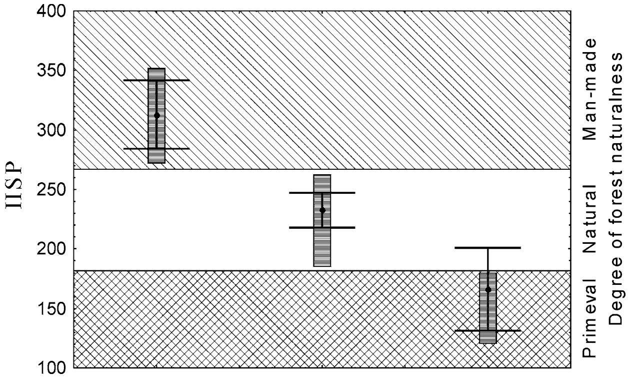

**Fig. 3** Intervals of the integrated indicator of forest naturalness (IISP) specified for the three degrees of forest naturalness (primeval, natural, man-made forests); *Legend*: percentile 26–74% = 48% of values, 95% confidence interval (1.96′ standard error), mean

The correctness of the model classification was determined on the base of the categorisation of individual plots into the degrees of forest naturalness. The overall correctness of the classification using *IISP* is 63.4%. The individual degrees of forest naturalness 1, 2, and 3 were correctly classified in 74%, 56%, and 89% of cases, respectively.

### Comparison of the Models

The results of the classification of the forest naturalness degree indicate that both variants of the classification model have a similar probability of the correct classification of the assessed object into the forest naturalness degree. The discriminant model behaves better, since its probability of correct classification is by approximately 11% higher than the probability of the additive model. Higher efficiency of the discriminant model is evident mainly in the proportion of correct classifications in 1st and 2nd degrees of forest naturalness. From the point of practical applicability, the additive model is simpler to use, but considering the current capacity of computers, it is also not difficult to apply the discriminant model in the form of a small computer program.

## Discussion

### Strengths and Weaknesses of the Approach

As we already stated in the introduction, the knowledge about the degree of naturalness of forest ecosystems is of great importance. Its objective assessment is essential in the decision-making process dealing with forest utilisation and subsequent forest management. According to Hoerr (1993) and Schmidt (1997), naturalness is the most significant and widely applied criterion for the evaluation of nature conservation, and serves as a key tool in analyses and as a support in planning nature conservation measures. Unfortunately, the assessment of the degree of forest naturalness lacks the application of the complex objective procedures and methods not only in Slovakia, but also in other countries. This situation results from the facts that research has not provided the practice with any suitable methodological mechanisms that would enable its scientifically based and statistically provable determination. The same fact has been reported by Bartha and others (2006) who mentioned that in the last decades, a number of authors developed procedures for the assessment of forest naturalness. However, in all these schemes subjective elements have been included. The assessed values of the indicators depend partially on the expert judgement and partially on their estimation. In addition, the experts make decisions, which attributes are to be assessed and what their weight is. The classification of forest naturalness proposed by Zlatník (1976) for Slovakia is also primarily based on subjective expert evaluation of the extent of human influence on forests (Table 2).

In Slovakia, several authors dealt with the evaluation of forest naturalness in protected areas using typological surveys (Šmídt 2002; Glončák 2007; Viewegh and Hokr 2003; Bublinec and Pichler 2001; Polák and Saxa 2005). These works are characterised by insufficiently complex evaluation of forest naturalness, since the authors primarily assess the suitability of tree species composition. For example, Glončák (2007) identified areas which require active management of forest ecosystems in protected areas by comparing real tree species composition with model using GIS tools. The disadvantage of this method is a high level of subjectivity needed for the development of the model of natural tree species composition. On the other hand, precise distribution of the values of naturalness of tree species composition in GIS environment is a practical advantage of this method.

In Slovakia, the proposal of the network Natura 2000 was based on the assessment of qualitative attributes of forest ecosystems using numerical quantifiers (Šmelko ex Polák and Saxa 2005; Šmelko and Fabrika 2007). However, this system assessed also features which were not directly connected to forest naturalness (e.g. forest health status, adverse external influences), and when evaluating the majority of attributes, artificial securing of forest status needed from the point of nature conservation was accepted. Hence, this system was more likely aimed at the assessment of nature conservation values than at naturalness of ecosystems. The final proposal of the network Natura 2000 is currently widely criticised, partly also for insufficient consideration for forest naturalness.

In contrast to the above-mentioned methods, our proposal is based on more precise data gathering methods, it deals with exclusive relationship with forest naturalness, and allows to account for the specifications of particular biotopes. And above all, it presents the proposal of mathematical and statistical assessment, formulation and presentation of results.

The developed model is easily applicable in practice and its application does not require intensive material and technical background. The applicability of the model for the classification of the degrees of forest naturalness has already been successfully tested on independent data (see Merganic and others 2009). The method is applicable outside SVZ or even outside Slovakia. In any other conditions, appropriate indicators of forest naturalness need to be selected, data need to be gathered, and the model needs to be re-parameterised. The coupling of the model with statistical inventory and GIS tools can enable the creation of detailed maps of naturalness of forest ecosystems. Such information is important for planning as well as for practical application of nature conservation measures. The model is a powerful tool for objectifying the assessment and the evaluation of the development of forest ecosystems within monitoring schemes.

It is important to realize, that some important indicators of forest naturalness, such as preservation of original genetic diversity or preservation of original gene pool of main tree species, are not included in the model. It is mainly due to low availability of necessary data and extremely demanding methods of their gathering. Therefore, in case of having “natural stand structure”, our method can evaluate forest stands established from the planting stock of non-native provenance. However, in reality, such cases should be rare, because the formation of natural stand structure is time-consuming and complicated, and majority of artificially planted forests had not enough time to develop to this state. Yet, old forest stands established from non-native planting stock and tended with methods imitating natural processes, could the model evaluate as natural. To summarise above mentioned, the inclusion of an indicator related to the gene pool of main tree species would improve the model, but limited availability of needed data makes this improvement just theoretical at the moment.

In Slovakia, 57.1% of forests are currently included in any type of the protected areas within the national and/or European network of protected areas (Moravčík and others 2008). The proportion of their area has been growing continuously in spite of the fact that current protected areas cover to a great extent altered forest ecosystems, where the restoration of natural biodiversity is not feasible or requires active management.

Nevertheless, there exists a group of supporters of passive nature conservation, who promote self-regulation also of the forest ecosystems with a very low degree of forest naturalness. Such an approach results in a large-scale breakdown causing the destruction of the protected element, the creation of hardly reforestable clearings, and the decrease of the required functional effectiveness of these forests.

Due to these reasons, we present the proposal for objective evaluation of the degree of forest naturalness in such a way that can be used as a basis for efficient application of differentiated methods of utilisation and subsequent forest management. Assessment Guidelines for Protected and Protective Forests and Other Wooded Land in Europe (MCPFE 2003) can be regarded as one tool for differentiated management of protected forests. In Guidelines, three classes of forests, in which biodiversity is the main management objective, were defined. Class 1.1 comprises the forests where no active direct human interventions can take place. In class 1.2, only minimum human interventions are permitted. Class 1.3 comprises the forests designated for biodiversity conservation through active management.

According to Greguš (1989), any forests can fulfil required functions best in such conditions, which correspond to the status of forests not influenced by humans. The better the approximation of such an untouched forest stand structure, the more likely it is that the forest is able to develop solely by means of its own self-regulating processes. Therefore, the basic goal of the classical concept (close-to-nature) of forest silviculture should be the preservation, enhancement, or restoration of the functionally effective forest stand structure similar to natural and primeval forests. This goal is inter alia of great economic importance, since it gives a manager the possibility to diminish the treatments to minimum, and to meet the required goals very efficiently and with minimum negative influences on nature and environment. In this context, the proposed assessment of forest naturalness has good preconditions for its application in the process of determining the need and the urgency of management measures in the scope of developing more efficient close-to-nature forest silviculture.

## Application Areas

### *Application of the Classification Model in the Decision-Making Process About the Designation and Management of Forest Ecosystems in the Spruce Vegetation Zone (SVZ)*

As we already stated in the introduction, currently there are no objective and widely applied methods for the determination of the degree of forest naturalness and for the decision-making whether the forest can or cannot be designated as a protected area. In this decision-making process, the naturalness degree is the most significant criterion. Hence, we suggest using the proposed methodology of the integrated indicator of forest naturalness in order to determine this degree of forest naturalness. The majority of scientists who deal with this issue recognise the close relationship between the degree of forest naturalness and the nature-conservation value of forest ecosystems. This knowledge is reflected in all relevant documents that deal with the assessment of protected forest areas. Owing to this, the degree of forest naturalness has a clear position in the decision-making process about the designation of the forest ecosystem a protected area. The higher its naturalness indicated by the degree of naturalness is, the more legitimate it is to designate it as a protected area. Apart from that, there tends to be the rule that the increasing naturalness of the ecosystem increases the degree of its protection. The forest ecosystems in the closest-to-nature state are the most precious, and in addition, they are also the most capable of existing by means of their own intrinsic regulating processes. Due to this, these ecosystems should be protected to the highest degree.

Apart from naturalness, the declaration scheme also accounts for the conservation of endangered species according to the Convention on Biological Diversity. If threatened species survive in unnatural ecosystems, it is required to conserve also these ecosystems by applying such a management that secures their preservation and consequently also the preservation of endangered species. A similar approach is applied in the case of *other natural values*. If, from any reasons, an ecosystem with a lower degree of forest naturalness is considered to be designated as a protected area, the necessity to protect it should be thoroughly explained, and the possibility to reconstruct it to a closer-to-natural forest ecosystem should be analysed and validated.

Following the above-stated facts, the decision-making process about the designation a protected forest area can be visualised as presented in Fig. 4.

Although this decision-making algorithm is generally valid, there exist several exceptions. It can happen that also ecosystems with a high degree of forest naturalness can disappear, since their conservation is in an unfavourable state. On the other hand, some unnatural ecosystems with a great nature conservation value, which do not require any management for their maintenance, can also exist. Such cases are, however, exceptional, and do not negate the given connections and relations between the degree of forest naturalness of forest ecosystems, their nature-conservation values, and eventually the necessity of the differentiated measures to be realised by a man with the aim of maintaining their stability. In the decision-making process of designating protected areas in such specific cases as described above, it is required to reach a consensus, based on objective justification, on the designation of less natural areas as protected, or on the application of inevitable correction measures in order to improve the condition of forests ecosystems that are natural to a high degree, but are disappearing.

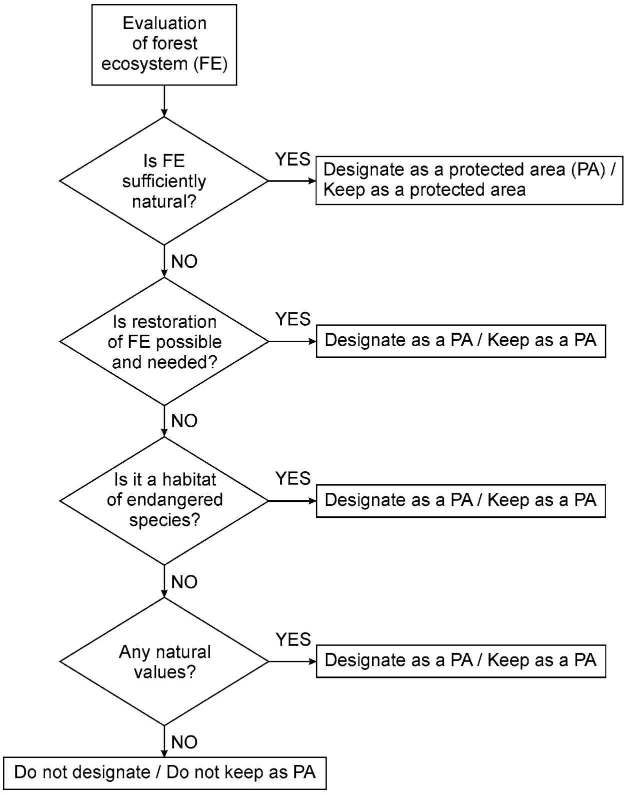

**Fig. 4** Decision-making process about the designation as a protected forest area

## Conclusion

The presented methodology for the evaluation of forest naturalness on the base of the selected indicators of tree species and structural diversity is an objective tool that can support decision-making process concerning the use of particular forest ecosystems for biodiversity conservation within the scope of nature and landscape conservation. It can not only contribute to the knowledge of the actual state of forest naturalness in protected areas, but can also provide decision-makers with the support in taking right actions aimed at enhancing the state, or in deciding to change the use of forests with low natural values (naturalness, biodiversity). The currently proposed methodology, if applied within the practical forest management, can lead to the improvement of ecological stability of forests and landscape.

Although the approach has already included several aspects of forest naturalness, it can be further enhanced by taking into account other components, e.g genetic diversity…? The coupling of the model with statistical inventory and GIS tools can enable the creation of detailed maps of naturalness of forest ecosystems. Such information can further improve planning and practical application of nature conservation measures.

**Acknowledgments** This publication was written thanks to the financial support of OP Research and development for the project "Centre of Excellence for Adaptive forest ecosystems" (ITMS: 26220120006) co-financed from the European Regional Development Fund. The authors express their gratitude to the whole scientific and technical staff of Forest Research Institute in Zvolen, who participated in the collection of data in permanent research plots established within the scope of the project "Research of the methods of mountain forest management following the principle of sustainable development" in the years 1999–2002. We mainly acknowledge Ing. Jozef Vladovič, PhD. and Ing. Vladimír Šebeň, PhD. for their cooperation in the selection and the primary classification of selected permanent research plots into degrees of forest naturalness.

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**Beskydy**, 17 (2004): 137-142
Ediční středisko MZLU v Brně

# ZÁSOBA ODUMRETÉHO DREVA V NPR BABIA HORA

**K. Merganičová, J. Merganič, J. Vorčák**

## Abstract

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The work presents the results from the survey of the volume of dead wood that can be observed in the forest stands of the national nature reserve Babia hora. The values are derived from 57 sample plots, each of size 500 $m^2$. The results showed that the expected average volume of dead wood lies with 68% probability in the range 144.62 $m^3$/ha ± 19.81 $m^3$/ha, or 56.95% ± 8.95% from the volume of living trees. The analysis of the factors influencing the amount of dead wood revealed that the development stage has a statistically significant impact on its volume, with the largest accumulated amount of dead wood in the stage of breakdown and the lowest in the stage of maturity. Elevation was found to be a statistically significant factor influencing its absolute volume, but it does not affect its relative amount, i.e. the proportion of dead wood from the living stand volume, due to the significant relationship between elevation and stand volume of living trees.

**Keywords**: dead wood, mountain spruce forest, Babia hora

## Úvod a problematika

Pohľad lesníkov na mŕtve-moderové drevo sa v súčasnosti mení. V minulosti sa chápal každý odumretý strom ako potencionálny zdroj nákazy pre les, napr. z dôvodu premnoženia podkôrneho hmyzu a iných „škodcov“ lesa. Prevádzkové opatrenia boli zamerané na spracovanie a speňaženie každého kusa dreva, ktorý sa v lese vyskytoval (Mössmer 1999). Dnes pri presadzovaní prírode blízkeho hospodárenia v lese význam moderového dreva v lesnom hospodárstve narastá, keďže najnovšie vedecké poznatky dokumentujú jeho dôležitosť pre biodiverzitu lesných ekosystémov (Müller a Schnell 2003), cyklus živín (Lexer *et al.* 2000) ako aj prirodzenú obnovu lesa najmä v extrémnych horských a severských podmienkach (Mai 1999, Hofgaard 1993).

Podľa najnovších pokynov pre prevádzku je preto odporúčané ponechať časť moderového dreva v lese, napr. Ammer (1991) odporúča ponechať vždy cca 5 – 10 $m^3$ dreva na hektár. Möller (1994) zase tvrdí, že v porastoch by malo ostať 5% z celkovej porastovej zásoby, čím by sa zabezpečil prirodzený vývoj daného lesného spoločenstva.

Na všetky potenciálne funkcie odumretého dreva vplýva nielen jeho výskyt v poraste, ale aj množstvo a dimenzie odumretých častí stromov a ich stupeň rozkladu (Hagan a Grove 1999). Keďže hospodárske lesy sú ovplyvnené činnosťou človeka, najvhodnejšími objektami na skúmanie množstva a stavu moderového dreva, ktoré sa v priebehu života lesa v poraste akumuluje, sú človekom nenarušené územia. V našej práci sme sa zamerali na národnú prírodnú rezerváciu Babia hora v Oravských Beskydách. Cieľom tejto práce bolo nielen kvantifikovať zásobu odumretého dreva na území NPR, ale aj zistiť, ktoré faktory ovplyvňujú jeho množstvo v poraste. Z možných potencionálnych vplyvov sme hodnotili vplyv vývojového štádia a nadmorskej výšky.

## Metodika a popis oblasti

Údaje použité v tejto práci pochádzajú z komplexnej inventarizácie NPR Babia hora vykonanej v roku 2002 (Merganič *et al.* 2003). NPR Babia hora patrí orograficky do sústavy vonkajších Západných Karpát, časti Oravských Beskýd, do komplexu horského masívu Babej hory. Národná prírodná rezervácia zaberá celkovo 503.94 ha a je umiestnená na západných, južných a juhozápadných svahoch Babej hory v nadmorskej výške 1100 až 1725 m n. m. (Korpeľ 1989). Geologické podložie je tvorené zo súvrství rozličných flyšových hornín. Hlavnými pôdnymi predstaviteľmi sú litozem, kambizem a podzol. Priemerné ročné teploty dosahujú vo vrcholových polohách 2°C a 4°C v nižších polohách a priemerný ročný úhrn zrážok je 1400 mm.

Lesné porasty sú tvorené prevažne smrekom obyčajným (*Picea abies* L.), vtrúsene sa vyskytujú jarabina vtáčia (*Sorbus aucuparia* L.), jedľa biela (*Abies alba* Mill.) a buk lesný (*Fagus sylvatica* L.). Les vystupuje približne do nadmorskej výšky 1500 m n. m. Nad touto hranicou lesa nastupuje pásmo kosodreviny vystriedané v najvyšších partiách NPR alpskými lúkami.

V rámci inventarizácie zalesneného územia NPR Babia hora sme okrem tradičných dendrometrických charakteristík lesného porastu zaznamenávali a hodnotili aj zásobu moderového dreva. Celkovo sa založilo 57 kruhových skusných plôch o rozlohe 500 m$^2$ tak, aby boli plochy rovnomerne rozdelené medzi tri vývojové štádiá (dorastanie – optimum - rozpad podľa Korpeľa 1989) a štyri výškové kategórie, t.j. do 1260 m, 1260 – 1360 m, 1360 – 1460 m a nad 1460 m n. m. V každej výškovej kategórii a v každom vývojovom štádiu sa založilo po 5 skusných plôch (okrem najvyššej výškovej kategórie, kde sa založili len 4 skusné plochy v každom štádiu, pretože táto kategória tvorí plošne nepatrnú časť záujmovej oblasti a bolo tu tiež problematické nájsť a vylíšiť jednotlivé vývojové štádiá).

Celkovo bolo pomeraných 2846 ks stromov. Z tohto počtu bolo 1738 ks živých stromov, 414 ks odumretých stojacich stromov (suchárov), 225 ks pňov a 469 ks ležiacich odumretých stromov vedených pod názvom ležanina. Na stojacich mŕtvych stromoch sa merala hrúbka $d_{1.3}$ a výška. Na odumretých ležiacich kmeňoch sa zaznamenávala hrúbka v polovici kmeňa $d_{½}$ a dĺžka, ale len tej časti kmeňa, ktorá spadá do skusnej plochy. Na pňoch sa merala hrúbka $d_{0.3}$.

Objem stojacich suchárov sa počítal pomocou trojparametrovej funkcie – hrúbka $d_{1.3}$, výška a nepravá výtvarnica, pričom výtvarnica bola určená podľa Pollanschütza (1974) a Schielera (1988). Keďže v záujmovej oblasti je pomerne veľa stromov poškodených snehom a námrazou, ktorých terajšia nameraná výška je zväčša nižšia ako ich pôvodná, pre výpočet výtvarnice sa v takýchto prípadoch používala tzv. modelová výška stromu odvodená z výškovej krivky (viď. Merganič *et al.* 2003). Týmto spôsobom sme zamedzili podhodnoteniu skutočného objemu stromu. Objem ležaniny bol počítaný podľa Huberovej metódy a objem pňov zjednodušene ako objem valca s výškou 0.3m.

## Výsledky

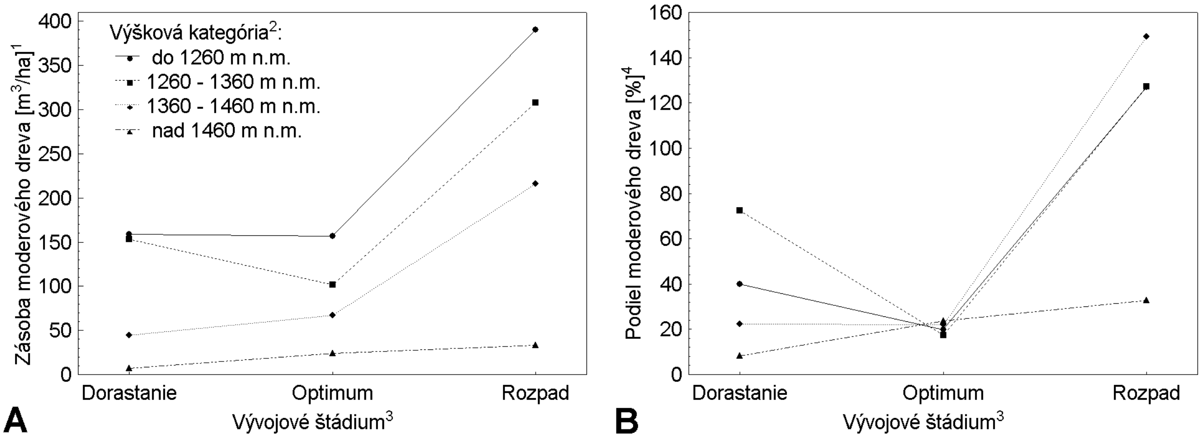
Priemerná zistená zásoba moderového dreva v NPR Babia hora bez ohľadu na nadmorskú výšku a vývojové štádium činila 144.62 m$^3$/ha ± 19.81 m$^3$/ha (stredná chyba), čo je v percentuálnom vyjadrení 56.95% ± 8.95% zo živej zásoby. S nadmorskou výškou absolútne množstvo moderového dreva klesá, keďže v prvej výškovej kategórii, t.j. v nadmorskej výške do 1260 m n. m. bola jeho priemerná zásoba 235.36 m$^3$/ha ± 44.98 m$^3$/ha, kým v druhej už len 187.75 m$^3$/ha ± 36.83 m$^3$/ha, v tretej 109.35 m$^3$/ha ± 29.78 m$^3$/ha a vo štvrtej iba 21.37 m$^3$/ha ± 5.42 m$^3$/ha. V relatívnom vyjadrení je priemerné množstvo moderového dreva vzhľadom k zásobe živého porastu v prvých troch výškových kategóriách takmer zhodné (71.24% v 1. kategórii, 72.39% v 2. kategórii a 64.54% v tretej kategórii ±

17.61%, 16.93% a 22.66%). V poslednej výškovej kategórii je však jeho podiel oveľa nižší, v priemere tvorí 21.55% ± 4.33% zo živej zásoby.

Pri porovnávaní zásoby moderového dreva v jednotlivých vývojových štádiách sme zistili, že maximálne absolútne ako aj relatívne množstvo moderového dreva sa pozorovalo v štádiu rozpadu (247.53 $m^3$/ha ± 44.62 $m^3$/ha, t.j. 113.15% ± 20.29% zo zásoby živého porastu). Štádium optima je naopak podľa očakávania charakteristické najnižšou priemernou zásobou moderového dreva, ktorá činí 90.85 $m^3$/ha ± 16.63 $m^3$/ha alebo 20.54% ± 2.45% zo živej zásoby. Aj keď priemerné absolútne množstvo moderového dreva v štádiu dorastania (95.48 $m^3$/ha ± 22.60 $m^3$/ha) sa takmer neodlišuje od štádia optima, percentuálne však tvorí moderové drevo o takmer 20% viac zo zásoby živých stromov ako v optime (37.16% ± 7.64%). Tento vyšší podiel zásoby moderového dreva je logický, keďže v dorastaní je v porovnaní s optimom nízka zásoba živého porastu.

Podobné výsledky dostaneme pri analýze priemerných absolútnych hodnôt pre určitú kombináciu vývojového štádia a výškovej kategórie, ktoré sú zobrazené na Obr. 1A. Štádiá dorastania a optima sa ani v jednotlivých výškových kategóriách významne nelíšia v zásobe moderového dreva, kým štádium rozpadu sa vo všetkých výškových kategóriách okrem poslednej vyznačuje výrazne vyšším množstvom moderového dreva. Obr. 1A tiež dobre dokumentuje pokles absolútneho množstva moderového dreva s nadmorskou výškou. Ak sa však zásoba moderového dreva relativizuje k zásobe živých stromov, tento trend sa stratí (Obr. 1B). Zaujímavosťou je, že štádium optima sa vyznačuje takmer zhodným podielom moderového dreva zo živej zásoby (cca 20%) v každej výškovej kategórii.

Obr. 1: Priebeh absolútnych (A) priemerných hodnôt zásoby moderového dreva a ich relativizovaných hodnôt ku zásobe živých stromov (B) v jednotlivých vývojových štádiách a výškových kategóriách.
Fig. 1: Performance of absolute average volumes of dead wood (A) and their ratios to the volume of living trees (B) in particular development stages and elevation categories.



[1]Volume of dead wood, [2]Elevation category in metres above sea level, [3]Development stage (from left to right: Stage of growth, maturity and breakdown), [4]Ratio of dead wood to the volume of living trees

Aj keď analýza priemerných hodnôt zistila určité trendy v zásobe moderového dreva v závislosti od vývojového štádia a nadmorskej výšky, pre štatistické potvrdenie ich vplyvu na sledovanú veličinu sme vykonali dvojfaktorovú analýzu variancie. Výsledky tejto analýzy potvrdili, že oba faktory sú štatisticky významné (Tab. 1), aj keď ich interaktívny vplyv sa nepotvrdil ako signifikantný. Ak však analyzujeme množstvo moderového dreva relativizované k živej zásobe porastu, zistíme, že nadmorská výška sa stáva nesignifikantným faktorom vplývajúcim na zásobu moderového dreva (Tab. 2). Vývojové štádium však ostáva štatisticky významným faktorom.

Tab. 1: Dvojfaktorová analýza variancie vplyvu výškovej kategórie a vývojového štádia na absolútne hodnoty zásoby moderového dreva.
Table 1: Bifactorial analysis of variance of the influence of elevation category and development stage on the absolute values of the dead wood volume.

| Drevina | Faktor | Suma štvorcov odchýlok | Počet stupňov voľnosti | Priemer štvorca odchýlok | F test | Hladina spoľahlivosti<br>*95%<br>**99%<br>***99.9% |
|---|---|---|---|---|---|---|
| 1 | 2 | 3 | 4 | 5 | 6 | 7 |
| Spolu[8] (Sm+Jr) | Výšková kategória[9] | 117453.95 | 3 | 39151.32 | 10.222 | 0.000 *** |
| | Štádium[10] | 136635.02 | 2 | 68317.51 | 11.891 | 0.000 *** |
| | Štádium*Výšková kategória | 13442.12 | 6 | 2240.35 | 1.170 | 0.339 |
| | Chyba – Reziduál[11] | 11490.21 | 45 | 255.34 | | |

Column No. 1 Tree species, 2 Factor, 3 Sum of squares of residuals, 4 Degrees of freedom, 5 Average sum of squares, 6 Fischer F test, 7 Significance level; [8]Together Norway spruce and rowan, [9]Elevation category, [10]Development stage, [11]Error - Residual

Tab. 2: Dvojfaktorová analýza variancie vplyvu výškovej kategórie a vývojového štádia na hodnoty zásoby moderového dreva relativizované ku zásobe živých stromov.
Table 1: Bifactorial analysis of variance of the influence of elevation category and development stage on the ratio of dead wood volume to the volume of living trees.

| Drevina | Faktor | Suma štvorcov odchýlok | Počet stupňov voľnosti | Priemer štvorca odchýlok | F test | Hladina spoľahlivosti<br>*95%<br>**99%<br>***99.9% |
|---|---|---|---|---|---|---|
| 1 | 2 | 3 | 4 | 5 | 6 | 7 |
| Spolu[8] (Sm+Jr) | Výšková kategória[9] | 6633.93 | 3 | 2211.31 | 2.538 | 0.068 |
| | Štádium[10] | 42163.05 | 2 | 21081.53 | 16.129 | 0.000 *** |
| | Štádium*Výšková kategória | 4216.30 | 6 | 702.72 | 1.613 | 0.166 |
| | Chyba – Reziduál[11] | 2614.03 | 45 | 58.09 | | |

Column No. 1 Tree species, 2 Factor, 3 Sum of squares of residuals, 4 Degrees of freedom, 5 Average sum of squares, 6 Fischer F test, 7 Significance level; [8]Together Norway spruce and rowan, [9]Elevation category, [10]Development stage, [11]Error - Residual

## Diskusia

Podobné absolútne množstvo moderového dreva ako v rámci našej inventarizácie zistili v sledovanej oblasti na slovenskej či poľskej strane Babej hory aj iní autori. Napr. Korpeľ (1989) nameral na svojich trvalých výskumných plochách v priemere 158 m$^3$/ha v roku 1980 a 161 m$^3$/ha o 7 rokov neskôr. Z údajov uvedených Jaworskim a Karczmarskim (1989, 1995) vyplýva, že priemerná zásoba moderového dreva bola na ich výskumných plochách na severnej, t.j. poľskej, strane Babej hory v roku 1984 82 m$^3$/ha, kým v roku 1994 stúpla na 165 m$^3$/ha. Podľa Jaworskeho a Palucha (2001) bola priemerná zásoba na nimi zisťovaných plochách 172 m$^3$/ha v roku 1986 a 126 m$^3$/ha v roku 1996. Holeksa (1998, 2001) zase uvádza v rámci svojho 14.4 ha výskumného objektu priemernú zásobu moderového dreva 131 m$^3$/ha.

Aj keď sa absolútne hodnoty jednotlivých autorov od seba veľmi neodlišujú, ich relativizácia k zásobe živých stromov odhaľuje zaujímavé skutočnosti. Kým v našom prípade tvorí moderové drevo priemerne 57% zo živej zásoby porastu, na základe Holeksových zistení (Holeksa 2001) dosahuje množstvo moderového dreva len tretinu, t.j. 32%, zo zásoby živých stromov. Naopak, na plochách Jaworskeho a Palucha (2001) sa zásoba moderového dreva blížila zásobe živých stromov, keď v roku 1996 činila 87% a v roku 1986 dokonca 121% zo živej zásoby porastu. Keďže hodnoty ostatných autorov pochádzajú z malého množstva výskumných objektov (3 až 5) o relatívne malej výmere (0.2 až 0.5 ha) s výnimkou Holeksovho objektu, zistené hodnoty sa ťažko môžu považovať za reprezentatívne pre celú oblasť Babej hory, pretože zásoba moderového dreva sa po ploche porastu veľmi mení (Müller a Schnell 2003).

Naše zistenia týkajúce sa množstva moderového dreva v jednotlivých vývojových štádiách sa odlišujú od poznatkov od Holeksu (1998), ktorý na svojom výskumnom objekte zistil, že zásoba moderového dreva rastie od štádia dorastania cez optimum po rozpad. Predpokladáme, že zistené rozdiely môžu byť spôsobené jednak malou reprezentatívnosťou jeho výskumného objektu pri zovšeobecňovaní výsledkov výskumu na celú skúmanú oblasť, ako aj vplyvom subjektu pri vylišovaní vývojových štádií.

Zaujímavým poznatkom je oveľa nižšia zistená zásoba moderového dreva v najvyššej výškovej kategórii v porovnaní s ďalšími troma skupinami. Vysvetlením môže byť registračná hranica 7cm, od ktorej sa moderové drevo zaznamenávalo. Porasty na hornej hranici lesa sa vyznačujú menšími dimenziami a preto môžeme predpokladať, že chyba v dôsledku merania presahuje v týchto častiach chybu, ktorej sme sa dopustili v nižších nadmorských výškach.

Významný vplyv nadmorskej výšky sa na základe analýzy variancie prejavil len na absolútne množstvá moderového dreva, ale nie na ich relativizované hodnoty k živej zásobe porastu. Tento fakt je možné vysvetliť závislosťou zásoby moderového dreva od zásoby živého porastu (Kühnel 1999), ktorá podobne ako moderové drevo s rastúcou nadmorskou výškou klesá (Merganič *et al.* 2003).

## Záver

V predkladanej práci hodnotíme zásobu moderového dreva v lesných porastoch národnej prírodnej rezervácie Babia hora. Výsledky poukazujú, že v skúmanej oblasti je možné so 68% pravdepodobnosťou očakávať priemernú zásobu moderového dreva v rozsahu 144.62 $m^3$/ha ± 19.81 $m^3$/ha, resp. 56.95% ± 8.95% zo zásoby živých stromov. Analýza vplyvu skúmaných faktorov potvrdila predpoklady, že vývojové štádium štatisticky signifikantne ovplyvňuje zásobu moderového dreva. Jeho vplyv je však významný v nižších polohách rezervácie, kým na hornej hranici lesa je štruktúra porastov natoľko rozpojená, že vývoj lesa prebieha v nepretržitom cykle a štádiá je už pomerne ťažko odlíšiť. Zaujímavým poznatkom je aj zistenie, že v štádiu optima predstavuje zásoba moderového dreva približne 20% zo zásoby živých stromov a to bez ohľadu na výškovú kategóriu. Ďalším skúmaným faktorom bola nadmorská výška, ktorej vplyv sa štatisticky preukázal len v rámci hodnotenia absolútneho množstva moderového dreva. Po relativizácii k zásobe živých stomov sa jej vplyv nepotvrdil, pretože množstvo moderového dreva koreluje so zásobou živých stromov a tá úzko koreluje s rastúcou nadmorskou výškou.

Uvedené poznatky sú významným príspevkom k obohateniu doterajších znalostí o zásobe moderového dreva, ktorú možno v človekom nenarušených lesných ekosystémoch očakávať. Hodnoty jeho množstva dokumentujú, že moderové drevo je trvalou súčasťou prirodzených lesných ekosystémov na hornej hranici lesa a preto je žiadúce venovať mu zvýšenú pozornosť.

**Poďakovanie**
Ďakujeme Ministerstvu životného prostredia Slovenskej republiky za finančnú podporu poskytnutú pre riešenie tejto práce.

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JOURNAL OF FOREST SCIENCE, *56*, 2010 (9): 397–405

# Coarse woody debris carbon stocks in natural spruce forests of Babia hora

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**ABSTRACT**: Although coarse woody debris (CWD) represents one of the major carbon pools in natural forest ecosystems, little information is available about its CWD carbon stocks. This study demonstrates the importance of proper estimation of carbon stocks in CWD, which accounts for the decay process of CWD, on an example of a natural mountainous spruce forest located in Central Europe. The study accounts for aboveground coarse woody debris including standing dead trees, lying deadwood, and naturally formed stumps. Basic mensurational information (diameter, height, decay class) about dead wood was collected in the field during the inventory of the forests of the nature reserve Babia hora. The data were used for the calculation of CWD timber volume. In the next step, CWD timber volume was converted to carbon stock using the carbon proportion of 50.1% and density values of decay classes derived from the information published elsewhere. The analysis revealed that when CWD timber volume was converted to carbon stocks using the basic wood density of fresh wood, C stocks were overestimated by 40% or more depending on the developmental stage and elevation. The results also revealed that as the elevation increases, CWD carbon stocks decrease and the differences between the developmental stages diminish.

**Keywords**: Babia hora nature reserve; deadwood; decay; elevation; natural forest; wood density

Recently, dead wood has become a widely discussed issue in forestry studies. The importance of its occurrence in forest stands has been emphasised in conjunction with the functioning and productivity of forest ecosystems (Humphrey et al. 2004); biodiversity (Ferris, Humphrey 1999; Humphrey et al. 2004; Saniga, Saniga 2004; Schuck et al. 2004); storage of nutrients and water (Harmon et al. 1986; Krankina et al. 1999); soil development and protection against soil erosion (Stevens 1997); rock fall and avalanches (Kupferschmidt et al. 2003); natural regeneration (Harmon, Franklin 1989; Mai 1999; Vorčák et al. 2005, 2006; Ulbrichová et al. 2006); climate change and accumulation of greenhouse gases in the atmosphere (Lombardi et al. 2008; Zell et al. 2009). In carbon sequestration studies, deadwood is recognised as an important component for conserving carbon stock. For example, in the USA 14% of the total forest carbon pool is stored in deadwood (Woodall et al. 2008).

Deadwood is usually divided into coarse and fine woody debris, although the minimum threshold diameter value varies a lot (0–35 cm, Cienciala et al. 2008). According to IPCC (2003), the border diameter is 10 cm. Harmon and Sexton (1996) found that below this diameter the decay rate increases exponentially, while above this diameter the decay rate decreases only slowly. From the two catego-

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Supported by the Slovak Research and Development Agency, Grant No. APVV-0632-07, and by the COST Action, Project No. COST-STSM-FP0603-04966, and by the Ministry of Agriculture of the Czech Republic, Project No. QH91077.

ries, coarse woody debris (CWD) is regarded as a more significant component due to its dimensions and substantial time during which it persists in the ecosystem. Hence, CWD acts as a long-term carbon sink until the decomposition process is completed, which can sometimes take up to 1,000 years (Feller 2003) depending on wood characteristics (tree species, dimensions), climate characteristics (temperature and moisture, Woodall, Liknes 2008) and the position on the ground (contact with the ground, Radtke et al. 2004).

In spite of the recognition of the importance of CWD for carbon sequestration, the studies dealing with carbon stock in deadwood in Europe are still scarce. Research of the forestry community usually deals with the volume of coarse woody debris (e.g. Saniga, Schütz 2002; Jankovský et al. 2004; Rahman et al. 2008; Sefidi, Mohadjer 2010; etc.). However, from the works realised elsewhere in the world it is known that during the decomposition process coarse woody debris looses not only its volume, but also mass and density (Krankina, Harmon 1995; Harmon et al. 2000; Coomes et al. 2002). Therefore, for the correct estimation of CWD carbon stock, additional parameters to those usually measured biometrical characteristics (diameter, length) are needed, namely the density of a particular decay class and carbon amount in CWD.

The goal of the presented paper is to examine the importance of taking into account the decomposition process in carbon stock estimation even though no nutrient analyses and measurements of wood density are available from the studied region. This is a usual case in forestry studies, since detailed analyses are both time-consuming (Zell et al. 2009) and cost demanding. On the basis of the published works on CWD decay, we hypothesised that using a single value of wood density for all decay classes can produce incorrect and misleading results. Therefore, for the estimation of CWD carbon stock in the presented paper we approximated wood densities of particular decay classes of CWD on the basis of published values from other regions. In the next step, we compared this approach with simple estimation of carbon stock using only one value of wood density for all decay classes.

## MATERIAL AND METHODS

Babia hora is an isolated mountain massif belonging to the outer Western Carpathian mountain range situated in the northern part of Slovakia at the border with Poland. The massif of Babia hora is built of tertiary flysch rocks, mainly sandstones, marl, claystones, slate and conglomerates. The soil types that occur in the area are raw soil, Andosol and most frequently Podzol. The mean annual precipitation is 1,600 mm, and the mean annual temperature 2°C. The forest stands are almost entirely composed of Norway spruce (*Picea abies* [L.] Karst.) with a small admixture of rowan (*Sorbus aucuparia* L.) and Silver fir (*Abies alba* Mill.).

In 1926, a nature reserve was established to preserve natural mountainous spruce forest ecosystems in this region. Originally the nature reserve encompassed 117.6 ha, but in 1974 the reserve was enlarged and currently its area is 503.94 ha (Korpeľ 1989). In the region of the nature reserve, 57 permanent circular sample plots were established in 2002 (Merganič et al. 2003), each with an area of 0.05 ha (i.e. radius = 12.62 m). The plots are located at an elevation ranging from 1,173 m to 1,503 m a.s.l., the latter representing the timber line in this region. The plots are equally divided between the three main developmental stages of virgin forests: stage of growth, maturity and breakdown as defined by Korpeľ (1989), i.e. each group

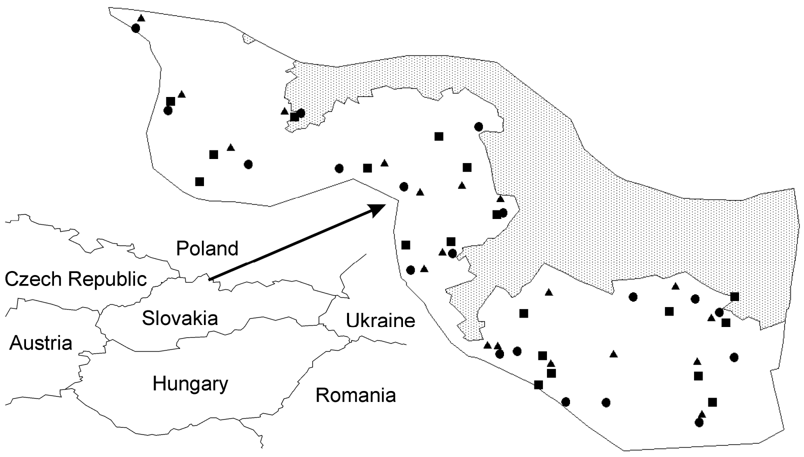

Fig. 1. Location of sample plots in the Nature Reserve Babia hora. Legend: ░ – alpine meadows and stands of mountain dwarf pine, Sample plots in the developmental stage of: ● – Growth, ■ – Maturity, ▲– Breakdown

consists of 19 plots (Fig. 1). The plots were further equally divided between four elevation categories (below 1,260 m; 1,261–1,360 m; 1,361–1,460 m; above 1,460 m a.s.l.) in order to detect an elevation gradient in data.

In each plot, dead standing trees or snags and lying dead wood (lying stems and stumps) above 7 cm in diameter were recorded. The category of stumps encompassed all naturally formed stumps and snags of the height smaller that 1.3 m, since the examined area is excluded from management practices. For dead standing trees and snags taller than 1.3 m in height, their tree height and diameter at breast height were assigned. In the case of lying dead wood, its total length and diameter at ½ of its length was measured, whereas for stumps only the diameter at 0.3 m height was determined.

The decay class was assessed using the 8-degree scale as proposed by Holeksa (2001). The decay classes are characterized on the basis of the presence or absence of bark, twigs and branches, log shape, texture, and position with respect to the ground. Decay class 1 represents the least decayed dead wood with intact bark, present twigs and branches, round shape, smooth surface, intact texture, and the position elevated on support points. As the decay process proceeds, the twigs, parts of branches and bark become traces to absent. For example, in decay class 4, only stubs of branches of diameter greater than 4–5 cm are present, a knife can slide up to 3 cm into a log, and crevices up to 0.5 cm deep are present. In the next decay classes, bark and branches are absent, wood becomes softer and fragmented, and the round shape becomes elliptical. Decay class 8 represents the most decomposed dead wood, when the log is on the ground overgrown by mosses and vascular plants. Due to a high frequency of crown and stem breakage, tree volume of dead standing trees was calculated using an integral equation, which was based on the models of stem shape derived by Petráš (1986, 1989, 1990). The simplified form for calculating the volume of stem inside bark is as follows:

$$v = \frac{\pi}{40{,}000} \times \int_{0}^{hR} d\left(h_i, hM, d_{1.3}, \vec{a}, sp\right)^2 dh \qquad (1)$$

Where:

$v$ – tree volume in m$^3$,

$hR$ – real (measured) tree height in m,

$hM$ – simulated tree height in m (estimated from the diameter-height curves derived from undamaged trees, Merganič *et al.* 2003),

$d_{1.3}$ – tree diameter at 1.3m height in cm,

$d$ – tree diameter at the $i^{th}$ tree height ($h_i$) in cm,

$a$ – vector of tree-species specific parameters in the model of stem shape,

$sp$ – tree species.

The volume of stumps was estimated as the volume of a cylinder of the height equal to 0.3 m. The volume of lying dead wood (logs) was calculated as the volume of a second degree paraboloid using Huber's formula:

$$v = h \times g_{1/2} \qquad (2)$$

Where:

$v$ – volume of the log in m$^3$,

$h$ – length of the log in m,

$g_{1/2}$ – cross-sectional area at ½ length of the log in m$^2$.

Total volume of coarse woody debris was given as a sum of the volumes of standing dead trees, stumps and lying logs.

Carbon storage in wood is obtained by converting the volume mass into the amount of carbon stored in this pool. For this conversion, carbon content in wood and wood density need to be known. Usually, carbon content in wood is estimated to be 50% (Coomes et al. 2002). Weiss et al. (2000) published more precise data for individual tree species of Central Europe. According to these authors, carbon content in Norway spruce wood is 50.1% of the dry mass and remains stable during the whole decomposition process of deadwood (Bütler et al. 2007). Basic wood density of Norway spruce living trees fluctuates between 0.41 g·cm$^{-3}$ (Bütler et al. 2007; Morelli et al. 2007) and 0.45 g·cm$^{-3}$ (Weiss et al. 2000). As wood decays, basic wood density decreases steadily (Harmon et al. 2000) depending on many factors as it is described e.g. in Radtke et al. (2004).

Since in our research object Babia hora no measurements of CWD wood density were performed, for the calculation of carbon amount in CWD we used the values published from other locations. Our literature review revealed that most of the studies dealing with the decay of CWD of Norway spruce (*Picea abies* [L.] Karst.) came from northern Europe (Krankina, Harmon 1995; Næsset 1999; Harmon et al. 2000; Yatskov 2001). From the two lately performed European studies, one comes from Italy (Morelli et al. 2007), while the other one comes from Switzerland (Bütler et al. 2007). For the purpose of our work we used the information about wood density of Norway spruce CWD in different decay stages provided by Næsset (1999), Harmon et al. (2000), Yatskov (2001), Bütler et al. (2007) and Morelli et al. (2007).

Table 1. Basic wood density of Norway spruce coarse woody debris per decay class calculated from the derived linear model Equation (3)

| | Decay class according to the scale of Holeksa (2001) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| | 0 (living trees) | 1 | 2 | 3 | 4 | 5 | 6 | 7 | 8 | Avg |
| Density (g·cm$^{-3}$) | 0.430 | 0.394 | 0.357 | 0.321 | 0.284 | 0.248 | 0.211 | 0.175 | 0.138 | 0.266 |

Since each of the mentioned studies uses another scale of wood deterioration with a different number of decay stages (3 to 8), the scales were first converted to the scale of Holeksa (2001) applied in Babia hora considering the verbal description of the decay degrees. Holeksa (2001) distinguishes 8 decay classes, while 0 stands for living trees, class 1 represents the least decomposed deadwood, and class 8 the most decomposed deadwood.

After the harmonisation of the different scales, the values of wood density were plotted against the harmonised degree of decay, and a regression was applied (Fig. 2). The analysis revealed that linear regression in the form

$$\text{density}_{\text{CWD}} = 0.430180 - 0.036464 \times \text{decClass}_{\text{CWD}} \qquad (3)$$

described the relationship best ($R^2 = 0.880$). The parameter density$_{\text{CWD}}$ stands for the basic wood density of coarse woody debris given in (g·cm$^{-3}$), and decClass$_{\text{CWD}}$ stands for the decay class (1 to 8) according to the scale of Holeksa (2001). The intercept equal to 0.430180 represents basic wood density of living trees, while the regression coefficient –0.036464 determines the reduction of basic wood density due to the deterioration. The statistical test of the regression coefficient revealed that it was highly significant from 0 ($t = -16.69$), which indicates a significant reduction of wood density in the course of decomposition process. The derived function (3) was used for the calculation of the final values of basic wood density for each decay class as given in Table 1.

The volume of coarse woody debris can then be converted to carbon stock using the following formula:

$$C_{\text{CWD}i} = V_{\text{CWD}i} \times \rho_{\text{CWD}i} \times C(\%) \times 10 \qquad (4)$$

Where:

$i$ – decay class [1 to 8 according to the applied scale of Holeksa (2001)],

$C_{\text{CWD}i}$ – carbon stock of CWD in the $i^{\text{th}}$ decay class in kg C·ha$^{-1}$,

$V_{\text{CWDi}}$ – wood volume of CWD in the $i^{\text{th}}$ decay class in m$^3$·ha$^{-1}$,

$\rho_{\text{CWD}i}$ – wood density of CWD in the $i^{\text{th}}$ decay class taken from Table 1 in g·cm$^{-3}$,

$C(\%)$ – carbon concentration in percent of the dry mass taken from Weiss et al. (2000) for Norway spruce (50.1%).

## RESULTS AND DISCUSSION

The results revealed that carbon storage in deadwood varies depending on the developmental stage of the forest, while the highest amount of carbon is stored in the stage of breakdown (Table 2). This stage is represented by more than 3 times higher carbon stock in deadwood than in the other two stages. The difference in carbon storage is higher than the difference in deadwood volume between the developmental stages, since the stage of breakdown is characterized by 2.6 and 2.7 higher volume of deadwood than the stage of growth and maturity, respectively (Merganičová et al. 2004). This difference results from the decomposition process, when the stage of breakdown is characterized by a significantly greater amount of the least decomposed deadwood (decay classes 1 and 2; Fig. 3a), which has higher wood density than the more decayed CWD (Fig. 2).

On the contrary, in the stage of growth the greatest timber volume of deadwood is accumulated in the last decay class 8 (Fig. 3a). Although this volume is significantly higher than the volume in all other

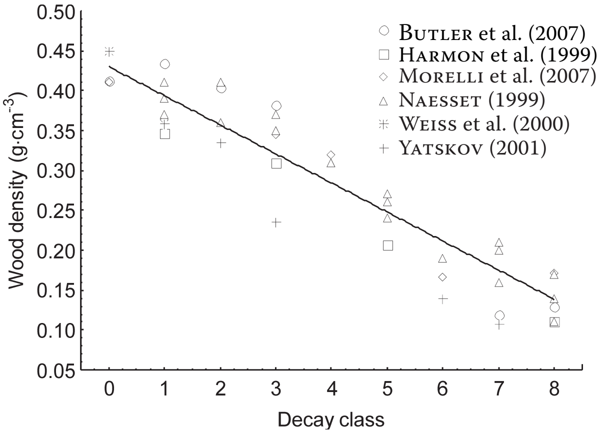

Fig. 2. Applied model for the estimation of the basic wood density of Norway spruce coarse woody debris (density$_{\text{CWD}}$ = 0.430180 – 0.036464 × decClass$_{\text{CWD}}$) using literature values for CWD decay classes (decClass$_{\text{CWD}}$) according to the scale of Holeksa (2001).

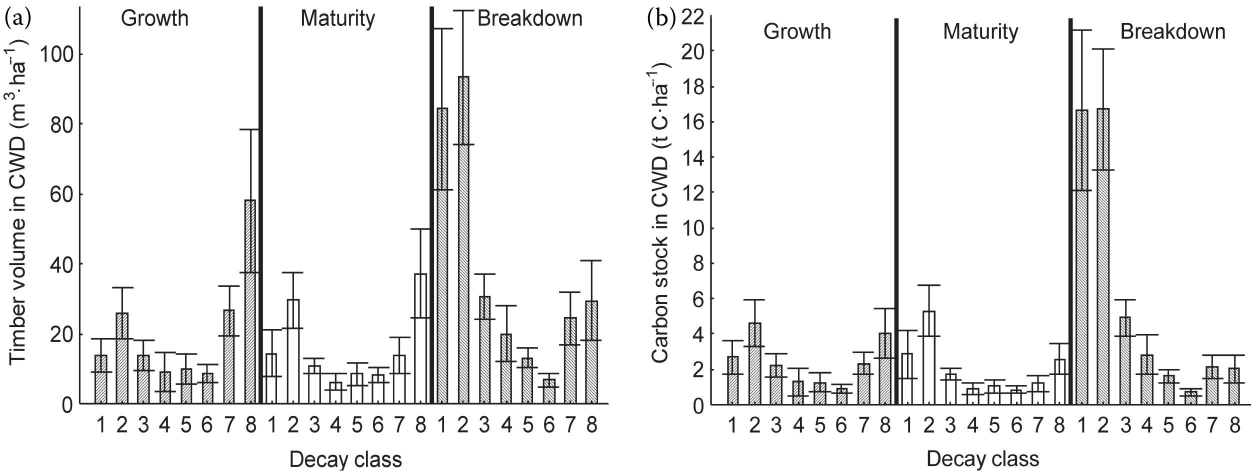

Fig. 3. Timber volume (a) and carbon storage (b) in coarse woody debris in particular developmental stages distributed along 8 decay classes defined by Holeksa (2001), where represents 95% confidence interval

classes in the stage of growth (Fig. 3a), the carbon stock in decay class 8 and the stage of growth is slightly lower than the carbon stock in decay class 2 in the same developmental stage (Fig. 3b) due to lower wood density (Table 1). The same pattern can be observed in the stage of maturity and decay classes 2 and 8 (Figs. 3a and 3b). In the stage of breakdown, large differences in the deadwood volume in early and late decay stages become even more profound in carbon stock.

If the elevation as a significant factor is accounted for in the analyses, both deadwood volume and carbon stock of CWD show a decline in all three developmental stages with increasing elevation (Fig. 4). This reduction follows the pattern of decreasing dimensions of trees with increasing elevation (Merganič et al. 2003). At upper elevations, climate characteristics are less favourable, which negatively affects forest productivity, and hence also the amount of CWD accumulated in the forest (Feller 2003). The highest deadwood volume as well as the highest carbon storage was found in the stage of breakdown and the first elevation category (Fig. 4). The other two stages, i.e. the stage of growth and maturity, are characterized by a very similar volume or carbon stock of coarse woody debris. As the elevation increases, the differences between the stage of breakdown and the other two stages diminish, and in the last elevation category become insignificant (Fig. 4).

The absolute values of carbon stock in CWD vary from 1.6 to 64.4 t C·ha$^{-1}$ depending on the developmental stage and the elevation category as it can be seen in Fig. 4. The values are higher than those reported by Krankina et al. (2002) for Russian boreal forests (0.1–0.7 t C·ha$^{-1}$) or by Woodall et al. (2008) for the USA (from 2.16 to 11.35 t C·ha$^{-1}$), since in our study we addressed natural forests excluded from forest management practice. However, our overall average value for the whole nature reserve (23.4 t C·ha$^{-1}$; Table 2) corresponds with the values from natural forests from other parts of the world, e.g. Chen et al.

Table 2. Average carbon stock in coarse woody debris in particular developmental stages. In the calculation we applied weights derived from the spatial proportion of the developmental stages in individual elevation categories, i.e. we used 12 weights as follows: 1st elevation category – stage of growth (G) 0.026, maturity (M) 0.051, breakdown (B) 0.026; 2nd elevation category – G 0.095, M 0.238, B 0.143; 3rd elevation category – G 0.058, M 0.25, B 0.077; 4th elevation category – G 0.012, M 0.012, B 0.012

| Developmental stage | Average (Ø) carbon stock in CWD (t C·ha$^{-1}$) | Confidence interval 95% | |
|---|---|---|---|
| | | Ø – 2 × SE | Ø + 2 × SE |
| Growth | $12.9^{S}$ | 5.0 | 20.9 |
| Maturity | $12.0^{S}$ | 4.1 | 20.0 |
| Breakdown | $44.5^{S}$ | 36.5 | 52.4 |
| Together | $23.4^{W}$ | 15.5 | 31.3 |

SE – Standard error; $^{S}$standardized for an average of a covariate variable elevation equal to 1,352.7 m a.s.l.; $^{W}$weighted average

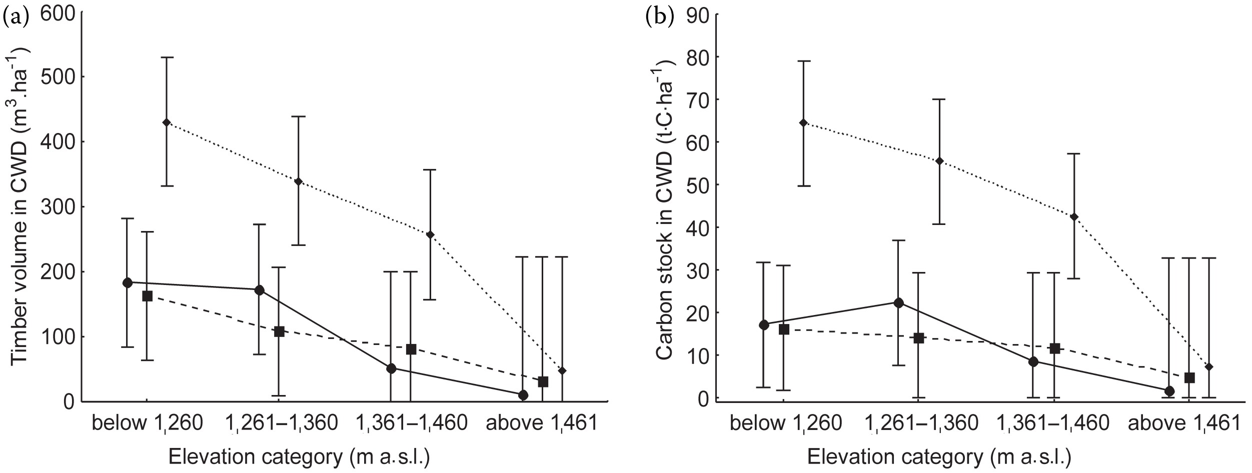

Fig. 4. Deadwood volume (a) and carbon storage (b) in developmental stages (● growth, ■ maturity, ◆ breakdown) and elevation categories, where represents 95% confidence interval

(2005) and Coomes et al. (2002) reported 17.3 ± 3.0 and 28.9 ± 8.5 t C·ha$^{-1}$ from old-growth riparian forests in Canada, and indigenous forests in New Zealand, respectively. Unfortunately, we have not found any information about CWD carbon stock in other virgin forests of Europe.

Expressed in relative values, in the area of interest the highest amount of carbon stored in CWD is present in standing dead trees and snags (61 ± 6.5%), followed by lying dead wood (38 ± 6.5%) and naturally formed stumps, in which on average only 1% (0–5%) of aboveground CWD carbon is stored. This distribution of carbon stock differs from the distribution of CWD volume among individual categories (50% dead standing trees, 48% lying deadwood, 2% stumps, Merganič et al. 2003) due to the effect of the decomposition process.

In order to examine whether it is important to account for the changes in wood density due to wood deterioration, we estimated carbon storage in deadwood in the Babia hora nature reserve in three different ways: (1) using the basic wood density of living trees (i.e. 0.430 g·cm$^{-3}$, see Table 1) for all decay classes, or (2) by applying the derived basic wood densities for each decay class (from Table 1), or (3) using the average basic wood density of coarse woody debris (i.e. 0.266 g·cm$^{-3}$) calculated from the derived linear regression (3).

The results show that if the deterioration is not accounted for and the basic wood density of living trees is used in the calculations, the estimated carbon stock in coarse woody debris can be as much as twice higher than if the effect of wood decomposition is included in the estimation of carbon storage (Table 2; Fig. 5a). Although the overestimation of carbon stock differs between the developmental stages and the elevation categories, it is significant in all cases (the ratio is always significantly different from 1, see Fig. 5a). In the stage of breakdown, the overestimation is the lowest although the absolute values of carbon stock are the greatest (Fig. 4), because this stage is characterized by a large input of deadwood in early stages of deterioration (Fig. 3). On average, carbon storage is overestimated by 35%, 65%, and 66% in the stage of breakdown, growth, and maturity, respectively.

If the average basic wood density of coarse woody debris is used for the conversion of deadwood volume to carbon stock, the results show that carbon stock is underestimated in the stage of breakdown (Fig. 5b). This is so because the highest proportion of CWD is in early decay classes 1 to 4 (Fig. 3) with greater basic wood density than the applied average density. The underestimation is significant in all but the first elevation category, where a large amount of CWD was also observed in decay classes 7 and 8 (Fig. 6a).

In the stage of growth and maturity, carbon stock is highly overestimated in the first elevation category (Fig. 5b). In the second elevation category, the estimation of CWD carbon stock using an average CWD density is equal to the estimation using individual values of CWD densities from Table 1. In upper elevation categories, CWD C stock was underestimated both in the stage of growth and in the stage of breakdown (Fig. 5b). This corresponds with the distribution of coarse woody debris in the decay classes, when with the increasing elevation the shift in the proportion of CWD in decay classes has been observed (Fig. 6). While in the first elevation category and the developmental stages of growth and breakdown the highest amount of deadwood is in the last decay class 8 (Fig. 6a), in the second elevation category the differences between the decay classes are much smaller with starting prevalence

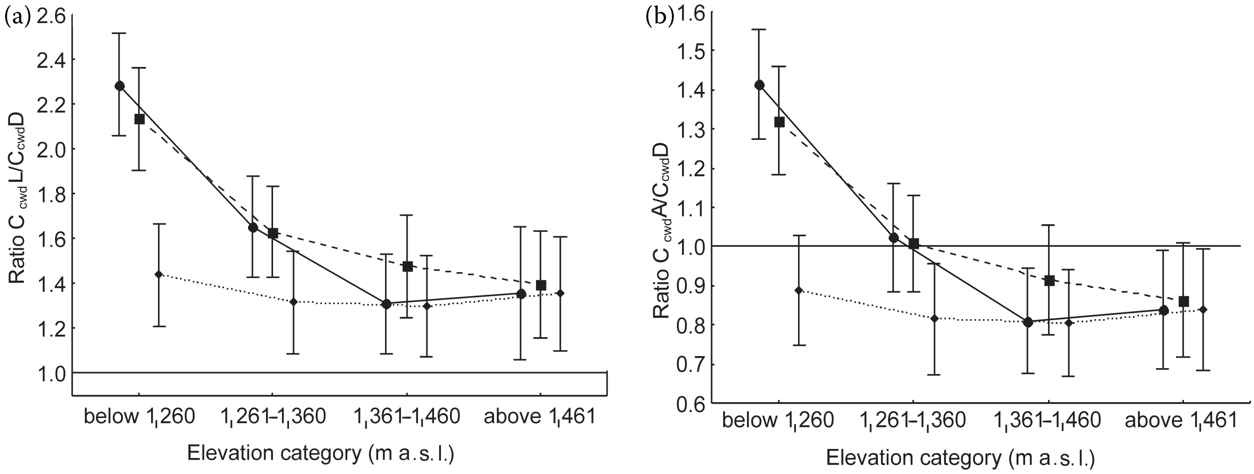

Fig. 5. Relative deviation of CWD carbon stock estimation when the decrease of deadwood density is not incorporated in the calculation (● growth, ■ maturity, ◆ breakdown, ⊥ represents 95% confidence interval). (a) represents the ratio between the carbon stock of coarse woody debris calculated with fresh wood basic density (i.e. 0.430 g·cm$^{-3}$, $C_{cwd}L$) and carbon stock of CWD using the decreasing wood densities from Table 1 ($C_{cwd}D$); (b) represents the ratio between the carbon stock of coarse woody debris calculated with average wood density of deadwood (i.e. 0.266 g·cm$^{-3}$, $C_{cwd}A$) and carbon stock of CWD using the decreasing wood densities from Table 1 ($C_{cwd}D$)

of early decomposed CWD (Fig. 6a). In the third elevation category, decay class 2 is the most abundant in both developmental stages of growth and breakdown (Fig. 6c), and the fourth elevation category is also characterized by higher CWD volume in early decay classes 1 to 4 (Fig. 6d).

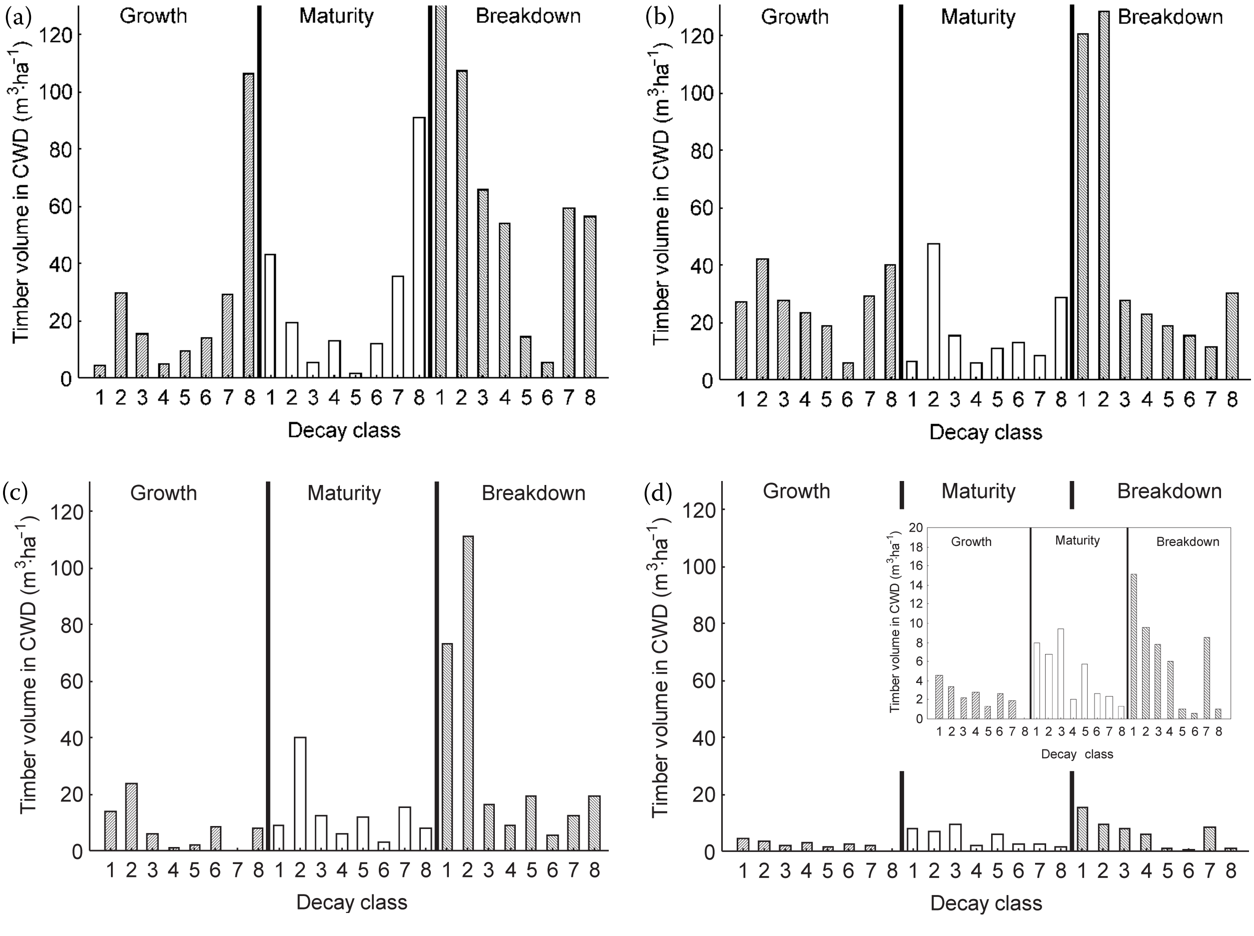

Fig. 6. Distribution of coarse woody debris volume between 8 decay classes separately in three developmental stages and four elevation categories (a) below 1,260 m; (b) 1,261–1,360 m; (c) 1,361–1,460 m, (d) above 1,460 m a.s.l.

## CONCLUSION

In the presented study we estimated the carbon stock in coarse woody debris in spruce virgin forests of the nature reserve Babia Hora in Slovakia, which has been found to be highly dependent on the developmental stage and the elevation. CWD carbon stocks are the greatest in the stage of breakdown characterized by the largest amount of the least decayed deadwood. As the elevation increases, CWD carbon stocks decrease due to lower forest productivity expressed in lower tree dimensions at the upper timberline, and the differences between the developmental stages diminish.

The current lack of exact information and knowledge of the decay process of coarse woody debris in Central Europe can hinder precise carbon inventories. We demonstrated that the carbon stock could be highly overestimated if the decay process of the deadwood is not accounted for. There is an urgent need for further research in the field of coarse woody debris decomposition in order to better understand the nutrient cycle of forest ecosystems, and to be able to provide reliable data on greenhouse gas emissions which are countries obliged to report under the United Nations Framework Convention on Climate Change.

## Acknowledgement

We thank three anonymous reviewers for their helpful comments.

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Recieved for publication January 27, 2010
Accepted after corrections April 26, 2010

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