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# Caddisfly assemblages of high mountain streams (The High Tatra Mts, Slovakia) influenced by a major windstorm event

Daniela Kalaninová[1], Eva Bulánková[1] & Ferdinand Šporka[2]

[1]*Department of Ecology, Faculty of Natural Sciences, Comenius University in Bratislava, Mlynská dolina, SK-84215 Bratislava, Slovakia; e-mail: kalaninova@fns.uniba.sk*
[2]*Institute of Zoology, Slovak Academy of Sciences, Dúbravská cesta 9, SK-84506 Bratislava, Slovakia*

**Abstract:** Hydrobiological research of high mountain streams in the High Tatras was carried out in 2009–2010. We evaluated the influence of windstorm on caddisfly assemblages. To assess the influence of windstorm we focused on river morphology using the RHS method and evaluation of TAM, TOM and BOM (CPOM, FPOM, UFPOM) amount in seven streams. Site 1 was a control and the other six were disturbed by the windstorm in different ways. The most remarkable differences compared to the control site was in feeding structure at sites most affected by erosion. In these streams there was a noticeable dominance of predators from the family Rhyacophilidae (mainly *Rhyacophila tristis*; sites 3, 4, 5, 6) and a remarkable decrease of passive filter feeders (site 4, 5). Using Spearman coefficient we confirmed a positive correlation between the proportion of predators and amount of TAM caused by erosion of steep deforested slopes of windstorm affected sites. In contrast, we did not observe such an increase in the proportion of predators at site 7, which was affected by both windstorm and subsequently by fire. This might be explained by the shallow vee valley with no observed erosion, where this site is situated. We also found a negative correlation between predators and evenness, indicating unstable community structure clearing succesion in the streams disturbed by erosion, and a negative correlation between passive filter feeders and UFPOM. We found out that overhanging tree boughs and LWD had an influence on species composition. RHS was a useful tool in characterising the influence of hydromorphology on caddisfly assemblages.

**Key words:** caddisflies; Trichoptera; High Tatras; windstorm; hydromorphology; erosion; feeding types

## Introduction

The windstorm in the High Tatras in 2004 was one of the most devastating in the history of this high-mountain ecosystem. Its intensity was likely conditioned by global climate change (Koreň 2005), which can influence the Earth's environment in a significant way. Despite an expected increase in frequency and intensity of windstorm activity (Hopkinson et al. 2008), little is known about the influence of such extreme events on stream biota.

In this article we focus on caddisflies, because they are: (1) well-suited to reflect the intensity of different stressors on aquatic ecosystems (Hering et al. 2009), particularly hydromorphological degradation (e.g., Statzner et al. 2001; Lorenz et al. 2004) and (2) recommended as flagship indicators for the assessment of climate change in mountain areas (Čiamporová-Zaťovičová et al. 2010).

Historical information on Trichoptera in the Tatra Mountains comes from the faunistical research from the turn of the 19$^{th}$ and 20$^{th}$ century (Mocsáry 1899 and others), which was later summarized in a detailed study of Mayer (1939). Further, fragmented data appeared in the second half of 20$^{th}$ century after the establishment of the Tatra National Park (TANAP), mainly focused on hydrobiological research (e.g., Krno et al. 1985, 1986; Bulánková et al. 2001; Šporka et al. 2002; Krno 2006). Results of Polish trichopterological research in the Tatra Mountains are reviewed by Szczesny (1986). The most detailed information about the occurrence and vertical distribution of Trichoptera species in the TANAP area comes from Chvojka (1992).

The first results of the effect of deforestation and altered temperature regime of streams caused by windstorm in given area were published by Lánczos et al. (2011) and Šustek (2011) who dealt with stoneflies and ripicol Carabidae, respectively.

The main objective of our study was to characterize the caddisfly assemblage of six streams impacted by windstorm and one near-natural stream in the High Tatra Mountains. Specific objectives were to: (1) assess the structure of caddisfly assemblages, (2) compare the assemblage structure in near natural stream with streams affected by windstorm and (3) evaluate the influence of altered river morphology due to windstorm on caddisflies.

## Material and methods

*Study area*

The study area was located in the High Tatras, Slovakia. The most important physiographical and chemical parameters of the sites are given in Table 1 and 2. The map, pictures and hydromorphological characteristic of study sites

Table 1. Physiographical characteristics of selected sites in the High Tatra Mts.

| Site number | Stream name | Longitude WGS84 E | Latitude WGS84 N | Altitude (m) | Stream order (Strahler, 1964) | Forest area (km$^2$) | Proportion of forest area damaged by windstorm (%) | Length of stream damaged by windstorm (m) | Slope (%) |
|---|---|---|---|---|---|---|---|---|---|
| 1 | Veľký Šum | 20°6′26.83″ | 49°7′44.95″ | 1253 | 3 | 0.940 | 0.00 | 0 | 31.7 |
| 2 | Poprad | 20°4′41.93″ | 49°7′12.49″ | 1232 | 4 | 4.286 | 6.39 | 458 | 10.2 |
| 3 | Biely Váh | 20°0′58.28″ | 49°7′18.95″ | 1232 | 3 | 1.157 | 37.18 | 1448 | 14.3 |
| 4 | Batizovský potok | 20°8′39.44″ | 49°7′11.84″ | 1044 | 3 | 1.620 | 45.48 | 1728 | 19.0 |
| 5 | Velický potok | 20°10′30.49″ | 49°7′22.13″ | 1001 | 3 | 1.617 | 28.06 | 1131 | 15.8 |
| 6 | Hromadná Voda | 20°10′18.95″ | 49°7′20.88″ | 1021 | 3 | 2.264 | 40.46 | 1706 | 18.4 |
| 7 | Slavkovský potok | 20°11′42.48″ | 49°7′58.77″ | 1047 | 3 | 1.109 | 24.11 | 762 | 23.1 |

Table 2. Values of transported, benthic in/organic mater and conductivity at sampling sites (site 1 – control site).

| Site | Date | CPOM AFDM (g m$^{-2}$) | FPOM AFDM (g m$^{-2}$) | UFPOM AFDM (g m$^{-2}$) | TAM (mg L$^{-1}$) | TOM AFDM (mg L$^{-1}$) | Conductivity (µS cm$^{-1}$) |
|---|---|---|---|---|---|---|---|
| 1 | 13.5.2009 | 1.223 | 14.316 | 0.94 | 0.0 | 4.9 | 29.5 |
| | 1.10.2009 | 7.468 | 17.820 | 2.66 | 0.0 | 3.4 | |
| 2 | 18.5.2010 | 2.251 | 31.386 | 3.42 | 0.0 | 3.5 | 26.5 |
| | 22.9.2010 | 1.306 | 20.747 | 4.45 | 0.0 | 3.7 | |
| 3 | 20.5.2010 | 2.039 | 3.214 | 0.99 | 3.4 | 1.5 | 21.0 |
| | 24.9.2010 | 6.892 | 14.685 | 2.15 | 0.0 | 3.9 | |
| 4 | 18.5.2010 | 1.143 | 10.779 | 3.13 | 0.0 | 2.6 | 16.0 |
| | 24.9.2010 | 7.088 | 51.749 | 4.70 | 2.6 | 3.7 | |
| 5 | 19.5.2010 | 0.348 | 4.096 | 37.08 | 84.5 | 4.5 | 19.0 |
| | 21.9.2010 | 3.638 | 10.798 | 1.53 | 0.0 | 2.2 | |
| 6 | 13.5.2009 | 66.899 | 253.229 | 1.77 | 0.3 | 5.6 | 27.0 |
| | 1.10.2009 | 6.449 | 36.844 | 7.43 | 13.7 | 11.2 | |
| 7 | 13.5.2009 | 4.329 | 40.600 | 39.20 | 0.0 | 4.0 | 32.0 |
| | 1.10.2009 | 2.344 | 3.384 | 5.40 | 0.0 | 3.2 | |

Explanations: CPOM – coarse particulate organic matter, FPOM – fine particulate organic matter, UFPOM – ultra fine particulate organic matter, TAM – transported inorganic matter, TOM – transported organic mater.

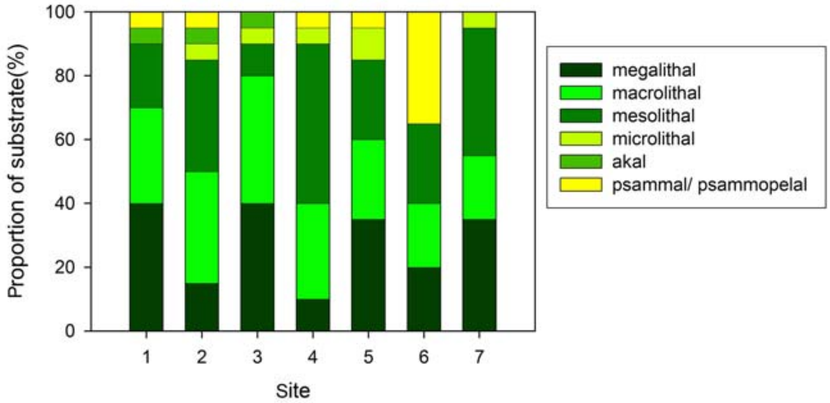


Fig. 1. Proportion of inorganic substrate types occuring in the study sites at 9. spot-checks, method described in AQEM Consortium (2002).

are given in Bulánková et al. (2013), where comparison among these sites is explained.

*Data sampling*
Macroinvertebrate samples were collected between late March 2009 and October 2010 using the standard AQEM method (AQEM Consortium, 2002). Spring and autumn sampling provided quantitative samples (exact dates of quantitative sampling are in Table 2). Macroinvertebrate samples were collected within and located at spot-check 9 of our RHS survey (a 50 m section of the stream). All present dominant microhabitats (with coverage at least 5%) (Fig. 1),

Table 3. Dominance values (%) of caddisfly taxa collected at seven sampling sites (spring and autumn samples only).

| Taxon/site | **1** | **2** | **3** | **4** | **5** | **6** | **7** |
|---|---|---|---|---|---|---|---|
| **Apataniidae** | **15.7** | **0.0** | **0.0** | **0.0** | **0.0** | **2.4** | **33.1** |
| *Apatania carpathica* Schmid, 1954 | 13.0 | 0.0 | 0.0 | 0.0 | 0.0 | 2.4 | 22.8 |
| *Apatania fimbriata* (Pictet, 1834) | 2.8 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 10.3 |
| **Glossosomatidae** | **0.0** | **3.8** | **0.4** | **1.2** | **2.0** | **2.1** | **0.0** |
| *Glossosoma conformis* Neboiss, 1963 | 0.0 | 3.8 | 0.4 | 1.2 | 2.0 | 2.1 | 0.0 |
| **Goeridae** | **1.9** | **0.0** | **0.8** | **0.2** | **0.0** | **1.7** | **6.2** |
| *Lithax niger* (Hagen, 1859) | 1.9 | 0.0 | 0.8 | 0.2 | 0.0 | 1.7 | 6.2 |
| **Limnephilidae** | **68.5** | **71.1** | **48.4** | **38.0** | **35.6** | **38.7** | **49.7** |
| *Acrophylax sowai* Szczesny, 2007 | 0.0 | 0.4 | 0.8 | 0.0 | 0.0 | 0.0 | 0.0 |
| *Allogamus auricollis* (Pictet, 1834) | 0.9 | 2.1 | 15.2 | 0.0 | 1.2 | 3.1 | 6.9 |
| *Allogamus uncatus* (Brauer, 1857) | 0.0 | 0.4 | 0.8 | 0.0 | 0.0 | 3.4 | 4.1 |
| *Chaetopteryx fusca* Brauer, 1857 | 0.0 | 0.4 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| *Drusus annulatus* (Stephens, 1837) | 1.9 | 25.1 | 13.6 | 0.0 | 0.0 | 5.1 | 4.8 |
| *Drusus biguttatus* (Pictet, 1834) | 3.7 | 3.4 | 1.6 | 1.0 | 1.6 | 0.0 | 1.4 |
| *Drusus discolor* (Rambur, 1842) | 19.4 | 5.5 | 6.8 | 2.9 | 4.0 | 4.5 | 0.7 |
| *Drusus monticola* McLachlan, 1876 | 0.0 | 3.4 | 2.8 | 0.0 | 0.0 | 0.0 | 0.0 |
| *Drusus* spp. juv. | 0.9 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| *Ecclisopteryx dalecarlica* Kolenati, 1848 | 0.0 | 0.0 | 0.0 | 0.0 | 1.2 | 0.0 | 0.0 |
| *Ecclisopteryx madida* (McLachlan, 1867) | 0.0 | 0.9 | 0.0 | 1.4 | 14.6 | 14.4 | 0.0 |
| *Halesus rubricollis* (Pictet, 1834) | 1.9 | 3.0 | 0.0 | 0.0 | 0.0 | 0.7 | 0.0 |
| Limnephilidae (juv.) | 39.8 | 25.5 | 6.8 | 32.7 | 13.0 | 5.5 | 31.0 |
| *Potamophylax cingulatus depilis* Szczesny, 1994 | 0.0 | 0.9 | 0.0 | 0.0 | 0.0 | 1.7 | 0.7 |
| *Potamophylax* sp. juv. | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.3 | 0.0 |
| **Philopotamidae** | **1.9** | **3.0** | **10.4** | **2.7** | **0.8** | **7.9** | **0.0** |
| *Philopotamus ludificatus* McLachlan, 1878 | 0.0 | 0.0 | 7.2 | 2.2 | 0.0 | 7.5 | 0.0 |
| *Philopotamus montanus* (Donovan, 1813) | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.3 | 0.0 |
| *Philopotamus* sp. juv. | 1.9 | 3.0 | 3.2 | 0.4 | 0.8 | 0.0 | 0.0 |
| **Rhyacophilidae** | **12.0** | **22.1** | **40.0** | **57.6** | **59.1** | **44.2** | **11.0** |
| *Rhyacophila fasciata* Hagen, 1859 | 0.0 | 0.0 | 0.8 | 0.6 | 0.0 | 0.0 | 0.0 |
| *Rhyacophila glareosa* McLachlan, 1867 | 9.3 | 0.0 | 2.4 | 1.2 | 0.0 | 0.0 | 0.0 |
| *Rhyacophila philopotamoides* McLachlan, 1879 | 0.0 | 0.0 | 0.0 | 0.0 | 0.4 | 0.0 | 0.0 |
| *Rhyacophila polonica* McLachlan, 1879 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 2.1 |
| *Rhyacophila* sp. juv. | 1.9 | 0.0 | 0.4 | 0.0 | 0.4 | 1.7 | 0.0 |
| *Rhyacophila tristis* Pictet, 1834 | 0.9 | 16.6 | 24.4 | 51.4 | 52.2 | 40.4 | 5.5 |
| *Rhyacophila vulgaris* Pictet, 1834 | 0.0 | 5.5 | 12.0 | 4.3 | 6.1 | 2.1 | 3.4 |
| **Sericostomatidae** | **0.0** | **0.0** | **0.0** | **0.4** | **2.4** | **3.1** | **0.0** |
| *Sericostoma personatum/flavicorne* | 0.0 | 0.0 | 0.0 | 0.4 | 2.4 | 0.0 | 0.0 |
| Sericostomatidae (juv.) | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 | 3.1 | 0.0 |

were assessed within 20 units, each representing an area of 25 × 25 cm, sampled by a 500 μm hand-net "jabbing". Qualitative samples of larvae and adults of aquatic insects were taken from all sites. Samples were preserved in 4% formaldehyde. Organisms were sorted under a stereo microscope (10 magnification), then stored in 70% ethanol. They were identified to the lowest taxonomic level, species level when possible according to Waringer & Graf (1997) and Lechthaler & Stockinger (2005). Sample collection procedure of TOM, TAM – BOM and evaluation of LWD is given in Bulánková et al. (2013).

*Data analysis*

Only the quantitative samples of caddisfly assemblages and data for TAM, TOM and BOM from the spring and autumn were calculated and further statistically evaluated. Similarity of the sites on the basis of caddisflies was evaluated using Past software, version 0.45 (Hammer & Harper 2001) by method of paired group, correlation. Feeding type proportion, number of taxa, diversity (Shannon – Wiener index and Margalef index) and evenness at the sites were calculated using Asterics software, version 3.3 (Vogl 2011). Abundance (ind.$m^{-2}$) was also calculated at each site. Correlation between selected proportions of feeding types, evenness and selected environmental factors was made using Spearman correlation test in Statistica software, version 7 (2004), $P = 0.05$. The influence of hydromorphological factors on caddisfly assemblages was tested using Canonical Correspondence Analysis (CCA) (Monte Carlo permutation test, 499 permutations, $P = 0.05$) in CANOCO software, version 4.56 (ter Braak & Šmilauer 1997).

## Results

At the seven study sites we found 2,310 caddisfly larvae belonging to 31 taxa. The 1,353 larvae from the spring and autumn quantitative samples belonged to 25 species (Table 3). Limnephilidae individuals prevailed at both site 2 (71.1%) and the control site (68.5%). In the most affected sites the family Rhyacophilidae (especially species *Rhyacophila tristis*) dominated (site 5 – 59.1%, site 4 – 57.6%, site 6 – 44.2%).

On the basis of species composition we can divide the study streams as follows (Fig. 2):

1. Group A: includes the control site and, surprisingly site 7, which was affected not only by windstorm but also by a subsequent fire. These separate from the streams clustered in group B by occurrence of species from the family Apataniidae (mainly *Apatania carpathica*, less *A. fimbriata*), the absence of the family Glos-

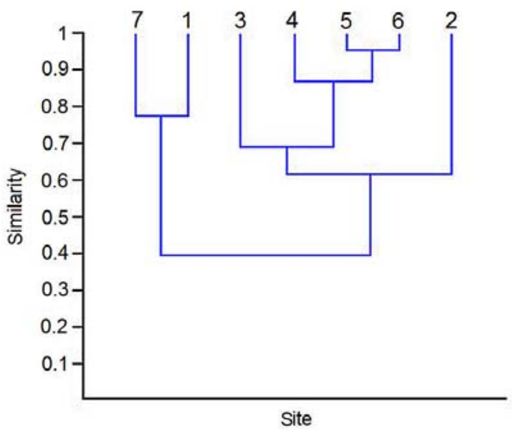


Fig. 2. Tree diagram of similarity of seven sites on the basis of taxonomic composition of caddisflies. Paired group, correlation, spring and summer sampling.

sosomatidae and fewer species of the family Rhyacophilidae (especially species *Rhyacophila tristis*).

2. Group B: streams variously affected by windstorm; they can be divided into subgroups:

a) site 2: the least affected site with greatest values of width, depth and discharge (4th order stream) is the most outstanding from other streams because of fewer Rhyacophilidae species, typical species is *Drusus annulatus*.

b) sites 3–6: these were characterised with higher proportion of predators of the family Rhyacophilidae; however sites 4–6 are even more closely linked together with the pronounced dominance of *Rhyacophila tristis*, the occurrence of species *Ecclisopteryx madida* (at site 5 also *E. dalecarlica*, which is even from 10 % epipotamal species (Schmedtje & Colling 1996)) and occurrence of the family Sericostomatidae.

Values of selected metrics of abundance and species richness are given in Table 4. Site 1 had average evenness, diversity and the lowest abundance. One of the most impacted sites – site 4 had the highest abundance (392 ind. $m^{-2}$) and together with site 5 had the lowest diversity (both Shannon-Wiener and Margalef index) as well as evenness. Together with site 7 they reached also the lowest number of taxa (13). The highest number of taxa (18) was recorded at impacted site 6. The highest diversity was recorded at site 6 (Margalef index = 2.995) as well as at site 3 (Shannon-Wiener index = 2.276), where the presence of not only mountain but also submountain species (e.g., *Allogamus auricollis* at site 3 or *Potamophylax* spp. at site 6) was typical.

The proportion of feeding types is shown in Fig. 3. The most erosion impacted sites had the highest proportion of predators (represented mainly by species *Rhyacophila tristis*), especially site 4 (55.3%) and 5 (53.8%) and the lowest proportion of passive filter feeders (3.8% and 2.8%, respectively). The highest proportion of grazers/scrapers (represented mainly by species *Drusus annulatus*) occurred in the largest stream Poprad – site 2 (37.7%), followed by site 7 (28.6%). The control site had, together with sites 4 and 5, the lowest proportion of grazers/scrapers (19.5%, 15.1% and 24.1%, respectively). Site 1 had the highest proportion of shredders (22.1%). The higher proportion of "not available" data (= data of species with unknown feeding ecology) at site 1 and 7 is represented mainly by

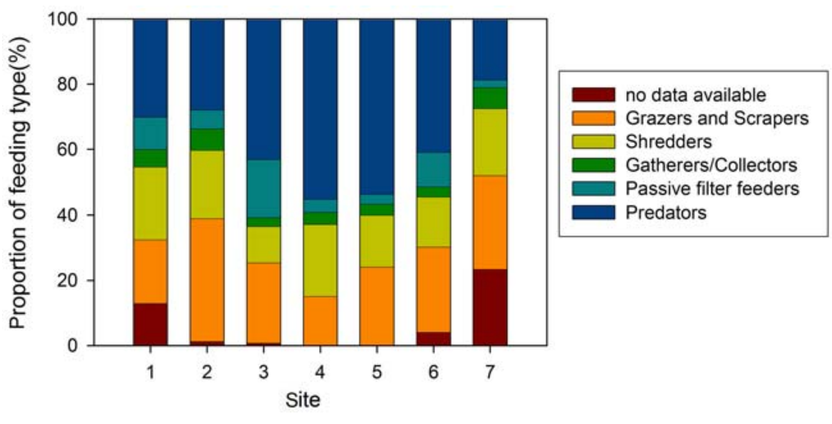


Fig. 3. The proportion of feeding types of caddisflies at all the sites according to ASTERICS, 3.3.

Table 4. Metrics of abundance and caddisfly species richness.

| Metric/site | 1 | 2 | 3 | 4 | 5 | 6 | 7 |
|---|---|---|---|---|---|---|---|
| Abundance (ind.$m^{-2}$) | 86.4 | 188 | 200 | 392 | 197.6 | 233.6 | 116 |
| Number of taxa | 14 | 16 | 17 | 13 | 13 | 18 | 13 |
| Diversity (Shannon-Wiener-Index) | 1.891 | 2.111 | 2.276 | 1.333 | 1.611 | 2.151 | 2.054 |
| Diversity (Margalef Index) | 2.777 | 2.747 | 2.898 | 1.937 | 2.178 | 2.995 | 2.411 |
| Evenness | 0.717 | 0.761 | 0.803 | 0.520 | 0.628 | 0.744 | 0.801 |

Table 5. Spearman correlation between selected proportion of feeding types and environmental parameters ($P < 0.05$).

| | Pred/known sp. | Pred/all | Pff/sp. | Evenness | TAM average | UFPOM average |
|---|---|---|---|---|---|---|
| Pred/known sp. | | 0.750 | 0.107 | **–0.786** | 0.445 | –0.143 |
| Pred/all | | | –0.107 | –0.500 | **0.815** | 0.179 |
| Pff/sp. | | | | 0.286 | –0.074 | **–0.786** |
| Evenness | | | | | –0.259 | –0.286 |
| TAM average | | | | | | 0.445 |
| UFPOM average | | | | | | |

Explanations: Pred – predators, Pff – passive filter feeders, all – all the larvae, sp. – the larvae determinable to species level, known sp. – the larvae determinable to species level with known feeding ecology, TAM – transported inorganic matter, UFPOM – ultra fine particulate organic matter, statistically significant values are in bold.

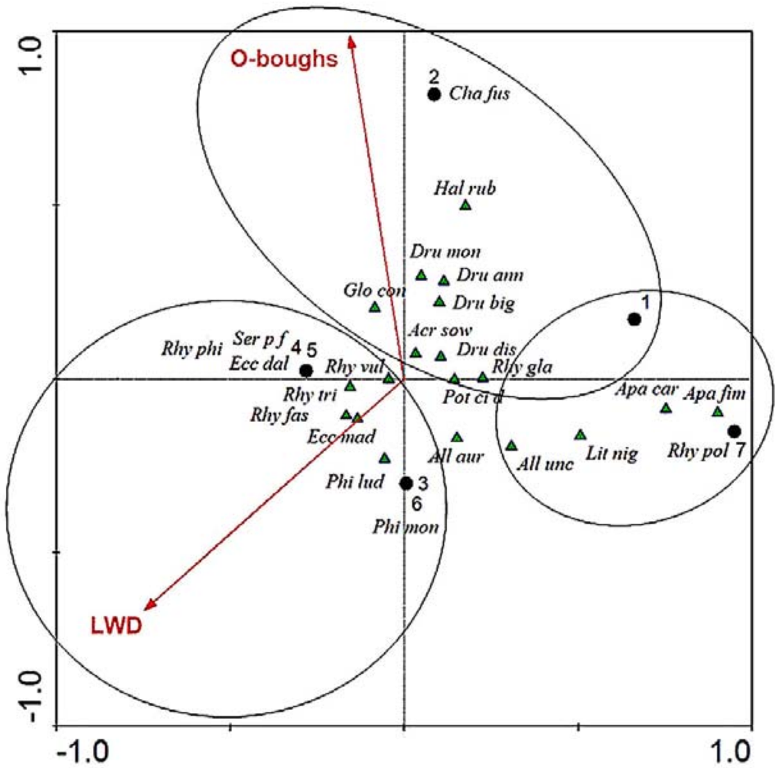


Fig. 4. Relation of caddisfly species to the selected hydromorphological features (CCA) (black circle: sample, green triangle: species, LWD: large woody debris, O-boughs: overhanging boughs, *Acr sow: Acrophylax sowai, All aur: Allogamus auricollis, All unc: Allogamus uncatus, Apa car: Apatania carpathica, Apa fim: Apatania fimbriata, Cha fus: Chaetopteryx fusca, Dru ann: Drusus annulatus, Dru big: Drusus biguttatus, Dru dis: Drusus discolor, Dru mon: Drusus montanus, Ecc dal: Ecclisopteryx dalecarlica, Ecc mad: Ecclisopteryx madida, Glo con: Glossosoma conformis, Hal rub: Halesus rubricollis, Lit nig: Lithax niger, Phi lud: Philopotamus ludificatus, Phi mon: Philopotamus montanus, Pot c d: Potamophylax cingulatus depilis, Rhy fas: Rhyacophila fasciata, Rhy gla: Rhyacophila glareosa, Rhy phi: Rhyacophila philopotamoides, Rhy pol: Rhyacophila polonica, Rhy tri: Rhyacophila tristis, Rhy vul: Rhyacophila vulgaris, Ser p f: Sericostoma personatum/flavicorne*)

the species *Apatania carpathica*, occuring extensively just at these two sites.

We recorded a positive correlation between the proportion of predator species to all the larvae and TAM in streams (0.815), a negative correlation between the proportion of predators to all the species which have known feeding ecology and evenness (–0.786) and a negative correlation between the proportion of passive filter feeders to all the species and UFPOM (–0.786) (Table 5).

Within the group of five tested hydromorphological parameters by CCA (Fig. 4) (shading of channel, overhanging boughs, exposed bankside roots, fallen trees, LWD), overhanging boughs were statistically significant ($F = 2.166$; $P = 0.046$). Importance of LWD was represented by high value of $P = 0.560$ ($F = 1.982$). Eigenvalues of axes 1 and 2 of the CCA are 0.341 and 0.362, respectively, and explained 58.5% of the overall variance of the species data (Table 6). According to these two factors we can divide the sites and species composition into three groups. In the first group we recorded a positive correlation between LWD and recorded species of genera *Ecclisopteryx, Philopotamus, Sericostoma* and most species of the genus *Rhyacophila* at sites 3, 4, 5 and 6. The second group showed a positive correlation with overhanging boughs especially at site 2 and

Table 6. Summary table of CCA based on quantitative insect data – eigenvalues and percent of variance of the first four ordination axes.

| Axes | 1 | 2 | 3 | 4 | Total inertia |
|---|---|---|---|---|---|
| Eigenvalues | 0.341 | 0.362 | 0.204 | 0.153 | 1.203 |
| Species-environment correlations | 0.888 | 0.000 | 0.000 | 0.000 | |
| Cumulative percentage variance: | | | | | |
| of species data | 28.4 | 58.5 | 75.5 | 88.2 | |
| of species-environment relation | 100.0 | 0.0 | 0.0 | 0.0 | |
| Sum of all eigenvalues | | | | | 1.203 |
| Sum of all canonical eigenvalues | | | | | 0.341 |

consisted mainly of species *Chaetopteryx fusca*, *Halesus rubricollis*, *Glossosoma conformis* and species of genus *Drusus*. The third group consisted of the species typical mainly for the site 7 (*Rhyacophila polonica*), but also the control site 1 (*Apatania fimbriata* and *A. carpathica*).

## Discussion

*Species composition*

Trichoptera assemblages in seven assessed streams of the High Tatras were represented mainly by the families Rhyacophilidae (especially *Rhyacophila tristis*) and Limnephilidae (mainly *Drusus* spp.). However, the family Limnephilidae was prevalent at the least disturbed and more shaded sites (1 and 2), whereas the family Rhyacophilidae dominated the most disturbed deforested sites (4, 5 and 6). Similarly Arscott et al. (2003) observed substantially higher abundance of Limnephilidae species in the forested floodplain and a greater abundance of *Rhyacophila* spp. in the open-gravel floodplain of headwater streams.

Our results resemble the species composition in the streams of the montane zone described by the most detailed work related to caddisflies of the TANAP (Chvojka 1992). Nevertheless, we did not record species *Chaetopterygopsis maclachlani* Stein, 1874 and *Psilopteryx psorosa* Kolenati, 1860, which the author states as common in these habitats. However, his results were based on the collection of imagoes only. We also did not confirm the High Tatra endemic *Allogamus starmachi* Szczesny, 1967 and the Carpathian endemic *Chaetopteryx polonica* Dziedzielewicz, 1889, which occur in streams or lakes mostly at higher altitudes than our sites. The other reason of not recording of above mentioned species and also expectable species *Melampophylax nepos* (McLachlan, 1880) could be the fact that not all the larvae of given genera are determinable to species level.

In contrast to Chvojka (1992) we recorded relatively high proportion of *Apatania carpathica* (especially at site 1 and 7), whereas the author refers to the very rare occurrence of this species (only a few imagoes at one site in the Belianske Tatry). We also found the species *Allogamus auricollis*, *Ecclisopteryx madida*, *Glossosoma conformis*, *Chaetopteryx fusca* and *Philopotamus montanus*, which are according to him spread distributed under 1000 m altitude. Species recorded from the lowest parts of the TANAP also appeared in our disturbed sites; namely *Ecclisopteryx dalecarlica* (site 5), *Serricostoma flavicorne/personatum* (sites 4, 5) and Carpathian endemic *Potamophylax cingulatus depilis* (so far refered for the TANAP only in the species level as *Potamophylax cingulatus*) (sites 2, 6 and 7). This could be caused by the shift of these species upstream due to increase of temperature in these streams (unpublished data). We also recorded Carpathian endemic *Acrophylax sowai* (sites 2 and 3) known from the High and the Low Tatras in Slovakia (Lukáš & Chvojka 2011), which Chvojka (1992) listed as *A. vernalis* Dziedzielewicz, 1912.

*Metrics*

Veľký Šum as a control site had among all the sites average evenness as well as diversity. We can assume, that this site is ecologically balanced, where no obvious changes are occurring. The six disturbed sites were affected in various ways in comparison to site 1.

Deforestation can severely affect species composition of Trichoptera assemblages (Chakona et al. 2009). Generally, taxonomic diversity decreases with disturbance (Stone & Wallace 1998). Higher sediment input due to erosion (Webster et al. 1992) can decrease species richness, density or biomass (Larsen et al. 2009; Couceiro et al. 2011; Angradi 1999). However, changed environmental conditions due to deforestation such as increased insolation, primary productivity and altered thermal regimes (Swift, 1983) create habitats similar to downstream reaches and so enable colonization of several taxa from these reaches (Stone & Wallace 1998). Moreover, sheet wash from the disturbed soil enriched with nutrients can increase abundance of macroinvertebrates (Hernandez et al. 2005). In our study we observed the highest divergence in diversity and evenness at the most impacted sites: the lowest values of these indexes at sites 4 and 5 (the likely explanation is an increase of predatory species of the family Rhyacophilidae at the expense of other species) and the highest diversity at site 3 and 6 (connected likely with shift of species from the lower reaches).

Our results suggest that erosion, resulting in an increase of suspended sediment concentrations, significanly affects feeding activity of benthic invertebrates (Aldridge et al. 1987; Cline et al. 1982; Graham 1990; Krno 2000). We found out a positive correlation between proportion of predators to all the larvae with known feeding ecology and TAM, what is related to increased erosion in disturbed streams. Some authors characterize sedimented sites in general by a reduced density of prey items (Peckarsky 1984) or by a reduced proportion of predators in macroinvertebrate assemblages (Larsen & Ormerod 2010). However, we explain a higher proportion of caddisfly predators at these sites (3, 4, 5 and 6) by an increase of Oligochaeta and Chironomidae (unpublished data), which represent their food source and which increase in the abundance in response to higher (autochthonous) fine sediment loads (Gray & Ward 1982; Larsen et al. 2009; Dudgeon 1994). A similar increase of macroinvertebrate predators due to increased sediment input was observed by Miliša et al. (2010). They supposed that a disrupted community may be easier to prey on and this is the reason why the predators thrive during the disturbance. We found a negative correlation between predators and evenness, which is most apparently seen at the most impacted sites, corresponding to the most disrupted community. At site 7, which was affected not only by windstorm, but also by the fire in 2006, we recorded a very low proportion of predators and very high evenness. This could be due to valley shape, which is not steep enough to allow high erosion. We also recorded a negative correlation between the proportion of passive filter feeders to all the species and UFPOM and the lowest deal of passive filter feeders at sites 4 and 5. This corresponds to several other studies stressing adverse effect of increased siltation to benthic invertebrates (e.g., Wood & Armitage 1997; Weigelhofer & Waringer 2003) or exactly to caddisflies (e.g., Hedrick et al. 2010; Minshall 1984; Monaghan et al. 2001; Pollard & Reed 2004).

At site 2 we recorded the highest proportion of grazers/scrapers, which usually occur in unshaded streams with supposed high amount of microscopic epilithic algae (Wiberg-Larsen et al. 2000). The second highest proportion was at site 7, which is situated in an open area like the most impacted sites 4 and 5, where the proportion of this feeding type was lower, but unlike them it was not so affected by erosion due to shallow vee. The control site had also a relatively low proportion of grazers/scrapers, but the stream was more shaded. At this site we observed the highest proportion of shredders, which prefer forested to open floodplain due to higher detrital food resources (Arscott et al. 2003), but the difference was insignificant low in comparison with the disturbed sites. We suppose that accumulation of LWD at impacted sites could represent a source of BOM, which shredders feed on (Huryn & Wallace 1987; Boulton & Lake 1992; Minshall et al. 1992; Angradi 1996; Voelz & Ward 1996; Wallace et al. 1999).

*Effects of river morphology on species composition*
Shading and LWD explained large variance of the species composition in studied streams. The most impacted sites (3, 4, 5 and 6) with species of genera *Ecclisopteryx*, *Sericostoma*, *Philopotamus* and most species of the genus *Rhyacophila* were positively correlated with LWD. Many of these belong to known wood associated fauna: *Sericostoma personatum* is a facultative xylophagous, wood serves as an attachement point for *Philopotamus montanus* and *P. ludificatus* (Hoffmann & Hering 2000), *Rhyacophila tristis* uses dead wood for searching of prey (Dittmar 1955). Overhanging boughs need for their development especially species *Chaetopteryx fusca* and *Halesus rubricollis*. These species were most closely associated with recorded species of the genus *Drusus* at site 1. Here, based on CCA analysis, Carpathian endemic *Acrophylax sowai* was closely matched to site 1 (Fig. 4), although it was not recorded at the control site. However, *A. sowai* occurs, according to Szczesny (2007), in similar habitats to *Rhyacophila glareosa*, which was recorded at site 7 only. Similarity of sites 1 and 7 is caused mainly by the presence of *Apatania carpathica* and *A. fimbriata*. They are according to Graf et al. (2008) eu-hypocrenal species, so zonation preference could probably be a more important factor than other environmental variables influencing their occurrence (as both sites had the highest slope and the shortest distance to the spring (Bulánková et al. 2013). However, not all the changes in species composition can be explained by an alteration in the hydromorphology. Deforestation due to windstorm causes other significant changes in environmental factors, such as temperature regime (Swift & Messer, 1971). Nevertheless, this is not the subject of this article, but the matter of our further analyses.

## Conclusion

Erosion, shading and LWD were considered the main factors influencing caddisfly assemblages at our study sites. In comparison to control and sites less affected by erosion (1, 2 and 7) the sites influenced by erosion (3, 4, 5, 6) had an increased proportion of predators, mainly species *Rhyacophila tristis*. Moreover, at sites 4 and 5 we recorded the lowest diversity, evenness and proportion of passive filter feeders. However, sites 3 and 6 had very high diversity and together with sites 4 and 5 are closely connected with other important effect of windstorm – an increased amount of LWD, which represents a food source, provides shelters and can have other positive effects on aquatic biota in streams (e.g., Hoffmann & Hering 2000).

The influence of windstorm is thus a combination of negative and positive effects, which the benthic invertebrates gradually adapt to. In disturbed streams considerable succession in various directions is in progress, whereas caddisfly assemblage structure at the control site is in relative ecological equilibrium (expressed also by average values of diversity). Analyses of other important factors, for example increased temperature in some disturbed streams (unpublished data) are required to help to better understand the responses of caddisfly assemblages in this unique high-mountain ecosystem, which is vulnerable to climate change.

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