FINE SCALE PATTERN OF TRUE BUG ASSEMBLAGES (HETEROPTERA) ACROSS TWO NATURAL EDGES

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The spatial pattern of epigeic true bug assemblages was studied across natural edges at a forest steppe region in Hungary. Two edges were investigated: a poplar forest – open sand grassland (site one) and a juniper forest – open sand grassland (site two). The abundance and variety of true bugs was sampled by pitfall traps. Altogether 1396 adult individuals belonging to 49 species were collected. Some species occurred exclusively at forest habitats and significant indicator species were found in both forest and grassland habitats. However, distinct Heteroptera assemblages were found only at grasslands. The response of the grassland Heteroptera assemblages to the proximity of the edge differed between the two sites. At site one, an approximately seven-meter wide transition zone could be observed from the grassland into the forest, which was characterized by the dissolution of the grassland true bug assemblage. The species richness decreased significantly toward the edge. At site two, an abrupt change in the structure of the Heteroptera assemblage denoted a sharp boundary near to the edge in the juniper forest. The species richness did not changed toward the juniper edge. We found that the cover of dicotyledonous plants, mosses and lichens, bare soil surface and leaf litter had a significant joint effect on true bug assemblages. The characteristic species of the grasslands responded differently to the edges. Specialist herbivour bugs seemed to be affected more strongly and negatively by edges than generalist herbivorous and predaceous species.

Key words: juniper, poplar, open sand grassland, epigeic

INTRODUCTION

The term edge effect was introduced to describe the increase in the abundance and species diversity in the transition zone between two habitats (Leopold 1933, Odum 1971). At present, the changes in both biotic and abiotic conditions, taking place at the transition zones between adjacent habitats, are referred to as edge effects (Murcia 1995). The edges may influence the distribution and abundance of populations (Molnár *et al.* 2001), the interactions between populations (Ferguson 2004) and the structure of communities (Burgess *et al.* 2001). The edge may have positive (Máthé 2006), negative (Bieringer & Zulka 2003) and neutral (Durăes *et al.* 2005) effects on organisms.

A number of studies contributed to the knowledge about patterns of various arthropod taxa across natural edges (e.g. Magura & Tóthmérész 1997,1998, Kotze & Samways 2001, Horváth *et al.* 2002, Taboada *et al.* 2004, Durăes

et al. 2005, YU *et al.* 2006, WERMELINGER *et al.* 2007, HOCHKIRCH *et al.* 2008, GAL-LÉ & TORMA 2009), however, only a few papers focused on true bugs (e.g. CAPPUC-CINO & ROOT 1992, MÜLLER *et al.* 2007), especially on epigeic true bugs. Col-lecting true bugs from the ground-level generally needs great effort and has trifling result compared with collecting from the vegetation (RÉDEI *et al.* 2003). However, epigeic true bugs are characteristic of, and rather abundant in dry, sandy habitats. SPUNGIS (2005) found a higher species richness of epigeic true bugs then plant dwelling true bugs at a coastal dune area. Numerous studies (e.g. GALLÉ *et al.* 1988, HARMAT 1984, KERÉNYI, 1997, TORMA & KÖRMÖCZI 2009; GALLÉ *et al.* 2010) recommended epigeic true bugs for community studies at dry sand grasslands.

The aims of this study were a) to reveal how the species richness and compo-sition of epigeic true bug assemblages changes across two natural sharp edges b) to find correspondence between the distance from the edge of the habitat properties of grasslands and Heteroptera assemblage patterns c) to detect indicator species in the different habitats and d) to reveal the influence of the edges on the abundance of characteristic species.

MATERIALS AND METHODS

Study site and sampling

Present study was carried out near Bugac village in the central part of the Danube–Tisza interfluve, which is situated at a higher elevation than the rest of the lowland, as a result, of a large Quaternary alluvial fan of the Danube, composed of sand, sandy loess and loess in restricted areas (TÖRÖK *et al.* 2003). The area lies in the warm temperate zone with an annual mean temperature be-tween 10.2 and 10.8 °C; the mean annual precipitation is approximately 550 mm with two precipita-tion peaks (spring and autumn) and a drought in summer, therefore the climate is strongly semiarid (TÖRÖK *et al.* 2003).

The remnants of the natural vegetation consisting of open grasslands (*Potentillo–Festucetum va-ginatae*) and juniper-poplar forests (*Junipero–Populetum albae*) are embedded in agricultural fields. The composition of juniper-poplar forests varies widely: some patches consist almost exclusively of juniper, composing a dense scrubland whereas in other patches junipers are absent (VIDÉKI 2004).

Two sharp vegetational edges were studied: poplar forest – open sand grassland (site 1: N 46°43.48', E 19°39.35') and juniper forest – open sand grassland (site 2: N 46°43.49', E 19°40.26'). The distance betwen the sites was aproximately two kilometers. At site one, the forest was character-ised by high poplar trees and accordingly by the presence of thick deciduous leaf litter. Junipers oc-curred sparsely. At site two, the approximately 2–3 meter high juniper bushes formed a closed canopy layer, and the ground level was covered with Juniper leaf litter. The dominant monocotyle-donous plant species of the grasslands were *Festuca vaginata*, *Stipa borysthenica* at site one and *Stipa borysthenica*, *Poa bulbosa* and *Koeleria glauca* at site two. The most dominant dicotyledonous plants of both grasslands were *Arenaria serpyllifolia*, *Euphorbia cyparissias* and *Alkanna tinctoria*. The high coverage of mosses and lichens (*Tortula ruralis*, *Cladonia magyarica*, *C. rangiferina*) was typical at both sites.

Although, sweep netting or suction sampling are generally used to sample true bug assemblages in grasslands (STANDEN 2000, COSCARON *et al.* 2009), but we used pitfall traps (65 mm in diameter, filled with ethylene glycol as killing-preservative solution) for several reasons. Sweep netting is not a suitable sampling method in the case of such a fine scale. Suction sampling collects arthropods from both the vegetation and ground surface (SAMU & SÁROSPATAKI 1995), but it is difficult to use in the case of loose sand soil, because of the great amount of sand sucked during the collecting.

At both two sites, 20 groups of traps arranged in rows were placed parallel to the edge (Fig. 1). Each group of traps contained 5 pitfall traps the same distance from the edge. The intertrap distance within the groups of traps was two meters. The distance between the groups of traps was one meter to obtain a fine resolution. To maximize the inter trap distance, every second group of traps was shifted by one meter parallel to the edge. The pitfall traps were open for two-week periods in April, May, July and September, in the years of 2004 and 2005. The majority of individuals (82%) were collected in the summer, thus, we did not analyze the data of the sampling periods separately. Only adult individuals were identified to species level. The nomenclature of true bugs followed the work of KONDOROSY (1999).

To characterize the properties of habitats, the number of plant species, the percentage cover of leaf litter and vegetation (mosses and lichens, monocotyledonous plants, dicotyledonous plants, separately) was recorded in 1×1 meter quadrats close to the pitfall-traps. The mean of data of the quadrats the same distance from the edge was used for analyses.
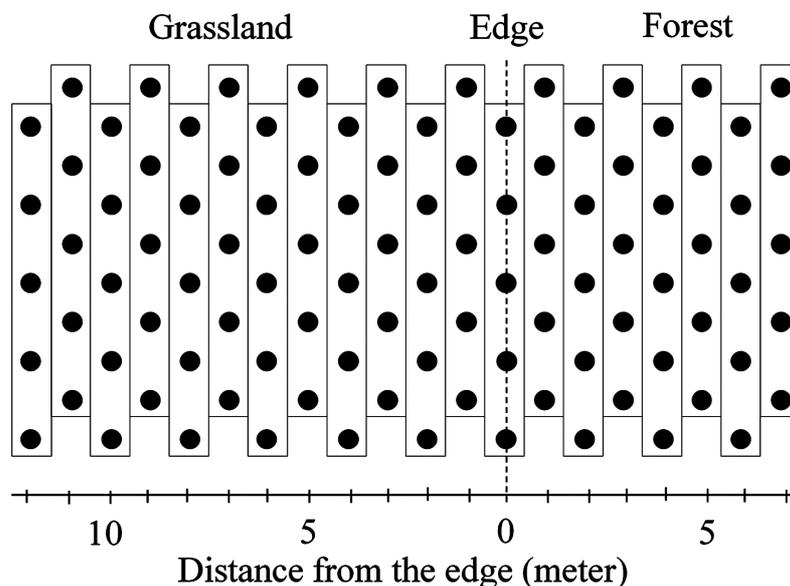

Fig. 1. The scheme of the sampling design. The black circles symbolise the pitfall traps. The groups of traps are surrounded by boxes. Twenty groups of traps containing 5 pitfall traps were placed parallel to the edge. The distance between the traps was two meters; the distance between the groups of traps was one meter. Every second group of traps was shifted by one meter. The data belonging to a group of traps was pooled, and used as a sample unit

Data analysis

The data of pitfall traps at the same distance from the edge were pooled for the analyses. Thus we gained 20 samples (equal to the number of the groups of traps) perpendicular to the edge with a distance of one meter between the subsequent samples.

To detect the changes in the structure of the true bug assemblages, the Chao's Jaccard-type and the Morisita-Horn indices were calculated between the adjacent samples using EstimateS 8.0 software (COLWELL 2004). The Chao's Jaccard-type index was used, as this index is less biased than the classic Jaccard index (CHAO *et al.* 2005). The Morisita-Horn index was applied as a widely used abundance-based index (MAGURRAN 2004).

To show the natural grouping of samples Non-metric Multidimensional Scaling (NMDS) with Bray-Curtis dissimilarity was used. NMDS is recommended by LEGENDRE and LEGENDRE (1998) and OKSANEN (2011), because in the case of NMDS various dissimilarities and distances can be applied and it uses rank information and maps the ranks non-linearly onto ordination space. To determine the main habitat variables which influenced the distribution patterns of true bugs constrained ordination (RDA – Redundancy Analysis) was used (LEPŠ & ŠMILAUER 2003). The marginal and partial effects of the variables were calculated. Marginal effects denoted the variability explained by a variable without considering other variables, and partial effects denoted the variability explained by a given variable after removing the confounding effect of one or more other variables (SCHWEIGER *et al.* 2005). Marginal and partial effects were tested for significance by Monte Carlo procedure with 1000 permutations. To visualise the effects of the significant variables, they were fitted to the scatterplot of NMDS. For the fitting the envfit function was used (OKSANEN 2011). The direction of the arrow shows the direction of the gradient, and the length of the arrow is proportional to the correlation between the variable and the ordination (OKSANEN 2011). We used the Vegan R package (OKSANEN *et al.* 2006) for the ordination methods.

To detect species related to the different habitats, we used indicator value approach (DUFRENE & LEGENDRE 1997). The habitats were identified primarily as grassland and forest according to the vegetation. Indicator values were calculated using the labdsv package (ROBERTS 2010) of R statistic. To test the statistical significance of indicator values we used the Monte Carlo procedure with 1000 permutations.

According to several studies (e.g. MURCIA 1995, BIERINGER & ZULKA 2003), edge effects do not necessarily lead to linear response patterns. Therefore we used Sperman's coefficient of rank correlation to analyse correlations between the species richness of true bugs, the measured habitat variables and the distance from the edge. We restricted the correlation analyses to the grassland samples to avoid spurious trends because of great differences between the habitat properties of forests and grasslands. Since we performed a number of tests on the same data set, a Bonferroni type error adjustment was used (SOKAL & ROHLF 1995). The correlations were calculated in R statistic (R DEVELOPMENT CORE TEAM 2007).

To reveal the species influenced by the edges, simple linear regressions were applied on the abundance of true bugs against the distance from the edge. Due to the indicator values, only the relevant grassland indicator species and additionally the frequent species were analised. The abundance of the species per samples were log (abundance+1) transformed.

RESULTS

Altogether 1396 adult individuals belonging to 49 species were collected (Table 1). At site one, a continuous change of the assemblage structure was observed which started approximately 7 m far from the edge in the grassland (Fig. 2a). At site two, changes in the assemblage structure (Fig. 2b) were observed in the forest and the assemblage structure changed abruptly, contrary to site one.

The grouping of samples in the ordination space (Fig. 3) confirmed the relevant changes of the assemlage structures. At site one, the samples situated far from the edge in the grassland were grouped whereas other samples which were situated in the forest and near the edge in the grassland were scattered. At site two, the sam-
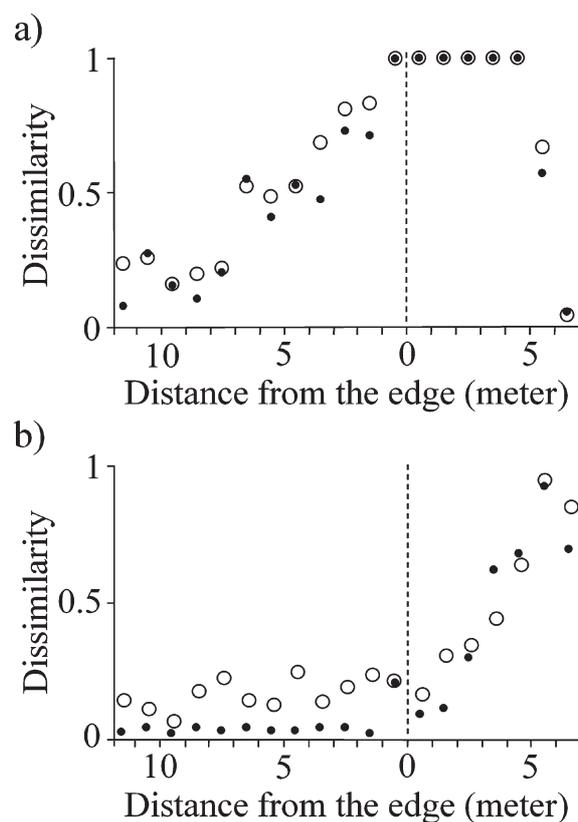

Fig. 2. The figures show the changes in the assemblage structure of true bugs according to the complement of Morisita-Horn (black circles) and the Chao's Jaccard-type (empty circles) indices as dissimilarities between the subsequent samples at site 1 (a) and at site 2 (b). The vertical line marks the edge

Table 1. The abundance and the indicator value of the collected Heteroptera species. G = grassland, F = forest, S = schrubland the number of samples in bracket, No: the number of individuals, InV: the indicator value of species. The significance of indicator values were tested with the Monte-Carlo procedure (1000 iterations). The level of significance: * < 0.05, ** < 0.01, *** < 0.001.

Taxa	Site one				Site two			
	G (13)		F (7)		G (13)		S (7)	
	No	InV	No	InV	No	InV	No	InV
Ceratocombidae								
Ceratocombus coleoptratus (ZETTERSTEDT, 1819)			4	0.57**				
Tingidae								
Acalypta gracilis (FIEBER, 1844)	47	0.85**			26	0.74**	2	0.02
Acalypta marginata (WOLFF, 1804)	2	0.15					1	0.14
Agramma minutum HORVÁTH, 1874	1	0.08						
Catoplatus carthusianus (GOEZE, 1778)					1	0.08		
Dictyla echii (SCHRANK, 1781)	1	0.08			1	0.03	1	0.09
Dictyla rotundata (HERRICH-SCHÄFFER, 1835)	6	0.38			1	0.08		
Nabidae								
Alloeorrchynchus flavipes FIEBER, 1836	1	0.08						
Nabis pseudoferus REMANE, 1949	1	0.08			2	0.08	1	0.07
Anthocoridae								
Xylocoris cursitans (FALLÉN, 1807)					1	0.08		
Reduviidae								
Coranus kerzhneri P. PUTSHKOV, 1982	16	0.77**			34	0.66	3	0.06
Piesmatidae								
Piesma maculatum (LAPORTE, 1832)					1	0.08		
Berytidae								
Neides tipularius (LINNAEUS, 1758)	13	0.47	1	0.02	35	0.46	19	0.43
Lygaeidae sensu lato								
Acompus rufipes (WOLFF, 1804)							1	0.14
Aellopus atratus (GOEZE, 1778)	84	0.77**			118	0.93***	5	0.02
Arocatus longiceps STÅL, 1872	1	0.08						
Dimorphopterus doriae (FERRARI, 1874)	16	0.31			1	0.08		
Emblethis ciliatus HORVÁTH, 1875	23	0.69*			511	0.86***	43	0.10
Eremocoris fenestratus (HERRICH-SCHÄFFER, 1839)							1	0.14
Graptopeltus lynceus (FABRICIUS, 1775)	2	0.15						
Lygaeosoma sardeum SPINOLA, 1837	1	0.08			8	0.21	2	0.05
Melanocoryphus tristrami (DOUGLAS et SCOTT, 1868)					2	0.08	1	0.07
Ortholomus punctipennis (HERRICH-SCHÄFFER, 1839)	3	0.23			3	0.15		

Table 1 (continued)

Taxa	Site one				Site two			
	G (13)		F (7)		G (13)		S (7)	
	No	InV	No	InV	No	InV	No	InV
Peritrechus nubilus (FALLÉN, 1807)	1	0.08						
Pionosomus opacellus HORVÁTH, 1895	62	0.77[**]			105	0.63	26	0.32
Plinthisus longicollis FIEBER, 1861	2	0.15						
Raglius alboacuminatus (GOEZE, 1778)					1	0.08		
Rhyparochromus vulgaris (SCHILLING, 1829)			1	0.14			2	0.29
Stygnocoris rusticus (FALLÉN, 1807)							1	0.14
Stygnocoris sabulosus (SCHILLING, 1829)							3	0.43[*]
Tropidophlebia costalis (HERRICH-SCHÄFFER, 1850)	2	0.15			5	0.15	4	0.26
Xanthochilus quadratus (FABRICIUS, 1798)	13	0.54[*]			23	0.56	3	0.08
Coreidae								
Bathysolen nubilus (FALLÉN, 1807)	1	0.08						
Ceraleptus gracilicornis (HERRICH-SCHÄFFER, 1835)							2	0.29
Ceraleptus obtusus (BRULLÉ, 1839)	2	0.15			1	0.08		
Gonocerus juniperi (HERRICH-SCHÄFFER, 1839)							1	0.14
Spathocera obscura (GERMAR, 1842)					1	0.08		
Syromaster rhombeus (LINNAEUS, 1767)	1	0.08			5	0.11	3	0.15
Rhopalidae								
Brachycarenus tigrinus (SCHILLING, 1817)	1	0.08			6	0.16	3	0.14
Chorosoma gracile JOSIFOV, 1968	4	0.31			17	0.69[*]		
Rhopalus parumpunctatus (SCHILLING, 1817)	4	0.31			5	0.31		
Aradidae								
Aradus depressus (FABRICIUS, 1794)			3	0.29				
Cydnidae								
Cydnus aterrimus (FÖRSTER, 1771)					1	0.08		
Scutellaridae								
Eurygaster maura (LINNAEUS, 1758)	2	0.15			1	0.08		
Pentatomidae								
Aelia acuminata (LINNAEUS, 1758)	1	0.02	1	0.09	3	0.14	1	0.05
Dolycoris baccarum (LINNAEUS, 1758)					1			
Eusarcoris aeneus (SCOPOLI, 1763)	1	0.08						
Menaccarus arenicola (SCHOLTZ, 1846)	12	0.46					8	0.28
Neottiglossa leporina (HERRICH-SCHÄFFER, 1830)					2	0.15		

ples situated near the edge in the Juniper forest were grouped with the samples of the grassland. Both the species richness and the abundance were very low in the forest habitats and the occurrence of species in the samples were sporadic, hence a consistent species composition was not found, distinct assemblages existed only in grassland habitats. The constrained axes of RDA explained 73.52% of the total variance. The implemented permutation test indicated the significant joint effect of
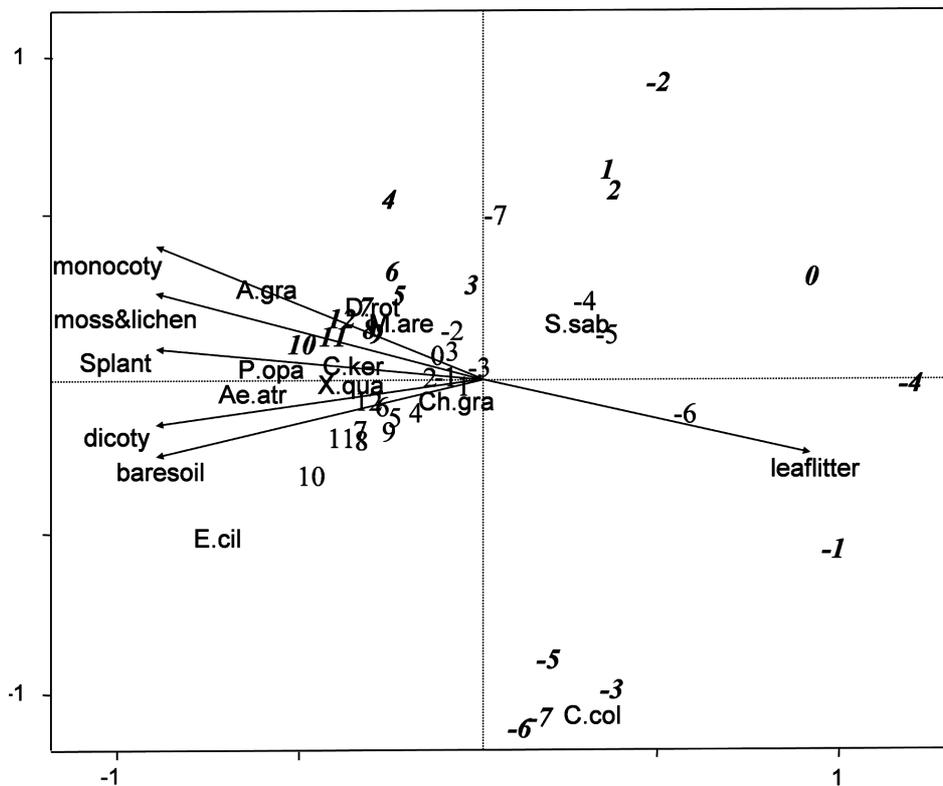
Fig. 3. The scatterplot of NMDS based on Bray-Curtis similarity shows the natural grouping of samples and the relevant species. The numbers mark the samples and mean their ditance from the edge (in meters). Positive numbers mark the samples of grasslands, negative numbers mark the samples of forests. Numbers in italic mark the samples at site one and numbers in bold mark the samples at site two. The arrows symbolise the effects of the habitat variables (plant species richness, percentage cover of monocotyledonous plants, dicotyledonous plants, mosses and lichens, leaf litter, bare soil surface). The variables were fitted passively onto the ordination diagram to visualise their effects. The abbreviations of the characteristic species: A.gra = *Acalypta gracilis*, Ae.atr = *Aellopus atratus*, C.col = *Ceratocombus coleoptratus*, C.ker = *Coranus kerzhneri*, Ch.gra = *Chorosoma gracile*, D.rot = *Dictyla rotundata*, E.cil = *Emblethis ciliatus*, M.are = *Menaccarus arenicola*, P.opa = *Pionosomus opacellus*, S.sab = *Stygnocoris sabulosus*, X.qua = *Xanthochilus quadratus*

Table 2. The table shows the explained variance of the habitat parameters (plant species richness, percentage cover of monocotyledonous plants, dicotyledonous plants, mosses and lichens, leaf litter, bare soil surface) according to RDAs. The marginal and partial effects of the parameters on the true bug assemblages are given in separated columns. Significance of parameters was tested by Monte-Carlo procedure with 1000 permutations.

Habitat parameters	Variance explained (%)	
	Marginal effect	Partial effect
Monocotyledonous plants	20.28 (p = 0.005)	3.00 (p = 0.032.)
Dicotyledonous plants	57.48 (p = 0.005)	3.21 (p = 0.022)
Mosses and lichens	8.20 (p = 0.055)	4.61(p = 0.013)
Leaf litter	36.55 (p = 0.005)	4.54 (p = 0.015)
Bare soil surface	58.04 (p = 0.005)	3.41 (p = 0.032)
Plant species richness	34.54 (p = 0.005)	1.90 (p = 0.090)

constraints (pseudo-F = 15.27, p = 0.005, based on Monte-Carlo procedure with 1000 permutations). According to variance partitioning, most of the measured environmental variables influenced the bug assemblages significantly, however, their effects overlapped highly (Table 2).

The correlation analyses (Table 3) showed significant correlations between the species richness of true bugs, the distance from the edge and some of the measured habitat variables at site one. The species richness of true bugs correlated positively with the distance from the edge and the coverage of mosses and lichens but

Table 3. Sperman's rho values show correlations between species richness of true bugs (Bugrich), the habitat parameters (plant species richness (Plarich), percentage cover of monocotyledonous plants (Monoc), dicotyledonous plants (Dicot), mosses and lichens (MoLi), leaf litter (Litter), bare soil surface (Soil)) and the distance from the edge (Dist). The triangle above contains the Sperman's rho values at site two. The triangle below contains the Sperman's rho values at site one. The level of p- values (*p < 0.05; **p < 0.01; ***p < 0.001) are given after Bonferrerroni type error adjustments. Significant values before the type error adjustments are marked in bold.

| site 1 \ site 2	Monoc	Dicot	MoLi	Litter	Soil	Plarich	Dist	Bugrich
Monoc		–0.07	–0.28	–0.46	–0.41	0.34	0.11	–0.02
Dicot	0.02		–0.25	–0.47	0.37	0.02	**0.87****	0.42
MoLi	–0.37	**0.59**		–0.20	–0.30	–0.43	–0.50	–0.51
Litter	0.14	**–0.84****	**–0.87****		–0.04	–0.19	–0.46	0.24
Soil	–0.28	**0.60**	0.42	**–0.62**		0.28	0.47	0.14
Plarich	–0.23	**0.77***	0.34	**–0.65**	**0.59**		0.19	0.19
Dist	–0.34	**0.62**	**0.92*****	**–0.86****	0.43	0.34		0.34
Bugrich	–0.25	**0.66**	**0.91*****	**–0.90*****	0.40	0.43	**0.93*****	
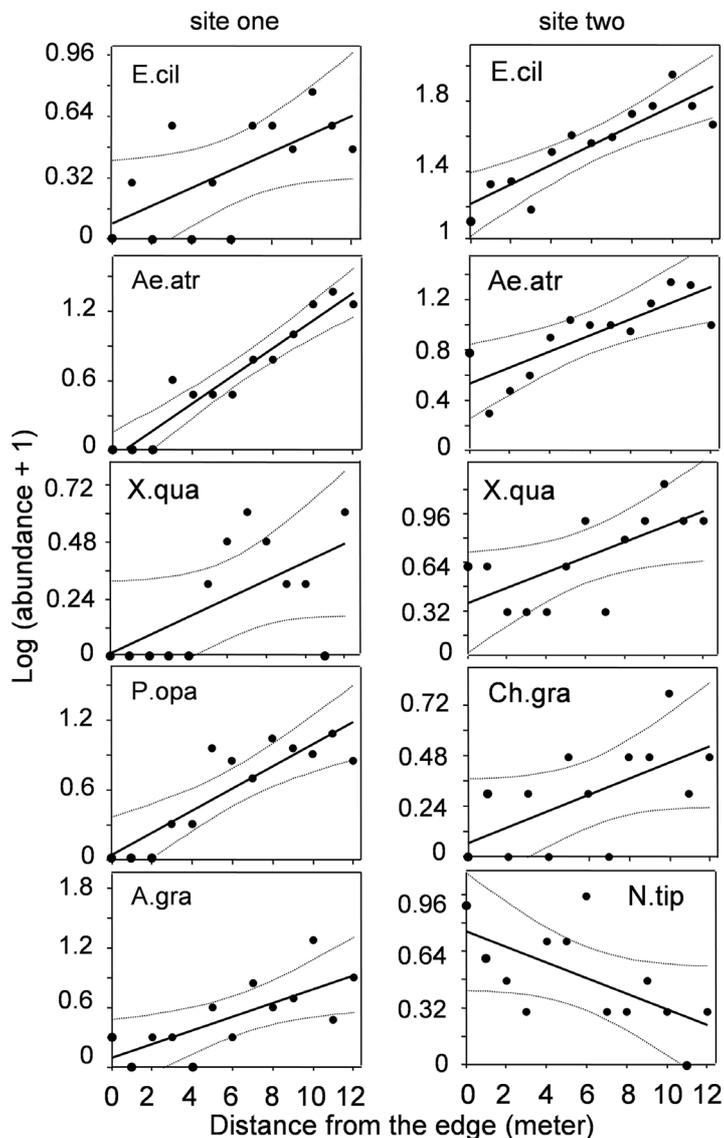
Fig. 4. The significant relations between the abundance of true bug species and the distance from the edge. At site one: E.cil ($r^2 = 0.42$, $F_{1,11} = 8.085$, $p = 0.0159$); Ae.atr ($r^2 = 0.92$, $F_{1,11} = 124.20$, $p < 0.0001$); X.qua ($r^2 = 0.37$, $F_{1,11} = 6.339$, $p = 0.0286$); P.opa ($r^2 = 0.76$, $F_{1,11} = 34.83$, $p = 0.0001$); A.gra ($r^2 = 0.54$, $F_{1,11} = 12.85$, $p = 0.0043$); site two: E.cil ($r^2 = 0.78$, $F_{1,11} = 38.55$, $p < 0.0001$); Ae.atr ($r^2 = 0.66$, $F_{1,11} = 21.06$, $p = 0.0008$); X.qua ($r^2 = 0.37$, $F_{1,11} = 6.339$, $p = 0.0286$); Ch.gra ($r^2 = 0.45$, $F_{1,11} = 9.17$, $p = 0.0114$); N.tip ($r^2 = 0.39$, $F_{1,11} = 6.971$, $p = 0.0229$); The error bands show the 95% confidence for the fitted line. The abbreviations of the species: A.gra = *Acalypta gracilis*, Ae.atr = *Aellopus atratus*, Ch.gra = *Chorosoma gracile*, E.cil = *Emblethis ciliatus*, N.tip = *Neides tipularius*, P.opa = *Pionosomus opacellus*, X.qua = *Xanthochilus quadratus*

negatively with the leaf litter. At site two, the species richness of true bugs did not correlate with any variables. We found significant correlation between the distance from the edge and the cover of dicotyledonous plants, only.

In spite of the fact that several species were observed in the forest habitats, only the indicator values of *Ceratocombus coleoptratus* and *Stygnocoris sabulosus* were significant, according to the indicator species analyses (Table 1). As it was expected, the highest number of species were related to grassland habitats. The most abundant species were *Emblethis ciliatus*; *Aellopus atratus*; *Pionosomus opacellus* and *Acalypta gracilis*, these had high indicator values in the case of both sites. Besides the above species, *Coranus kerzhneri* and *Xanthochilus quadratus* had significant indicator values in the case of site one and *Chorosoma gracile* in the case of site two.

The relations between the abundance of species and the distance from the edge showed that species responded differently to the edge (for the significant relations see Fig. 4). The abundance of *E. ciliatus*, *Ae. atratus* and *X. quadratus* decreased significantly toward the edge at both sites. *P. opacellus* and *A. gracilis* responded negatively to the poplar edge, but they were not affected by the proximity of the juniper edge ($r^2 = 0.11$, $F_{1,11} = 1.352$, $p = 0.2696$ and $r^2 = 0.00$, $F_{1,11} = 0.0435$, $p = 0.8385$, respectively). *Ch. gracile* seemed to respond negatively to edges, however, it was not analysed at site one, because of its low abundance. *Neides tipularius* showed positive relation to the proximity of the edge at site two however, at site one, significant relationship was not found ($r^2 = 0.10$, $F_{1,11} = 1.295$, $p = 0.7293$). *Coranus kerzhneri* was not influenced by the distance from the edges ($r^2 = 0.30$, $F_{1,11} = 4.692$, $p = 0.0531$ and $r^2 = 0.14$, $F_{1,11} = 1.749$, $p = 0.2129$ at site one and site two, respectively).

DISCUSSION

The fine scale pattern of Heteroptera assemblages was studied across two natural sharp vegetational edges: poplar forest–open grassland and juniper forest–open grassland. Among the species associated with forest habitats, *C. coleoptratus* lives under dead vegetation, litter and wet mosses at various habitat types (RÉDEI & HUFNAGEL 2002). This species, presumably, found the adequate humidity in deciduous forests at such xeric areas. At site two more species were collected exclusively in the scrubland as several of them e.g. *E. fenestratus*, *G. juniperi* are associated to junipers (WAGNER 1966, PÉRICART 1998). The most collected species were associated to the grassland habitats. These species typically occur in the sandy grasslands of the Kiskunság (BAKONYI & VÁSÁRHELYI 1987).

In spite of the similar species composition, the response of the grassland assemblages to the edge differed between the sites. In general, grassland insect species respond negatively to forest or schrubland edges (BIERINGER & ZULKA 2003, SAMWAYS & MOORE 1991). The decreasing species richness of true bugs toward the edge at site one also suggested a negative edge effect. Moreover, the abundance of the analysed species, expect for *C. kerzhneri*, decreased toward the poplar edge. In accordance with these decreasing trends, the change of the assemblage structure denoted an approximately 7 m wide transition zone in the grassland. In the case of site two, however, we found that both the species richness and the assemblage structure of true bugs changed abruptly, close to the edge in the juniper forest. These patterns suggested that the true bug assemblage responded strongly to the change of the habitat quality, but weakly to the proximity of the edge (DURĂES *et al.* 2005).

The different responses of grassland bug assemblages to the edge were probably due to the influence of high trees at site one e.g. on the light regime, leaf litter, vegetation coverage on the nearby part of the grassland. Several authors (e.g. SPARKS & GREATOREX-DAVIES 1992, ENTLING *et al.* 2007, GOSSNER 2009) reported the influence of shading on different arthropods. Moreover, the light level also affected the plant species resulting in linked patterns of light regime, plant and insect species richness at edge zones GREATOREX-DAVIES *et al.* 1994). The impact of various vegetation properties e.g. vegetation species composition (SANDERSON *et al.* 1995, TORMA *et al.* 2010), vegetation structure and flower abundance (ZURBRÜGG & FRANK 2006), plant diversity or plant species richness (FRANK & KÜNZLE 2006) on true bug assemblages is well known. The coverage of mosses and lichens can influence the specialist herbivores feeding on them, e.g. the *Acalypta* species (RÉDEI *et al.* 2004) and it may affect the diversity and population density of epigeic true bugs in general as they may serve as shelters for the species (SPUNGIS 2005). On the other hand, the bare soil surface and leaf litter also had a great impact on the bug assemblages. The movement of true bugs is mainly walking (DOLLING 1991), therefore the complexity of surface is an important factor for their locomotion (YANG 2000). The proportion of bare soil and the type of the leaf litter as a property of the soil surface may be important factors for the ground-dwelling bugs in this sense.

However, various feeder groups (e.g. plant feeders, predators, etc.) presumably are not influenced by the same factors, moreover influential factors may differ between the degrees of feeding specialization, e.g. between specialist and generalist herbivores (JONSEN & FAHRIG 1997). For example, the decreasing patterns of the specialist herbivorous species *E. ciliatus* (feeds on *Euphorbia spp.*) and *Ae. atratus* (feeds on Boraginaceae) may be explained by the decrease in the density of

their host plants. Note, the dominant dicotyledonouse plants were *Euphorbia cyparissias* and *Alkanna tinctoria* and the coverage of dicotyledonous plants also decreased toward the edge at both sites. Whereas the abundance of zoophagous (e.g. *C. kerzhneri*) and polyphagous herbivore (e.g. *N. tipularius*) species did not decrease or it decreased only towards the poplar edge (e.g. *P. opacellus*). For zoophagous and polyphagous herbivore species the habitat structure is an important factor (DI GIULIO *et al.* 2001) which changed notably in the case of the poplar edge.

The edge effect is taxon dependent (DANGERFIELD *et al.* 2003) so it was not surprising that different species responded differently to the edges. In spite of this fact, the observed changes of assemblage structure of true bugs seemed to be general patterns at small scale. Both similar sharp boundaries and dissolving zones of epigeic true bug assemblages were observed previously in sand grasslands (TORMA & KÖRMÖCZI 2009, GALLÉ *et al.* 2010). However, a higher species richness in the transition, than adjacent habitats and species turnover in the transition between two habitats were not observed. These patterns seem to be not typical at such small scale. It seems that the general small-scale patterns of true bug assemblages may be driven by the variation in the environmental conditions than by biotic interactions. Sharp changes of environmental conditions cause a sharp boundary while along a gradual change of environmental conditions epigeic true bug assemblages of sand grasslands dissolved.

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