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CHANGES IN THE DIVERSITY OF DRY CALCAREOUS GRASSLANDS AFTER ABANDONMENT OF TRADITIONAL MANAGEMENT IN DEVÍNSKA KOBYLA NATIONAL NATURE RESERVE (SOUTHWESTERN SLOVAKIA, CENTRAL EUROPE)

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Abstract

Rendeková A., Miškovic J., Mičieta K., Michalová M., Eliašová M., Ballová D.: Changes in the diversity of dry calcareous grasslands after abandonment of traditional management in Devínska Kobyla National Nature Reserve (Southwestern Slovakia, Central Europe). Ekológia (Bratislava), Vol. 39, No. 2, p. 101–115, 2020.

This study is focused on the evaluation of the values of the diversity indices of semi-natural dry calcareous grasslands of the *Festuco-Brometea* class in the area of Devínska Kobyla National Nature Reserve 50 years after abandonment of traditional management (grazing and mowing). The values of the species richness, the Shannon–Wiener index of diversity, the Simpson index, and Pielou's measure of species evenness in the communities in the old and recent data, and the values of the environmental variables based on the Ellenberg indicator values of species were analyzed. For most of these analyses, we used Kruskal–Wallis in R 3.5.1. Compared to the past, we have recorded a significant lower values of the species richness, biodiversity, and species evenness in some communities. In some communities, no significant changes were recorded. Only in one community—*Poo badensis-Festucetum pallentis*, the higher values of Shannon–Wiener index was recorded compared to the past. The analysis of the Ellenberg indicator values revealed a statistically significant higher Ellenberg indicator values for moisture and nutrients and lower values for light in more recent period compared to the old period. The recorded results could be caused by the changes in the management after the year 1965, after abandonment of grazing and the gradual overgrowing by woody species took place in the area. The frequency of occurrence of some woody species (e.g., *Populus alba*, *P. nigra*, and *Robinia pseudoacacia*) in some communities increased compared to the past.

Key words: Devínska Kobyla Mts, *Festuco-Brometea*, succession, vegetation biodiversity conservation.

Introduction

Semi-natural dry calcareous grasslands (*Festuco-Brometea* class) have spread in Central Europe after forest clearing and consecutive traditional land use such as grazing and mowing (Poschlod, WallisDeVries, 2002; Chytrý, 2007). These specific habitats cannot exist without the repeated removal of biomass by grazing, mowing, burning, or by other ways and control of species not adapted to such disturbance regimes. Formation of dry grasslands is influenced by these disturbances and stress caused by extreme habitat conditions (low content of nutrients, low humidity, and high temperature), which allows the existence of many adapted species indirectly by limiting the growth of competitively stronger species (Kubíková, 1999; Münzbergová, 2001; Chytrý, 2007). Because of disturbance regime and long history of development, dry calcareous grasslands belong to the most species-rich types of vegetation ever (Willems, 1983; During, Willems, 1984; Bobbink et al., 1987; Poschlod, WallisDeVries, 2002; Chytrý, 2007; Schrautzer et al., 2009). They also harbor a large number of endangered species (Wolkinger, Plank, 1981; Pipenbaher et al., 2013).

The abandonment of traditional forms of land use such as grazing and mowing over the last decades has brought an increasing concern about the consequences of the resulting expansion of shrubs and trees on species-rich dry calcareous grasslands (Dzwonko, Loster, 1998; Hansson, Fogelfors, 2000; Willems, 2001; Bąba, 2003; Alard et al., 2005; Köhler et al., 2005; Dostálek, Frantík, 2008; Schrautzer et al., 2009; Jacquemyn et al., 2010; Ruprecht et al., 2010; Török, Szitár, 2010; Tälle et al., 2018). Some studies have shown that the abandonment of traditional land use had negative impact on the species richness, diversity, and area of dry grasslands in Central Europe (Poschlod, WallisDeVries, 2002; Galvánek, Lepš, 2008; Schrautzer et al., 2009; Hegedüšová, Senko, 2011; Pipenbaher et al., 2013), and in the last few years, more attention is being devoted to conservation management (Poschlod et al., 1998; Barbaro et al., 2001; Münzbergová, 2001; Willems, 2001; Kahmen et al., 2002; Riecken et al., 2002; WallisDeVries et al., 2002; Bąba, 2004; Masé, 2005; Bornkamm, 2006; Stadler et al., 2007; Dostálek, Frantík, 2008; Peter et al., 2009; Kuzemko et al., 2016). The importance of conservation management and maintenance of traditional ways of farming is visible not only in the case of grasslands but also in the case of various other types of habitats (Mojses, Petrovič, 2013; Špulerová et al., 2016) and other scientists have also devoted their research to changes in various types of vegetation (Palaj, Kollár, 2018).

Devínska Kobyla Mt. (southern part of Male Karpaty Mts., Slovakia, Central Europe) is a unique area with specific environmental conditions. Dry grasslands at Devínska Kobyla Mt. have formed due to agricultural activity and deforestation of the area (Kaleta, 1965; Hajdúk, 1986). Grazing by goats and sheep and mowing were the main factors that influenced the development and the maintenance of the species-rich dry grassland communities at the area. During the mid-20th century, the traditional land use such as grazing, mowing, and burning was ceased. The gradual expansion of shrubs and trees took place in the area (Hegedüšová, Senko, 2011).

Dry grasslands in Devínska Kobyla were studied by Domin (1931); Kaleta (1965); Michalko (1977); Hajdúk (1986); Baláž (1994); Hajdúk (1997); Maliníková (2003), Miškovic, Dúbravcová (2004a;b); Zlínska (2004); Senko et al. (2008) and Hegedüšová, Valachovič (2015). Hegedüšová and Senko (2011) reported changes in species composition of dry grasslands in Devínska Kobyla Mt., but they did not pay attention to changes in diversity indices and environmental conditions. Our study overcomes this gap and also bring new results from survey performed in the last years and help in the management of protected area. The aim of this study is to assess the values of the diversity indices in reference (1964) and recent (2012–2018) phytosociological data. Hypothesis: we expected that the values of the species richness, Shannon–Wiener index of diversity, Simpson index, and Pielou's measure of species evenness in vegetation would be lower in comparison to the past, because of previously mentioned abandonment of traditional management of the area.

Material and methods

Study area

Devínska Kobyla National Nature Reserve is located on the southwestern and western slopes of the Devínska Kobyla Mt., which is the southernmost part of the Little Carpathians Mts. (Slovakia, Central Europe). The entire massif of Devínska Kobyla is part of the Malé Karpaty Protected Landscape Area. In 1964, the Devínska Kobyla State Nature Reserve was established to protect thermo-xerophilous habitats covering the area of 27.97 ha. Later, in 1986, it was enlarged to its current area of 101.12 ha. On the basis of later legal regulations, the reserve was included in the list of national nature reserves under the currently valid name Devínska Kobyla National Nature Reserve (Feráková, Kocianová, 1997).

There is an unusual variety of environmental phenomena concentrated on the area of Devínska Kobyla. It is unique because of its geographical position, history, extraordinary environmental conditions such as topography and climate, and mainly because of its thermophilous grassland- and forest-steppe flora and vegetation (Feráková, Kocianová, 1997; Hegedüšová, Valachovič, 2015).

The study area belongs to the warm climatic region, and it is one of the warmest and driest parts of Slovakia. It has an average 50 summer days per year or more, with a daily maximum temperature of 25 °C and higher. The average duration of sunshine is one of the longest in Slovakia (in the vegetation period, it is 60% = 1,600 h) (Feráková, Jarolímek, 2011).

The most common soil types in the Devínska Kobyla NNR are Cambisols (Calcaric) and Rendzic Leptosol, on deluvium of carbonate–silicate weathered rock. On a smaller area, especially in the territories with the local names Sandberg and Merice, occur soil types Haplic Regosol (Calcaric) on neogene sand and sandstones, while on weathered limestones and dolomites dominate Rendzic Leptosol and Lithic Leptosol. Rarity in the territory of Devínska Kobyla NNR is the occurrence of 1 ha of soils formed on the loess, namely Cutanic Luvisols and Haplic Chernozem, as well as several cubic meters of Rendzic Leptosol on the weathered Tertiary calcareous conglomerates (Feráková, Jarolímek, 2011).

The Devínska Kobyla NNR is also known by the near proximity of a well-known paleontological site Sandberg. It is the old sandpit, where remnants of rocks of the Tertiary Sea, Miocene marine, and nonmarine sediments are situated. The horizontally deposited layers of the Tertiary sea with the estimated age of 14–16 million years can be found here. Fossils of the sea fauna were conserved in Sandberg (Feráková, Kocianová, 1997; Hyžný et al., 2012).

Data source

We have analyzed 153 phytosociological relevés of dry grassland communities of *Festuco-Brometea* class, which includes the dry grassland vegetation of the Eurosiberian steppes and warm regions in the temperate zone of Europe. These plant communities are dominated by grasses and have high proportion of dicotyledonous perennial herbs. They usually grow on nutrient poor calcareous soils and developed secondary on areas of former forests (Chytrý, 2007). We used the relevés of four associations from the *Festuco-Brometea* class: *Poo badensis-Festucetum pallentis*, *Festuco pallentis-Caricetum humilis*, *Festuco valesiacae-Stipetum capillatae*, and *Polygalo majoris-Brachypodietum pinnati*; grasslands of these four associations cover most of the area of Devínska Kobyla NNR. The *Poo badensis-*

Festucetum pallentis association includes open dry grassland communities growing on shallow dry soils on the steep slopes. The stands of the *Festuco pallentis-Caricetum humilis* association usually grow on shallow to moderately deep soils, and the *Festuco valesiacae-Stipetum capillatae* occurs on deeper soils. Among the four analyzed associations, the stands of the *Polygalo majoris-Brachypodietum pinnati* association grow on the relatively deepest soils (Chytrý, 2007; Hegedüšová-Vantarová, Škodová, 2014).

The data were sampled during two time periods with the time span of ~50 years. Data (40 relevés) sampled in 1964 (Kaleta, 1965) are used as reference data (Fig. 1). Recent data (113 relevés) were sampled in 2012–2018 (Miškovic, 2018) (Fig. 2). The number of recorded relevés in the studied vegetation types was as follows: *Poo badensis-Festucetum pallentis*—8 old relevés and 36 recent relevés, *Festuco pallentis-Caricetum humilis*—8 old relevés and 22 recent relevés, *Festuco valesiacae-Stipetum capillatae*—12 old relevés and 6 recent relevés, and *Polygalo majoris-Brachypodietum pinnati*—12 old relevés and 49 recent relevés.

The recent plot sizes corresponded to old plot sizes (Kaleta, 1965) and the plot size of 25 m^2 was used. We tried to sample some relevés in 2012–2018 at the same localities as old relevés. This could be done in the case of ~20 relevés, but we were not able to do this in the case of all relevés because of the absence of permanent plots in historical research. Since Devínska Kobyla is a relatively small area, the possibility of mistakes was low. Because of the missing permanent plots, we did more new relevés, which made the research more precise and this could also eliminated mistakes. The same process of plot selection as used in old time period was used to obtain comparable dataset. The relevés were sampled according to the actual occurrence and distribution of the communities with the intention to record all types of dry grasslands and cover entire area of reserve. The phytosociological research in both time periods was performed according to the methodology of the Zürich-Montpellier school (Braun-Blanquet, 1964). In 1964, the old Braun-Blanquet cover-abundance scale was used. In 2000–2018, the modified Braun-Blanquet cover-abundance scale was used, extended by categories 2m (cover 1–5%, abundance high), 2a (cover 5–12.5%), and 2b (cover 12.5–25%) values (Barkman et al., 1964).

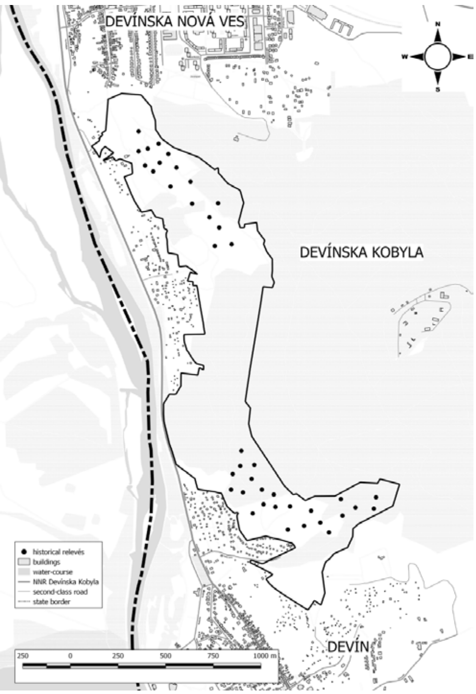

Fig. 1. Distribution of recorded relevés in the study area in 1964.

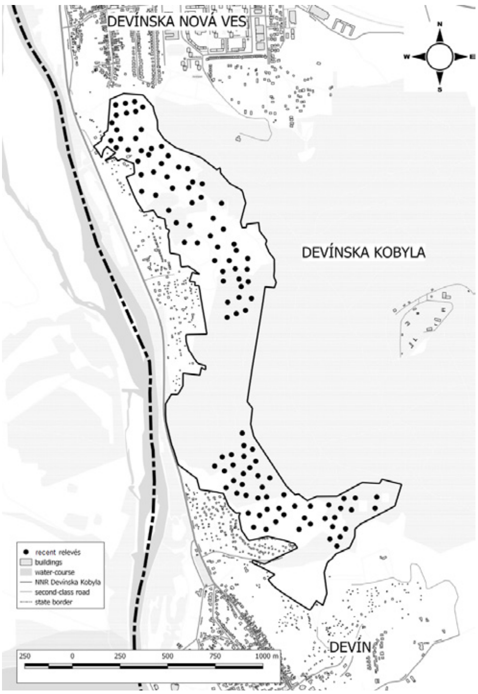

Fig. 2. Distribution of recorded relevés in the study area in the years 2012–2018.

Data analysis

Relevés were imported into a TURBOVEG database (Hennekens, Schaminée, 2001) and analyzed in the program JUICE (Tichý, 2002). Bryophytes, lichens, and taxa of vascular plants determined only to the genus level were excluded from the analysis. Taxa that occurred in more than one layer were merged. Some taxonomically problematic species were merged to higher or more broadly defined taxa: *Achillea millefolium* agg. (*A. collina* Becker ex Rchb., *A. millefolium* L., *A. pannonica* Scheele, and *A. setacea* Waldst. et Kit.), *Arenaria serpyllifolia* agg. (*A. leptoclados* (Rchb.) Guss. and *A. serpyllifolia* L.), *Erophila verna* agg. (*E. spathulata* Láng and *E. verna* (L.) Chevall), *Erysimum diffusum* agg. (*E. andrzejowskianum* Besser and *E. diffusum* Ehrh.), *Galium mollugo* agg. (*G. album* Mill. and *G. mollugo* L.), *Lotus corniculatus* agg. (*L. borbasii* Ujhelyi, *L. corniculatus* L. and *L. tenuis* Waldst. et Kit. ex Willd.), and *Poa pratensis* agg. (*P. angustifolia* L. and *P. pratensis* L.). Because different cover-abundance scales have been used in old and recent relevés, the values 2m, 2a, and 2b in both datasets were converted to value 2.

Phytosociological relevés were analyzed by numerical classification (cluster analysis) in the program SYN-TAX 2000 (Podani, 2001). Data were not transformed. The Group average method in combination with Ružička's coefficient proved to be the most effective linkage method and distance measure and the results obtained by them corresponded to our field experience. Clusters were assigned to associations by diagnostic species (Jarolímek, Šibík, 2008; Hegedüšová-Vantarová, Škodová, 2014) and by electronic expert system for identification of syntaxa of grassland vegetation (Janišová, 2007).

The species richness (number of species), the Shannon–Wiener index of diversity [*H*'] (Hill, 1973), the Simpson index (Hill, 1973), and the Pielou's measure of species evenness [$J = H'/\ln(S)$, where *H*' is the Shannon diversity index and *S* is the number of taxa per relevé] were calculated for each relevé in the program JUICE. Subsequently, the number of species and the values of indices were calculated for each association, both for reference and recent time periods. The normality of data distribution was tested by Shapiro–Wilk test. As the data were not normally distributed, the Kruskal–Wallis analysis in R 3.5.1 (R Core Team, 2018) was applied to test the differences in these values between the periods in each association and also in whole dataset.

As an explanatory variable to show possible changes in vegetation ecological characteristics, the Ellenberg indicator values for light, temperature, continentality, moisture, soil reaction, and nutrients (Ellenberg et al., 1992) based on species presence were calculated, using the program JUICE. We analyzed the Ellenberg indicator values in total old and recent datasets (all associations together) using the Kruskal–Wallis analysis in R 3.5.1 (R Core Team, 2018), because by testing the Ellenberg indicator values between the two time periods and associations differences were found not significant.

Nomenclature

The nomenclature of taxa follows Marhold (1998), and the nomenclature of syntaxa follows Hegedüšová-Vantarová, Škodová (2014).

Results

The hypothesis was confirmed in the case of some associations (Table 1). The Kruskal–Wallis analysis revealed a statistically significant lower values of the median of species richness in recent relevés from the association *Festuco pallentis-Caricetum humilis* ($p = 0.0002$, level of significance $\alpha = 0.05$, $\chi^2 = 13.77$) and from the association *Polygalo majoris-Brachypodietum pinnati* ($p = 0.0002$, $\alpha = 0.05$, $\chi^2 = 13.44$) compared to old relevés (Table 2 and Fig. 3).

The values of the median of Shannon–Wiener index of diversity were significantly lower over time in the association *Polygalo majoris-Brachypodietum pinnati* ($p = 0.001$, $\alpha = 0.05$, $\chi^2 = 11.10$) (Table 2 and Fig. 4).

The median values of Simpson index were significantly lower in the association *Polygalo majoris-Brachypodietum pinnati* ($p = 0.0001$, $\alpha = 0.05$, $\chi^2 = 14.68$) (Table 2 and Fig. 5).

The median values of Pielou's measure of species evenness were significantly lower in the associations *Poo badensis-Festucetum pallentis* ($p = 0.0039$, $\alpha = 0.05$, $\chi^2 = 8.35$) and *Polygalo majoris-Brachypodietum pinnati* ($p = 0.0047$, $\alpha = 0.05$, $\chi^2 = 7.98$) (Table 2 and Fig. 6).

T a b l e 1. Summarization of significancy of the results of analysis of differences in the biodiversity in the associations between the old and recent data.

	Poo badensis-Festucetum pallentis	***Festuco pallentis-Caricetum humilis***	***Festuco valesiacae-Stipetum capillatae***	***Polygalo majoris-Brachypodietum pinnati***
Species richness	No significant difference	Significantly lower	No significant difference	Significantly lower
Shannon–Wiener index of diversity	Significantly higher	No significant difference	No significant difference	Significantly lower
Simpson index	No significant difference	No significant difference	No significant difference	Significantly lower
Pielou‘s measure of species evenness	Significantly lower	No significant difference	No significant difference	Significantly lower

T a b l e 2. Results of Kruskal–Wallis analysis testing differences in the species richness and diversity indices in the communities between the old and recent data.

Association	**χ^2**	**p Value**	**Significancy**
Species richness			
Poo badensis-Festucetum pallentis	0.81	0.37	NO
Festuco pallentis-Caricetum humilis	13.77	**0.0002**	**YES**
Festuco valesiacae-Stipetum capillatae	0.05	0.82	NO
Polygalo majoris-Brachypodietum pinnati	13.44	**0.0002**	**YES**
Shannon–Wiener index of diversity [*H*‘]			
Poo badensis-Festucetum pallentis	7.38	**0.01**	**YES**
Festuco pallentis-Caricetum humilis	1.01	0.32	NO
Festuco valesiacae-Stipetum capillatae	0.001	0.98	NO
Polygalo majoris-Brachypodietum pinnati	11.10	**0.001**	**YES**
Simpson index			
Poo badensis-Festucetum pallentis	0.03	0.87	NO
Festuco pallentis-Caricetum humilis	0.70	0.40	NO
Festuco valesiacae-Stipetum capillatae	0.08	0.78	NO
Polygalo majoris-Brachypodietum pinnati	14.68	**0.0001**	**YES**
Pielou‘s measure of species evenness			
Poo badensis-Festucetum pallentis	8.35	**0.004**	**YES**
Festuco pallentis-Caricetum humilis	0.04	0.83	NO
Festuco valesiacae-Stipetum capillatae	0.01	0.99	NO
Polygalo majoris-Brachypodietum pinnati	7.98	**0.005**	**YES**

The median values of Shannon–Wiener index of diversity were higher in the association *Poo badensis-Festucetum pallentis* ($p = 0.01$, $\alpha = 0.05$, $\chi^2 = 7.38$) (Table 2 and Fig. 4).

The differences in values in the median of species richness and indices in other associations were not statistically significant (Tables 1, 2).

By Kruskal–Wallis analysis of differences in Ellenberg indicator values between old and recent relevés of the studied associations, no statistically significant results were obtained

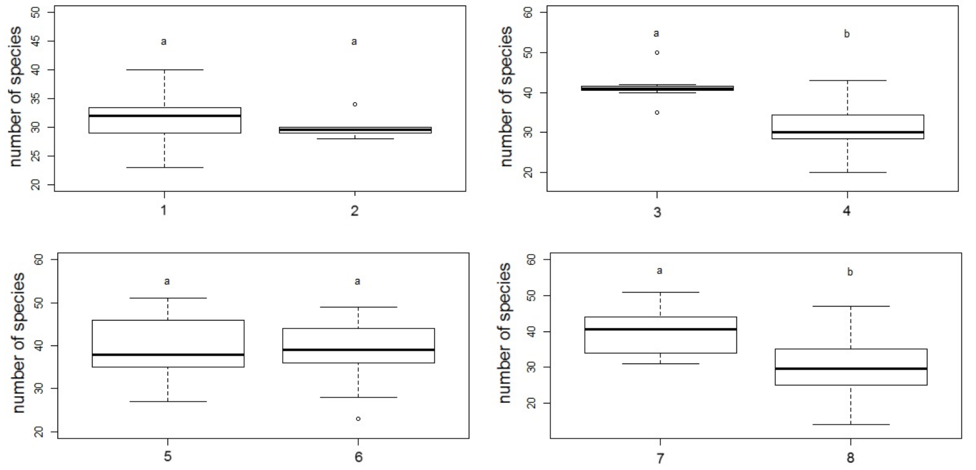

Fig. 3. The species richness of the old and recent relevés of studied vegetation types of Devínska Kobyla. Notes: - ☐median; 0 25 75%; I min-max; o outliers; a b – significant difference; a a – not significant difference;
1 – *Poo badensis-Festucetum pallentis*, reference relevés—1964 (Kaleta, 1965);
2 – *Poo badensis-Festucetum pallentis*, recent relevés—2000–2018 (Miškovic, 2018);
3 – *Festuco pallentis-Caricetum humilis*, reference relevés—1964 (Kaleta, 1965);
4 – *Festuco pallentis-Caricetum humilis*, recent relevés—2000–2018 (Miškovic, 2018);
5 – *Festuco valesiacae-Stipetum capillatae*, reference relevés—1964 (Kaleta, 1965);
6 – *Festuco valesiacae-Stipetum capillatae*, recent relevés—2000–2018 (Miškovic, 2018);
7 – *Polygalo majoris-Brachypodietum pinnati*, reference relevés—1964 (Kaleta, 1965);
8 – *Polygalo majoris-Brachypodietum pinnati*, recent relevés—2000–2018 (Miškovic, 2018).

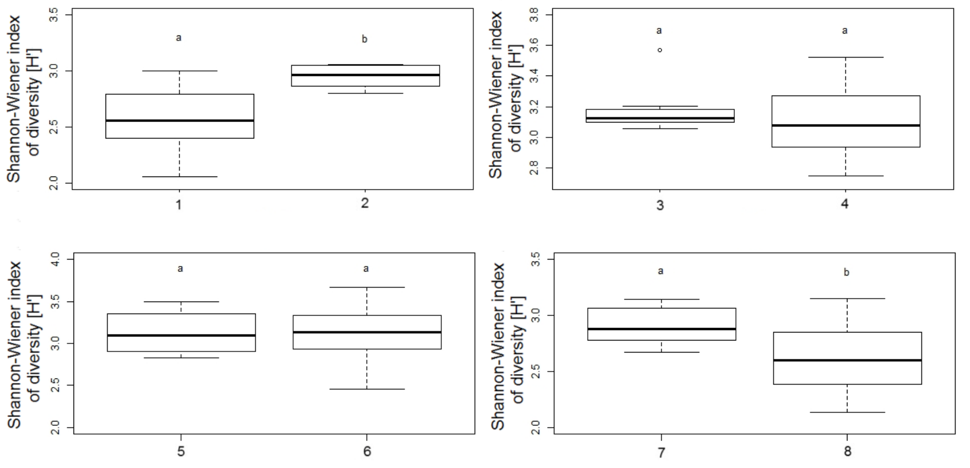

Fig. 4. The Shannon–Wiener index of diversity [*H*'] for the old and recent relevés of studied vegetation types of Devínska Kobyla. Explanation corresponds to those in Figure 3.

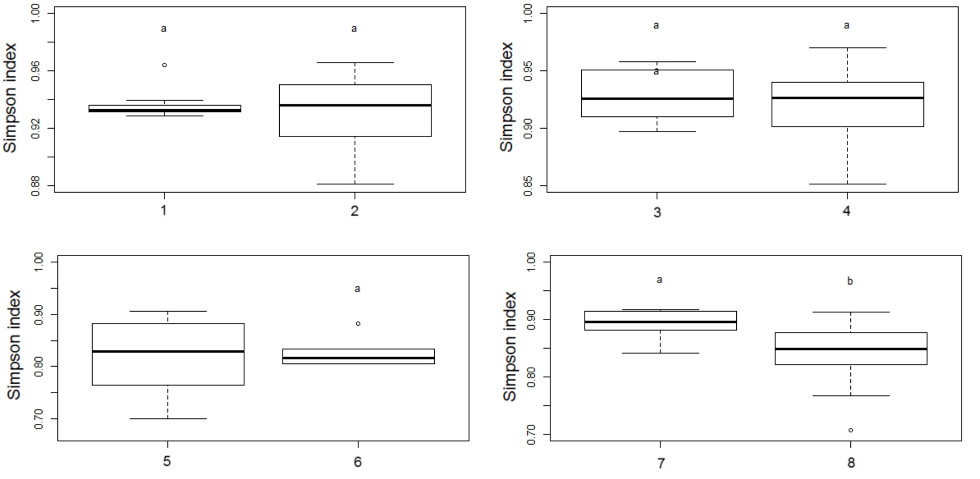

Fig. 5. The Simpson index for the old and recent relevés of studied vegetation types of Devínska Kobyla. Explanation corresponds to those in Figure 3.

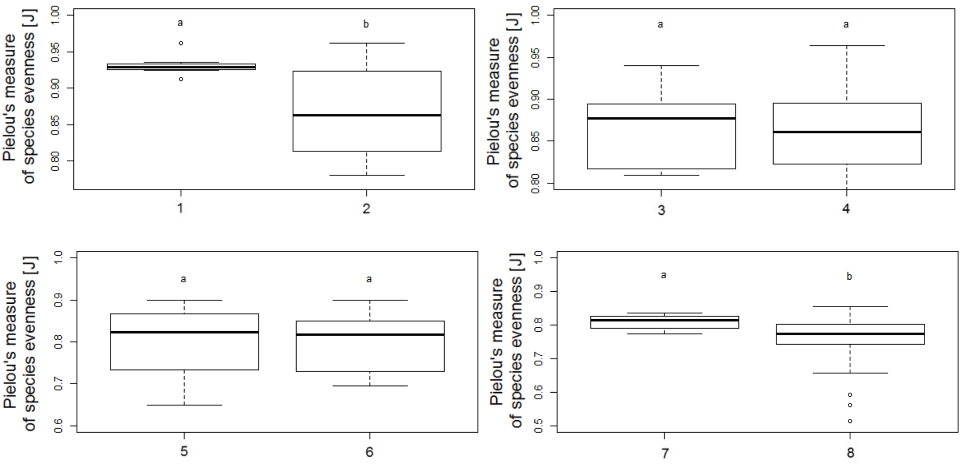

Fig. 6. The Pielou's measure of species evenness for the old and recent relevés of studied vegetation types of Devínska Kobyla. Explanation corresponds to those in Figure 3.

(Table 3). The analysis of the Ellenberg indicator values in total dry grassland vegetation by Kruskal–Wallis analysis revealed a statistically significant lower Ellenberg values for light in recent period compared to the past ($p = 0.004$, $\alpha = 0.05$, $\chi2 = 8.5$) (Table 3 and Fig. 7a). There was no significant difference in Ellenberg indicator values for temperature (Fig. 7b), continentality (Fig. 7c). The analysis revealed a significant higher Ellenberg indicator values for moisture (p = 0.01, α = 0.05, χ2 =6.2) (Table 3 and Fig. 7d). There was no significant difference in Ellenberg indicator values for soil reaction (Fig. 7e) between past and recent period. There were a significant higher Ellenberg indicator values for nutrients (p = 0.0001, α = 0.05,

T a b l e 3. Results of Kruskal–Wallis analysis testing differences in the Ellenberg indicator values between the old and recent data.

	χ^2	***p* Value**	**Significancy**
Light	8.45	**0.004**	**YES**
Temperature	0.99	0.32	NO
Continentality	1.49	0.22	NO
Moisture	6.2	**0.01**	**YES**
Soil reaction	1.59	0.21	NO
Nutrients	15.8	**0.0001**	**YES**

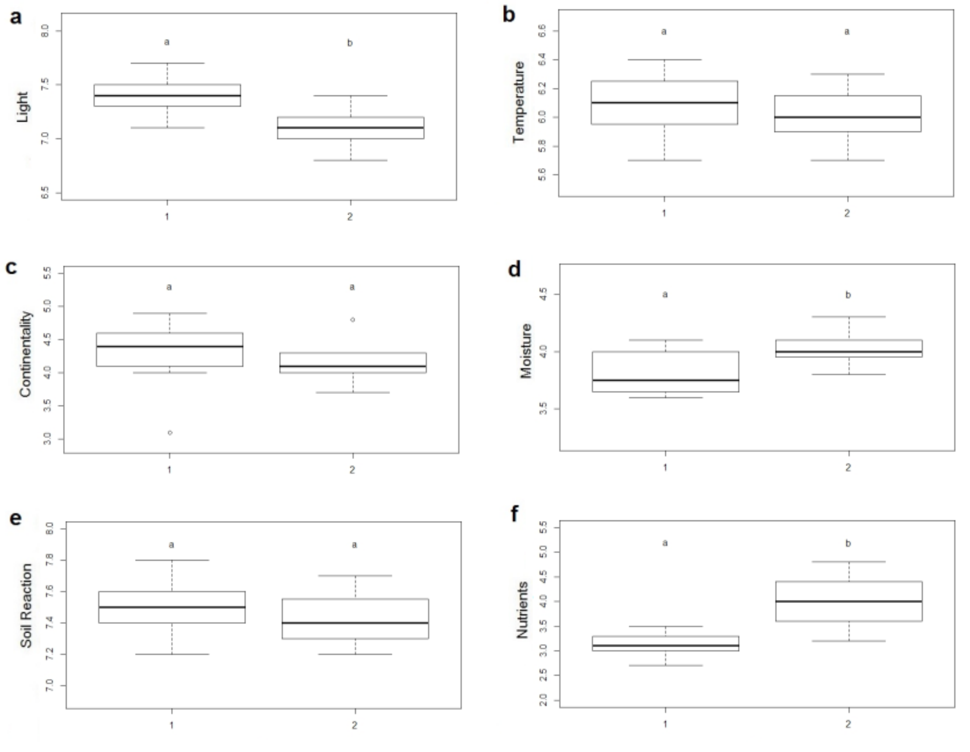

Fig. 7. The Ellenberg indicator values for light (a), temperature (b), continentality (c), moisture (d), soil reaction (e), and nutrients (f) for the reference (year 1964) and recent (years 2000–2018) relevés.
Notes: 1 – reference relevés (1964); 2 – recent relevés (2000–2018); a b –significant difference; a a –not significant difference; - median; 0 25 75%; I min-max; o outliers.

T a b l e 4. Frequency of occurrence and cover range of woody species in recent relevés (2000–2018), which were not recorded in the past (1964).

	Pb-Fp	*Pm-Bp*
Cornus mas	–	10^{1}
Fraxinus ornus	–	6^{+-2}
Pinus sylvestris	6^{+}	–
Populus alba	11^{2-3}	–
Populus nigra	11^{+-2}	–
Robinia pseudoacacia	3^{2}	–
Swida sanguinea	8^{1-2}	4^{+}
Ulmus minor	–	4^{+}

Notes: The numbers given in the table are frequency values of a species for a particular vegetation type; their upper indices represent the cover range of species. *Pb-Fp—Poo badensis-Festucetum pallentis* and *Pm-Bp—Polygalo majoris-Brachypodietum pinnati.*

$\chi2$ =15.8) in recent period compared to the past (Table 3 and Fig. 7f).

As an explanatory feature we present the table with the values of frequency of occurrence and cover range of woody species in recent relevés (2000–2018), which were not recorded in the past (1964) (Table 4). The numbers given in the table are frequency values of a species for a particular vegetation type and their upper indices represent the cover range of species. The frequency of occurence of some woody species increased in associations *Poo badensis-Festucetum pallentis* and *Polygalo majoris-Brachypodietum pinnati* (Table 4).

Discussion

The majority of significant results of analyses revealed the lower values of species richness, diversity, and evenness in studied dry grassland communities of Devínska Kobyla in recent period compared to the past (Tables 1,d 2 and Figs 3–6). This decline of values of diversity indices is probably caused mainly by the abandonment of traditional land use (grazing and mowing) in the area after 1965, because the disturbances caused by grazing and mowing are necessary for the maintenance of the species richness and diversity of dry grasslands (Kubíková, 1999; Münzbergová, 2001; Chytrý, 2007; Galvánek, Lepš, 2008).

The significant higher values in recent period were recorded only in the case of Shannon–Wiener in one association—*Poo badensis-Festucetum pallentis* (Table 2 and Fig. 4). Hegedüšová, Senko (2011) in study from Devínska Kobyla found out that species richness in this association declined over time, which is in the contrast with our result of increase in Shannon–Wiener index. Hegedüšová, Senko (2011) recorded the increase of species richness in the other association—*Festuco valesiacae-Stipetum capillatae*. Authors consider *Festuco valesiacae-Stipetum capillatae* the least endangered by successional changes.

We did not record significant differences of any diversity index in the association *Festuco valesiacae-Stipetum capillatae* between old and recent data. The possible reason can be that the association is species rich and grows on relatively steeper soils, where succession is slower than in deeper soils, where grows the association *Polygalo majoris-Brachypodietum pinnati*, which showed decrease of all diversity indices.

The explanation of the higher diversity index in the association *Poo badensis-Festucetum pallentis* in recent period can be that this association occurs on rocks with the shallowest soils, where significant changes in management did not occur. There were no grazing or mowing in the rocky shallow soil in the past. That could be the reason why this association is less endangered by successional changes after abandonment of traditional management than other analyzed associations, which grow on sites with deeper soil, where management changes occurred. On the other hand, it is still a fact that some changes occurred also in the association *Poo badensis-Festucetum pallentis*, for example, overgrowing by woody species (Table 4).

Fig. 8. Overgrowing by woody species in the stands of the association *Poo badensis-Festucetum pallentis* at Sandberg, Devínska Kobyla (Author of photo: Ján Miškovic).

Although no significant differences of the Ellenberg indicator values between old and recent period were found for some of the environmental variables, the Ellenberg indicator values for nutrients and moisture were significantly higher in recent period (Fig. 7). This result could have connection with overgrowing by woody species and increased accumulation of died biomass after abandonment and its increased decomposition, and thus enrichments of soils by nutrients. Similar results were achieved by Ruprecht et al. (2010), and the accumulation of biomass is a consequence of abandonment of grasslands. After the abandonment of the traditional management, the gradual overgrowing by woody species took place in the Devínska Kobyla (Hegedüšová, Senko, 2011). The frequency of occurrence of some woody species was higher in the recent relevés from the associations *Poo badensis-Festucetum pallentis* and *Polygalo majoris-Brachypodietum pinnati* (Table 4). The overgrowing by species such as *Populus alba*, *P. nigra*, and invasive species *Robinia pseudoacacia* in the stands of the *Poo badensis-Festucetum pallentis* association in Devínska Kobyla is also evident in the photography from the studied area (Fig. 8). The stands of *Poo badensis-Festucetum pallentis* are open (average total cover of species in relevé is 40%) because of extreme environmental conditions (shallow dry soils and steep slopes) of places where the stands grow. The open vegetation is more prone to introduction of new species such as *Populus alba* and *P. nigra*. The stands of the association *Polygalo majoris-Brachypodietum pinnati*, on the other hand, grow on deeper soils. From a long-term perspective, these conditions fit to some woody species. Also other authors found the overgrowing by woody species as one of the main sign of degradation of dry grasslands (Dostálek, Frantík, 2008; Schrautzer et al., 2009; Hegedüšová, Senko, 2011) or habitat alteration (Smit, 2004).

Our study confirmed the trend of decline in biodiversity of species-rich dry calcareous grasslands after abandonment of traditional management during recent decades shown in other studies from Central Europe (Galvánek, Lepš, 2008; Schrautzer et al., 2009; Hegedüšová, Senko, 2011; Pipenbaher et al., 2013). From numerous studies, it is clear that the dry calcareous grasslands should receive attention in regard to conservation management (WallisDeVries, 1999; Barbaro et al., 2001; Münzbergová, 2001; Willems, 2001; Kahmen et al., 2002; Riecken et al., 2002; WallisDeVries et al., 2002; Bąba, 2004; Masé, 2005; Bornkamm, 2006; Stadler et al., 2007; Dostálek, Frantík, 2008; Peter et al., 2009; Hegedüšová, Senko 2011; Kuzemko et al., 2016). Our study showed similar conclusions as these studies. As the diversity of dry grasslands in the Devínska Kobyla NNR decreased after the grazing cessation, restoring the grazing should be the part of the conservation management in the area. Grazing is recognized as one of the most natural methods of management of dry grasslands (Hadar et al., 1999; Peter et al., 2009; Hegedüšová, Senko, 2011).

In the last few years, the traditional management has begun to be partly restored in Devínska Kobyla NNR. The biomass of *Robinia pseudoacacia* L. has been being removed from the year 2016, and the goat grazing was reintroduced in 2017 again. In the recent years, the goats have been grazed also on the steep slopes, where they promote heightened erosion with potentially negative impact on the dry grassland communities. The best possible way is to combine goat and sheep co-grazing on the moderate slopes with removal of the biomass of woody species on the steep slopes. Also other authors (Barbaro et al., 2001; Hegedüšová, Senko, 2011; Bojkovski et al., 2014) found the combination of shrub clearing and grazing as one of the best ways of conservation management of dry grasslands.

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References

Alard, D., Chabrerie, O., Dutoit, T., Roche, P. & Langlois E. (2005). Patterns of secondary succesion in calcareous grasslands: can we distinguish the influence of former land uses from present vegetation data?. *Basic and Applied Ecology*, 6, 161–173. DOI: 10.1016/j.baae.2005.01.010.

Bąba, W. (2003). Changes in the structure and floristic composition of the limestone grasslands after cutting trees and shrubs and mowing. *Acta Soc. Bot. Pol.*, 72(1), 61–69. DOI: 10.5586/asbp.2003.008.

Bąba, W. (2004). The species composition and dynamics in well-preserved and restored calcareous xerothermic grassland (South Poland). *Biologia*, 59, 447–456.

Baláž, D. (1994). The succession of xerothermic communities at Devínska Kobyla and their practical protection (in Slovak). *Daphne*, 2, 6–7.

Barbaro, L., Dutoit, T. & Cozic P. (2001). A six-year experimental restoration of biodiversity by shrub-clearing and grazing in calcareous grassland of the French Prealps. *Biodivers. Conserv.*, 10, 119–135. DOI: 10.1023/A:1016629507196.

Barkman, J.J., Doin, H. & Segal S. (1964). Kritische Bemerkungen und Vorschläge zur quantitativen Vegetationsanalyse. *Acta Botanica Neerlandica*, 13(3), 394–419. DOI: 10.1111/j.1438-8677.1964.tb00164.x.

Bobbink, R., During, H., Schreurs, J., Willems, J. & Zielman R. (1987). Effects of selective clipping and mowing time on species diversity in chalk grassland. *Folia Geobotanica et Phytotaxonomica*, 22, 363–376. DOI: 10.1007/BF02853233.

Bojkovski, D., Štuhec, I., Kompan, D. & Zupan M. (2014). The behavior of sheep and goats co-grazing on pasture with different types of vegetation in the karst region. *J. Anim. Sci.*, 92(6), 2752–2758. DOI: 10.2527/jas.2013-7199.

Bornkamm, R. (2006). Fifty years vegetation development of a xerothermic calcareous grassland in Central Europe after heavy disturbance. *Flora*, 201(4), 249–267. DOI: 10.1016/j.flora.2005.06.012.

Braun-Blanquet, J. (1964). *Pflanzensoziologie. Grundzüge der vegetationskunde*. Wien, New York: Springer Verlag.

Chytrý, M. (Ed.) (2007). *Vegetation of the Czech Republic 1. Grassland and heathland vegetation* (in Czech). Praha: Academia.

Domin, K. (1931). *Brachypodietum* at Kobyla near Bratislava (in Czech). *Věda Přírodní*, 12, 57–61.

Dostálek, J. & Frantík T. (2008). Dry grassland plant diversity conservation using low-intensity sheep and goat grazing management: Case study in Prague (Czech Republic). *Biodivers. Conserv.*, 17, 1439–1454. DOI: 10.1007/s10531-008-9352-1.

During, H.J. & Willems J.H. (1984). Diversity models applied to a chalk grassland. *Vegetatio*, 57, 103–114. DOI: 10.1007/BF00047305.

Dzwonko, Z. & Loster S. (1998). Dynamics of species richness and composition in a limestone grassland restored after tree cutting. *J. Veg. Sci.*, 9, 387–394. DOI: 10.2307/3237103.

Ellenberg, H., Weber, H.E., Düll, R., Wirth, W., Werner, W. & Paulissen D. (1992). Zeigerwerte von Pflanzen in Mitteleuropa Ed. 2. *Scripta Geobotanica*, 18, 1–258.

Feráková, V. & Kocianová E. (Eds.) (1997). *Flora, geology and paleontology of Devínska Kobyla* (in Slovak). Bratislava: Litera.

Feráková, V. & Jarolímek I. (2011). Bratislava. In J.K. Kelcey & N. Müller (Eds.), *Plants and habitats of European cities* (pp. 79–129). New York: Springer.

Galvánek, D. & Lepš J. (2008). Changes of species richness pattern in mountain grasslands: abandonment versus restoration. *Biodivers. Conserv.*, 17, 3241–3253. DOI: 10.1007/s10531-008-9424-2.

Hadar, L., Noy-Meir, I. & Perevolotsky A. (1999). The effect of shrub clearing and grazing on the composition of a Mediterranean plant community: functional groups versus species. *J. Veg. Sci.*, 10, 673–682. DOI: 10.2307/3237082.

Hajdúk, J. (1986). Results of vegetation changes research on permanent trial plots and their importance for the management of the State Nature Reserve Devínska Kobyla. *Ochrana Prírody*, 7, 79–105.

Hajdúk, J. (1997). Experimental research and vegetation records on permanent plots at Devínska Kobyla (in Slovak). In V. Feráková & E. Kocianová (Eds.), *Flora, geology and paleontology of Devínska Kobyla* (pp. 155–157). Bratislava: Litera.

Hansson, M. & Fogelfors H. (2000). Management of a seminatural grassland; results from a 15-year-old experiment in southern Sweden. *J. Veg. Sci.*, 11, 31–38. DOI: 10.2307/3236772.

Hegedüšová, K. & Senko D. (2011). Successional changes of dry grasslands in southwestern Slovakia after 46 years of abandonment. *Plant. Biosyst.*, 145(3), 666–687. DOI: 10.1080/11263504.2011.601605.

Hegedüšová, K. & Valachovič M. (2015). Vegetation pattern on the western slope of Devínska Kobyla Mts – Southern Slovakia. *Phyton*, 55(1), 41–68. DOI: 10.12905/0380.phyton55(1)2015-0041.

Hegedüšová-Vantarová, K. & Škodová I. (Eds.) (2014). *Plant communities of Slovakia 5. Grassland vegetation* (in Slovak). Bratislava: Veda, vydavateľstvo SAV.

Hennekens, S.M. & Schaminée J.H.J. (2001). TURBOVEG, a comprehensive data base management system for vegetation data. *J. Veg. Sci.*, 12(4), 589–591. DOI: 10.2307/3237010.

Hill, M.O. (1973). Diversity and evenness: a unifying notation and its consequences. *Ecology*, 54, 427–432. DOI: 10.2307/1934352.

Hyžný, M., Hudáčková, N., Biskupič, R., Rybár, S., Fuksi, T., Halásová, E., Zágoršek, K., Jamrich, M. & Ledvák P. (2012). Devínska Kobyla – a window into the Middle Miocene shallow-water marine environments of the Central Paratethys (Vienna Basin, Slovakia). *Acta Geologica Slovaca*, 4(2), 95–111.

Jacquemyn, H., Van Mechelen, C., Brys, R. & Honnay O. (2010). Management effects on the vegetation and soil seed bank of calcareous grasslands: An 11-year experiment. *Biol. Conserv.*, 144, 416–422. DOI: 10.1016/j.biocon.2010.09.020.

Janišová, M. (Ed.) 2007. *Grassland vegetation of Slovak Republic – electronic expert system for identification of syntaxa*. Bratislava: Institute of Botany SAS.

Jarolímek, I. & Šibík J. (Eds.) (2008). *Diagnostic, constant and dominant species of the higher vegetation units of Slovakia*. Bratislava: Veda, vydavateľstvo SAV.

Kahmen, S., Poschlod, P. & Schreiber K.F. (2002). Conservation management of calcareous grasslands. Changes in

plant species composition and response of functional traits during 25 years. *Biol. Conserv.*, 104, 319–328. DOI: 10.1016/S0006-3207(01)00197-5.
Kaleta, M. (1965). *Vegetation terms of Devínska Kobyla* (in Slovak). Master's Thesis, Faculty of Natural Sciences, Comenius University, Bratislava.
Köhler, B., Gigon, A., Edwards, P.J., Krüsi, B., Langenauer, R., Lüscher, A. & Ryser P. (2005). Changes in the species composition and conservation value of limestone grasslands in Northern Switzerland after 22 years of contrasting managements. *Perspect Plant Ecol. Evol. Syst.*, 7, 51–67. DOI: 10.1016/j.ppees.2004.11.003.
Kubíková, J. (1999). Xerothermic lawns and semi-xerothermous forests (in Czech). In V. Petříček (Ed.), *Péče o chráněná* území *I* (pp. 213–236). Praha: AOPaK ČR.
Kuzemko, A.A., Steinbauer, M.J., Becker, T., Didukh, Y.P., Dolnik, C., Jeschke, M., Naqinezhad, A., Ugurlu, E., Vassilev, K. & Dengler J. (2016). Patterns and drivers of phytodiversity in steppe grasslands of Central Podolia (Ukraine). *Biodivers. Conserv.*, 25, 2233–2250. DOI: 10.1007/s10531-016-1060-7.
Maliníková, E. (2003). *Xerothermic and sub-xerothermic vegetation of Devínska Kobyla* (in Slovak). Master's Thesis, Faculty of Natural Sciences, Comenius University, Bratislava.
Marhold, K. (1998). Ferns and flowering plants. In K. Marhold & F. Hindák (Eds.), *Checklist of non-vascular and vascular plants of Slovakia* (pp. 333–687). Bratislava: Veda, vydavateľstvo SAV.
Masé, G. (2005). The management of dry grassland in Switzerland. A Swiss federal program and its local practical application. *Biotechnol. Agron. Soc. Environ.*, 9(2), 133–138.
Michalko, M. (1977). *Forest communities of Devínska Kobyla* (in Slovak). Master's Thesis, Faculty of Natural Sciences, Comenius University, Bratislava.
Miškovic, J. & Dúbravcová Z. (2004a). The community *Poo badensis-Festucetum pallentis* Klika 1931 corr. Zólyomi 1966 at Devínska Kobyla over 36 years (in Slovak). *Bulletin Slovenskej Botanickej Spoločnosti,* 26(Suppl. 1), 160–165.
Miškovic, J. & Dúbravcová Z. (2004b). Changes in the community *Ranunculo illyrici-Festucetum valesiacae* Klika 1931 at Devínska Kobyla over 36 years (in Slovak). *Bulletin Slovenskej Botanickej Spoločnosti*, 26, 185–192.
Miškovic, J. (2018). *Dynamic changes of grasslands of Devínska Kobyla hill* (in Slovak). Dissertation, Faculty of Natural Sciences, Comenius University, Bratislava.
Mojses, M. & Petrovič F. (2013). Land use changes of historical structures in the agricultural landscape at the local level - Hriňová case study. *Ekológia (Bratislava)*, 32(1), 1–12. DOI : 10.2478/eko-2013-0001.
Münzbergová, Z. (2001). Recovery of species-rich xerothermic lawns on the example of the Stráně u splavu and Stráně u Chroustova reservations (in Czech). *Příroda*, 19, 101–121.
Palaj, A. & Kollár J. (2018). Changes in alpine vegetation over 50 years in the Western Tatras (Slovakia). *Ekológia (Bratislava)*, 37(2), 122–133. DOI: 10.2478/eko-2018-0012.
Peter, M., Gigon, A., Edwards, P.J. & Lüscher A. (2009). Changes over three decades in the floristic composition of nutrient-poor grasslands in the Swiss Alps. *Biodivers. Conserv*., 18, 547–567. DOI: 10.1007/s10531-008-9521-2.
Pipenbaher, N., Kaligarič, M., Mason, N.W.H. & Škornik S. (2013). Dry calcareous grasslands from two neighboring biogeographic regions: relationship between plant traits and rarity. *Biodivers. Conserv.*, 22, 2207–2221. DOI: 10.1007/s10531-013-0520-6.
Podani, J. (2001). *SYN-TAX 2000, Computer program for data analysis in ecology and systematics for Windows 95, 98 & NT, User's manual*. Budapest: Scientia Publishers.
Poschlod, P., Kiefer, S., Tränkle, U., Fischer, S. & Bonn S. (1998). Plant species richness in calcareous grasslands affected by dispersability in space and time. *Appl. Veg. Sci.*, 1, 75–90. DOI: 10.2307/1479087.
Poschlod, P. & WallisDeVries M.F. (2002). The historical and socioeconomic perspective of calcareous grasslands. Lessons learnt from the distant and recent past. *Biol. Conserv.*, 104, 361–376. DOI: 10.1016/S0006-3207(01)00201-4
R Core Team (2018). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing, Vienna, Austria. Accessed October 22, 2019, from https://www.R-project.org/.
Riecken, U., Finck, P. & Schröder E. (2002). Significance of pasture landscape for nature conservation and extensive agriculture. In B. Redecker, P. Finck, W. Härdtle, U. Riecken & E. Schröder (Eds.), *Pasture landscape and nature conservation* (pp. 423–435). Heidelberg, Berlin: Springer. DOI: 10.1007/978-3-642-55953-2.
Ruprecht, E., Enyedi, M.Z., Eckstein, R.L. & Donath T.W. (2010). Restorative removal of plant litter and vegetation 40 years after abandonment enhances re-emergence of steppe grassland vegetation. *Biol. Conserv.*, 143, 449–456. DOI: 10.1016/j.biocon.2009.11.012.
Schrautzer, J., Jansen, D., Breurer, M. & Nelle O. (2009). Succession and management of calcareous dry grasslands in the Northern Franconian Jura, Germany. *Tuexenia*, 29, 339–351.

Senko, D., Miškovic, J., Gallay, J,. Senková, M. & Baldaufová K. (2008). Dynamics of the changes of vegetation at Devínska Kobyla and its prediction (in Slovak). *Geografický Časopis*, 60, 319–338.

Smit, G.N. (2004). An approach to tree thinning to structure southern African savannas for long-term restoration from bush encroachment. *J. Environ. Manag.*, 71, 179–191. DOI: 10.1016/j.jenvman.2004.02.005.

Stadler, J., Trefflich, A., Brandl, R. & Klotz S. (2007). Spontaneous regeneration of dry grasslands on set-aside fields. *Biodivers. Conserv.*, 16, 621–630. DOI: 10.1007/s10531-005-0604-z.

Špulerová, J., Drábová, M. & Lieskovský J. (2016): Traditional agricultural landscape and their management in less favoured areas in Slovakia. *Ekológia (Bratislava)*, 35(1), 1–12. DOI: 10.1515/eko-2016-0001.

Tälle, M., Deák, B., Poschlod, P., Valkó, O., Westerberg, L. & Milberg P. (2018). Similar effects of different mowing frequencies on the conservation value of semi-natural grasslands in Europe. *Biodivers. Conserv.*, 27, 2451–2475. DOI: 10.1007/s10531-018-1562-6.

Tichý, L. (2002). JUICE, software for vegetation classification. *J. Veg. Sci.*, 13, 451–453. DOI: 10.1111/j.1654-1103.2002.tb02069.x.

Török, K. & Szitár K. (2010). Long-term changes of rock grassland communities in Hungary. *Community Ecol.*, 11, 68–76. DOI: 10.1556/ComEc.11.2010.1.10

WallisDeVries, M.F. (1999). The dilemma facing nature conservation and the role of large herbivores. In B. Gerken & M. Görner (Eds.), *The development of European landscapes with large herbivores – history, models and perspectives* (pp. 24–31). Natur und Kulturlandschaft 3.

WallisDeVries, M.F., Poschlod, P. & Willems J.H. (2002). Challenges for the conservation of calcareous grasslands in northwestern Europe: integrating the requirements of flora and fauna. *Biol. Conserv.*, 104, 265–273. DOI: 10.1016/S0006-3207(01)00191-4.

Willems, J.H. (1983). Species composition and above ground phytomass in chalk grassland with different management. *Vegetatio*, 52, 171–180. DOI: 10.1007/BF00044994.

Willems, J.H. (2001). Problems, approaches, and results in restoration of Dutch calcareous grassland during the last 30 years. *Restor. Ecol.*, 9, 147–154. DOI: 10.1046/j.1526-100x.2001.009002147.x.

Wolkinger, F. & Plank S. (1981). *Dry grasslands of Europe*. Strasbourg: Nature and Environment Series.

Zlínska, J. (2004). *Adonido-Brachypodietum pinnati* (Libbert 1933) Krausch 1961 at Devínska Kobyla (in Slovak). *Biosozologia*, 2, 49–61.

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THE INFLUENCE OF THE SLOPE EXPOSURE ON THE SOIL AGGREGATION AND STRUCTURE, WATER STABILITY OF AGGREGATES, AND ECOLOGICAL MICROSTRUCTURE FORMATION OF THE RAVINE FOREST SOILS IN PRE-DNIPRO REGION (UKRAINE)

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Abstract

Bozhko K., Bilova N.: The influence of the slope exposure on the soil aggregation and structure, water stability of aggregates, and ecological microstructure formation of the ravine forest soils in Pre-Dnipro region (Ukraine). Ekológia (Bratislava), Vol. 39, No. 2, p. 116–129, 2020.

The soil aggregation and structure, water stability of aggregates, and peculiarities of microstructure formation of the ravine forest soils in Dnipropetrovsk region on the example of the northern variant of the ravine forest "Kapitanivskiy" have been identified. The soil properties of southern and northern ravine exposures have been compared. The soil structure, aggregate composition, water stability of aggregates as well as soil-forming processes of the ravine ecosystem have been analyzed. Micromorphological studies have shown a high degree of aggregation of the upper (0–60 cm) horizons of the soil profile. The structure-forming process is of a zoogenic origin. Aggregates of coprolite nature contain well-disintegrated plant remains. Dark gray, almost black color along the entire area of the micromorphological slide is due to a large amount of organic compounds, which indicates active processes of humification. Fine-dispersed humus consists of a large number of evenly spaced humus clusters. The type of humus is mull. The skeleton consists of minerals of various sizes, dominated by quartz and feldspars. Plasma is humus-clay, homogeneous throughout the entire slide, anisotropic with speckled glowing. Microstructure is mainly aggregated and, in some places, spongy, depending on a microzone of the soil slide. Elemental microstructure is of plasma-silty type. The area of the visible surface of the pores in the upper horizons of the soil profile is fairly large (40%). Pores are round and elongated, of regular shape, here and there with remains of small invertebrates. The deeper the soil slide is, the smaller the area of visible pores along with aggregation becomes. Correlating with micromorphological characteristics, water resistance of structural aggregates reaches very high (90.01% ± 3.07) values in the upper horizons of the soil slide, decreasing at depths. The coefficient of pedality is rather high (7.83 ± 0.81) in the upper horizons, decreasing at depths.

Key words: ravine forest, soil, structure, aggregation, water resistance, micromorphology.

Introduction

Forests are of paramount importance for improvement of the ecological situation of Ukraine. Multifunctional properties of forests contribute to a significant increase in soil fertility by means of transforming the surface water flow into the deep one. Preventing from the destructive influence of dry winds, forests hinder soil erosion (Bilova, 1997). The conclusions of the Spanish and French scientists, who regard phytostabilization as the most successful solution to the chemical, toxicological, and environmental problems related to soil contamination (Epelde et al., 2014; Nsanganwimana, 2014), coincide with our ideas.

In the ecological network of Ukrainian forests, ravine forests are of great importance. Throughout the steppe area, there are a number of locations of a significant topographic low (ravines) covered by wild ravine forests, which are situated in their natural environment. The absence of molehill, as opposed to steppe biogeocoenosis, the presence of the traces of old roots of dead trees, the well-formed specific sedentary-illuvial soil profile and others testify the early occurrence of ravine forests in the steppe (Bilova, 1997; Bilova, Travleev, 1999).

Ravine forests are of great scientific value to study the peculiarities of the formation of natural forests, where rare and endangered species of plants and animals have found shelter. In addition, ravine forests can set an example for growing anti-erosion plants as well as become a treasury of the seeds of wood and shrub breeds.

Soil is the main, resultant forest ecosystem unit. It is impossible to maintain and restore ravine forests without a thorough study of soil characteristics.

The properties of soils are studied by scientists from all over the world. Spanish scientists research the influence of the abiotic factors (temperature, soil moisture, and ultraviolet irradiance ratio) on the soil ecosystem (Morgado et al., 2015), the impact of long-term use of chemical and organic fertilizers on a large number of nitrogen-containing microbial communities of soil (Sun et al., 2015). Brazilian scientists investigate the role of earthworms in soil formation (Zúñiga et al., 2013), the effect of worms on quality characteristics of soils (Bartz et al., 2013). Iranian scientists conduct researches connected with the influence of the climate humidity grade during soil-forming processes of loess soils (Khormali et al., 2012), micromorphological aspects of the development of forest soils evolved from eruptive rocks in Lahidzhany (Ramezanpour, Pormasoumi, 2012) influence of technogenic disturbances on understory of oak forests (Eshaghi et al., 2017). German scientists investigate the relationship between macropores of the soil and its hydrological properties (Bogner et al., 2014). Slovak scientists carry out a number of substantial studies connected with the characteristics of physical properties of soil profiles under introduced trees (Polláková et al., 2017), development of a soil water regime (Tužinský et al., 2017), determination of Organic Fractions and Enzymatic Activity in Forest Spruce Soil of Tatra National Park (Gáfriková et al., 2018), dependence of the soil reaction on the number of mites (Buza, Divos, 2016), fluctuations of nutrients in the upper soil horizons under the bedding of different wood species (Polláková et al., 2015), assessment of the organic substance of soils from different ecosystems in relation to the carbon parameters (Tobiašová et al., 2015), and the Contingency of Soil Microorganisms and the Selected Soil Biotic and Abiotic Parameters Under Different Land-Uses (Júdová et al., 2019).

Ukrainian scientists comprehensively and thoroughly investigate the properties of soils, in particular, the selective absorption of heavy metals by soil and humic acids at different pH levels (Miroshnychenko, Kutz, 2016), the coloristic criteria of the S-matrix of brownish ashy gleyed soils of the Pre-Carpathian region (Nikorych, Chervonogrodska, 2016), the issues of diagnostics of elemental soil processes and profile-differentiated soils in the Pre-Carpathian region (Smaga, 2016), and the ecological and evolutionary analysis of the content of lithium in soils (Dmytruk, 2016). They also predict the content of chemical elements in soils of different genesis for assessing the ecological and energy status (Samokhvalova et al., 2016), study soil features of floodplain soils that limit the growth of energy crops (Kholodna, 2016), and assess the antideflationary efficiency of the "No-till" technology in the conditions of the southern steppe of Ukraine (Chornyy, Volosheniuk, 2016). We fully agree with the ideas of Prof. Medvedev, who believes that one of the most important tasks of the state is monitoring of soil cover on the basis of the latest software, mathematical, instrumental, and cartographic principles in compliance with the European experience (Medvedev, 2016).

Scientists of Dnipropetrovsk school of soil have been studying soil properties, the nature of soil formation, and the genesis of soils in the southeast of Ukraine (Belgard, 1950, 1971; Bilova, Travleev, 1999). The monitoring of the influence of a mole rat's digging activity on the restoration of proteolytic activity of soils under conditions of their technogenic contamination (Zamesova, 2016), the presence of heavy metals in the subsoil waters of Prisamarja Dniprovskogo (Kotovych, 2016), the dielectric penetration of the ravine soils (Gorban, 2016), the macro- and micromorphological differentiation of the humus-accumulative horizons of forest soils (Yakovenko, 2016), the effect of soil on spatial variation of the herbaceous layer modulated by overstorey in an eastern European poplar-willow forest (Zhukov et al., 2019) as well as the long-term monitoring of land recultivation (Zverkovsky, Zubkova, 2016), the features of the structural and aggregate composition of black soil in ravine forests (Gorban et al., 2016), and lithologic heterogeneity of the profile of ravine soils (Yakovenko, 2017) is being conducted.

The ravine forests of southeastern Ukraine are of great interest for scientific researchers-ecologists. In ravine forests, there are soils characterized by unique ecological, in particular, microclimatic features of soil-forming processes. The purpose of this study is to specify the structural and aggregate composition, water resistance of aggregates and microstructure formation of soils of the southern exposure of the ravine as well as to compare the study results with the soil properties of the northern exposure of the research object, which we studied earlier (Bozhko, Bilova, 2010).

Material and methods

Natural ravine forests of the northern variant of Dnipropetrovsk region emerged on the plateau located on the right bank of the Dnipro River. As an example of such ecosystems, we have chosen "Kapitanivskiy" ravine situated at 48°46.1502'N, 35 °37.3164'E (Fig. 1).

The methodological approach of our research is based on the typological principles developed by A. L. Belgard (Belgard, 1971) for steppe forests as well as the methodological principles of ecological micromorphology of soils proposed by N. A. Bilova and A. P. Travleev (Bilova, Travleev, 1997). Both field studies, geobotanical description and bioecological characteristics of flora are based on generally accepted methods and approaches. The micromorphological structure of soils has been studied in accordance with the methods developed by O. I. Parfyonova and K. A. Yarilova (Parfyonova, Yarilova, 1977), S. A. Shoba (Shoba, 1981). Transparent slides have been made in accordance with E. F. Mochalova's method (Mochalova, 1956), "Methodological manual on micro-morphology of soils" edited by G. V. Dobrovolsky (Dobrovolsky, 1983) has been used during decryption; identification of aggregate composition has been carried out by dry sifting of soil samples through a sieve with each soil profile divided into 10 zones, 10 cm each, and soil samples have been taken from top to bottom along the cut; the coefficient of soil structure $K = C/B$

Fig. 1. Location of the researched object: "Kapitanivskiy" ravine (1).

has been determined in accordance with I. B. Revut's method (Revut, 1964; Nweke, Nnabude, 2014), where *C* is the number of structural separates of 0.25–8.00 mm, *B* is the sum of separates larger than 8.00 mm, and silty separates smaller than 0.25 mm; analyses of water resistance of structural aggregates have been carried out in compliance with M. E. Bekarevich's and M. V. Krechun's method (Bekarevich, Krechun, 1964; Molina et al., 2001); the average square deviation was determined by formula:

$$\sigma = \sqrt{\frac{\sum_{i=1}^{n}(x_i - \bar{x})^2}{N-1}} \quad ;$$

the coefficient of indicators correlation was determined by formula:

$$r = \frac{\sum_{i=1}^{n}(x_i - \bar{x})(y_i - \bar{y})}{\sqrt{\sum(x_i - \bar{x})^2 \times \sum(y_i - \bar{y})^2}} \quad .$$

Results

The sample area under study is laid in the middle third of the slope at 20° of the southern exposure. Humidifying is atmospheric-transit, inflowing-outflowing. Microrelief is wavy. The type of forest biogeocoenosis is a fresh ash-tree and maple wood. The timber stand consists of 40% *Fraxinus excelsior* L., 30% *Acer compestre* L., and 30% *Quercus robur* L. Light structure is half-shade. Crown density is 0.7. The total coverage of the grass stand is 50%. Bioecological certification of the grass stand is presented in Table 1.

T a b l e 1. Bioecological certification of the ravine grass stand.

№	Plantname	Layer	Height (cm)	Vegetation stage	Abundance according to Drude	Coverage (%)	Vitality points
1	*Pulmonaria obscura* Dumort	H_2	16	v, ~	Cop_1(gr)	2	4–5
2	*Scutellaria altissima* L.	H_1	45	#, ~	Sp	2	4
3	*Stella riaholostea* L.	H_2	12	–	Cop_1(gr)	3	4
4	*Viola odorata* L.	H_3	5	–,	Sp	2	3–4
5	*Chaerophyllum temulum* L.	H_3	5	v, +	Sp	1	4

Notes: Layer: H_1 – the highest layer; H_2 – middle layer; H_3 – the lowest layer; vegetation stage: – vegetating; ~ – vegetating after fruiting; v – the state of rosette; + – unripe fruits; # – mature fruits; abundance according to Drude: cop1 – very scattered; sp – scattered (few); gr – groups.

The dead cover is fragmentary consisting of the leaves of tree species and dead herbs. There are diggings of mouse-type rodents. The soil is forest black soil, carbonate, low forest covered, low-eroded, medium-humus, medium-loamy on forest loams.

2.5–0 cm. The forest floor consists of semi-rotten, semi-stuck wood species, brown rotten wood that separated from the soil.

0–25 cm horizon. The horizon is dark gray, almost black, moist, humus-sedentary, loose, with many roots, coarse-pored. It almost entirely consists of the excrements of earthworms and other representatives of soil mesofauna (Fig. 2a). There is a large number of plant residues at different stages of development. The color is dark brown, almost black, homogeneous throughout the ground slide due to the high content of organic compounds. The plasma of humus is clayey, homogeneous throughout the ground slide, anisotropic, but largely hidden by organic compounds, the glowing is speckled. Plant residues are mostly fresh and low-rotten. Humus consists of humus clusters and colloform fresh-brown humus. The type of humus is mull. The visible pore space occupies a significant area (40–45%). Pores are of irregular structure, interaggregate. This area is dominated by aggregates of zoogenic origin (coprolites), which are mainly isometric and slightly elongated, of organic-mineral structure. Interaggregate pores contain remains of small invertebrates. Microstructure is of an aggregated and spongy type. Elemental microstructure is plasma-silty. The skeleton is dominated by quartz and feldspars of isometric and slightly elongated shape.

25–60 cm horizon. Dark gray, moist, granular and fine granular, soft, loamy. Effervescence of $CaCO_3$ is effected by HCl at a depth of 42 cm. The color is dark brown, uniform throughout the ground slide, along with dense, opaque organic clusters of round shape with a diffusive outline. The plasma is humus-clayey, homogeneous throughout the ground slide, anisotropic (Fig. 2b). The glow is speckled, inhomogeneous throughout the ground slide, organic matter consists of slightly rotten and fresh plant residues. The humus includes a sufficient number of scattered humus clusters and light brown humus. The type of humus is mull. Pores are of interaggregate irregular shape, there are also round inwardly aggregate pores and cracks (Figs 2c, d). Visible pores occupy a sufficiently large area (30–40%). Aggregates are mostly coprolite-type, isometric, with organic-mineral structure, of a round and isometric shape. Microstructure is aggregated and spongy (Fig. 2d). Elemental microstructure is plasma-silty.

The skeleton is dominated by quartz and feldspars of slightly elongated isometric shapes of various sizes, mostly medium-rounded.

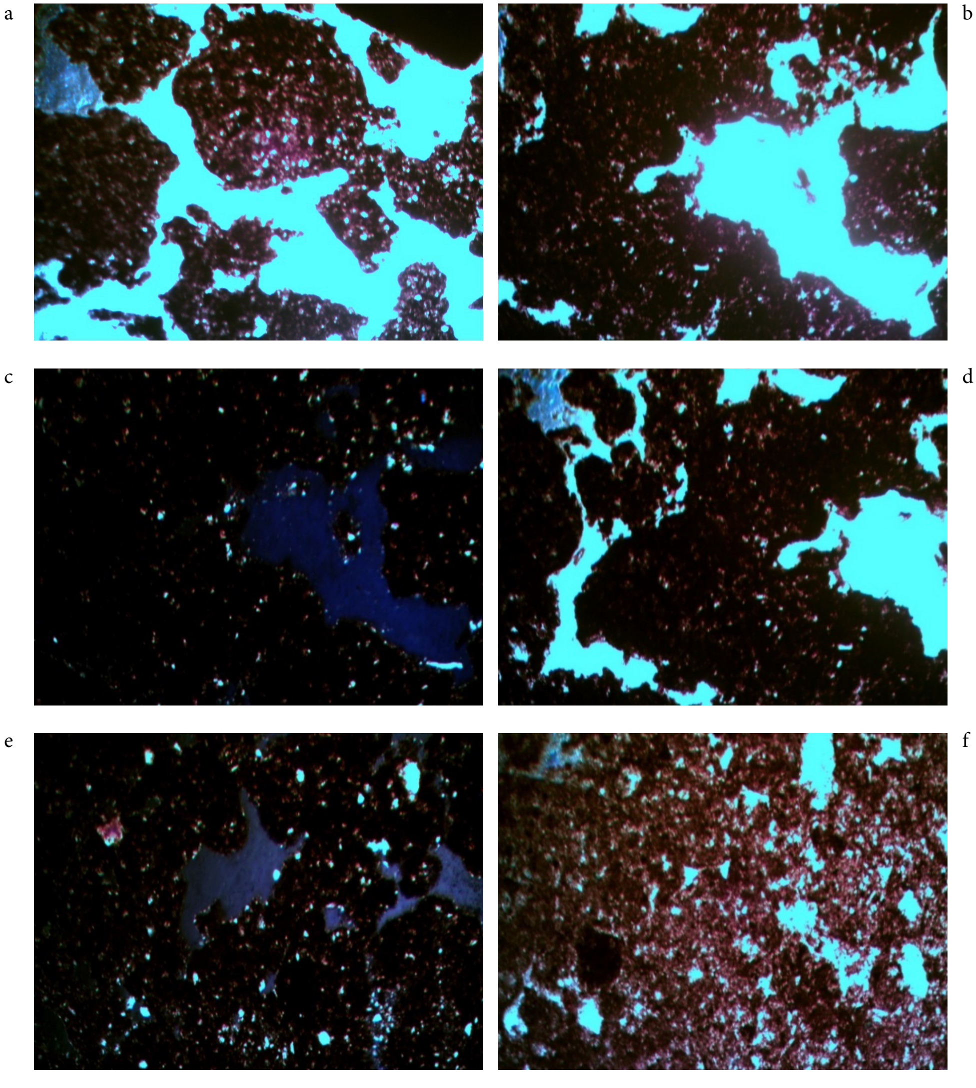

Fig. 2. Micromorphological structure of soils of the northern ravines of Dnipropetrovsk region, ×60:
a – nicols are parallel, there is coprolite in pore space (0–25 cm); b – nicols are parallel, a microstructure is of an aggregate and spongy type (0–25 cm); c – nicols are crossed, humus–clay anisotropic plasma, elemental plasma-silty microstructure (25–60 cm); d – nicols are parallel, the system of pores, skeleton, plant remains (25–60 cm); e – nicols are crossed, pores, skeleton (60–85 cm); f – humus carbonate-clay plasma, anisotropic, visible porosity takes much less space.

60–85 cm horizon. Dark gray, with a brown shade, gradually becoming lighter, noticeably dense, granular and fine granular, loamy. The transition between horizons is gradual. The color is brownish, heterogeneous due to less humus content. The plasma is humus-carbonate-clayey, inhomogeneous, anisotropic. The glow is speckled throughout the ground slide (Fig. 2e). The walls of the pores have anisotropic films (cutans), of mineral structure, which is the result of lessivage. There are few plant remains, mainly half-rotten. Organic compounds are much less. Microstructure is spongy, not aggregated. Pores are mostly irregular, rounded, of a narrow elongated shape (Fig. 2f). There is a large number of parallel and intersecting cracks.

Microstructure is of a spongy type. Elemental microstructure is plasma-silty. The skeleton is dominated by feldspars and quartz of various sizes, dominated by isometric shapes, slightly rounded.

85–120 cm horizon. Dark brown with a straw-colored shade, which is getting much lighter and denser at depths. The color of a sample is from light brown to brown. There are dense opaque rounded organic clusters with a diffusive outline. Elementary microstructure is plasma-silty. The plasma is humus-carbonate-clayey, inhomogeneous, anisotropic. The glow is speckled throughout the ground slide. The walls of the pores have signs of lessivage. The plant remains are scarce, mainly slightly rotten. Organogenic components are much fewer. The type of humus is mull having the same shapes, but in a much smaller number. Microstructure is of a spongy type. Elementary microstructure is plasma-silty. The skeleton is dominated by feldspars and quartz of various sizes, mainly isometric. Pores are narrow and elongated, irregular. Visible pores take much less (10–15%) area.

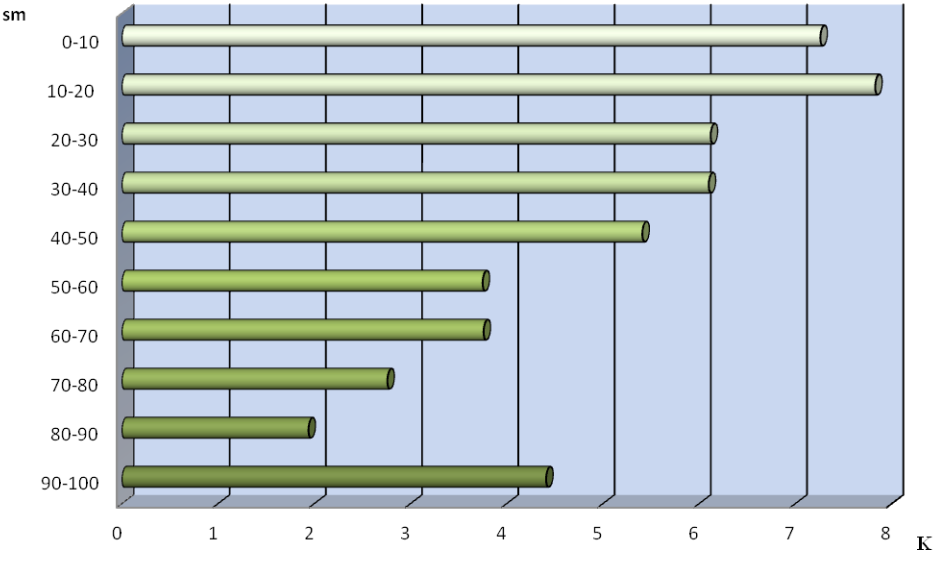

Fig. 3. The coefficient of pedality of the ravine soils.

The deeper the horizon is, the lower the coefficient of pedality (*K*) becomes (Fig. 3). Maximum value of *K* is in the 10–20 cm horizon (7.83 ± 0.81), and minimum value is in the 80–90 cm horizon (1.94 ± 0.15). The sum of 0.5–2.0 mm aggregates has a fairly high content. The greatest value of this indicator is in the 10–20 cm horizon (65.98 ± 2.07%), and the smallest is in the 50–60 cm horizon (30.51 ± 0.74%) (Fig. 4).

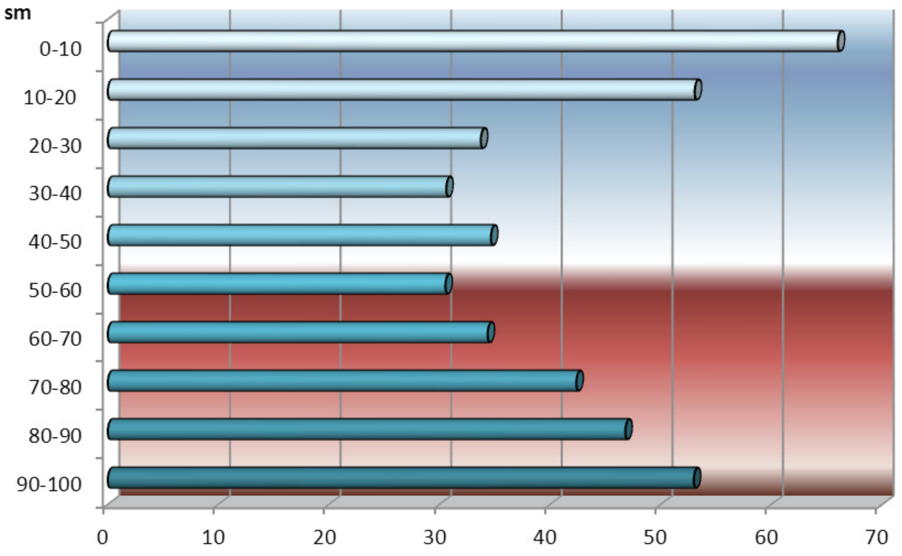

Fig. 4. The sum of 0.50–2.00 mm aggregates of the ravine soils.

The biometric correlation analysis as a combination of methods for detecting correlation dependence between two or more random features has been used by us to obtain more accurate information on the nature and strength of the relationship between the characteristics of the studied soils. The correlative connection is characterized by the correlation coefficient *r*. We have examined the relationship between two indicators of the aggregate analysis results: X_i value is the sum of aggregates having size from 0.5 to 2.0 mm (Σ0.5–2.0 mm), Y_i value is the coefficient of pedality (*K*) of the same samples of the ravine soil of the middle third of the slope of the southern exposure. The results of the calculations have shown that the correlation coefficient between the indicators Σ0.5–2.0 mm and *K* is 0.76, which is a high correlation.

The deeper the horizon is, the lower the water stability of soil aggregates becomes (Table 2). The coefficient of pedality reaches maximum value (90.01 ± 3.07% and 86.13 ± 4.11%) in the 0–10 cm horizon (1.00–2.00 mm fraction and 0.50–1.00 mm fraction, respectively) and minimum value (37.78 ± 3.84%) in the 90–100 cm horizon (fraction of 0.25–0.50 mm). The

T a b l e 2. Water stability of structural aggregates of the ravine soils.

Horizon (cm)	**Fracture (mm)**	**Aggregate structure (%) as a result of percolation**			**Total sum of all aggregates (%)**
		1.00	**0.50**	**0.25**	
1	**2**	**3**	**4**	**5**	**6**
0–10	1.00–2.00	54.18 ± 1.03	23.89 ± 0.76	11.94 ± 0.56	90.01 ± 3.07
	0.50–1.00	61.71 ± 1.62	16.92 ± 2.08	6.50 ± 0.09	86.13 ± 4.11
	0.25–0.50	13.98 ± 1.02	34.15 ± 1.74	8.78 ± 0.19	56.91 ± 3.08
10–20	1.00–2.00	34.29 ± 0.97	21.06 ± 2.19	26.02 ± 0.04	81.37 ± 3.75
	0.50–1.00	16.08 ± 0.87	41.40 ± 1.77	16.69 ± 0.61	74.17 ± 3.94
	0.25–0.50	11.08 ± 0.73	24.39 ± 1.50	14.76 ± 0.24	50.23 ± 2.88
20–30	1.00–2.00	29.91 ± 1.81	36.51 ± 1.03	8.31 ± 0.28	74.73 ± 3.61
	0.50–1.00	17.12 ± 0.63	35.13 ± 1.27	11.68 ± 0.99	63.93 ± 3.15
	0.25–0.50	11.22 ± 1.02	26.04 ± 2.05	7.11 ± 0.08	44.37 ± 3.37
30–40	1.00–2.00	24.96 ± 1.84	29.99 ± 0.71	8.58 ± 0.81	63.53 ± 2.62
	0.50–1.00	26.05 ± 2.09	28.17 ± 1.31	3.65 ± 1.04	57.87 ± 4.78
	0.25–0.50	16.54 ± 0.69	28.12 ± 1.66	12.36 ± 0.32	57.02 ± 3.87
40–50	1.00–2.00	34.05 ± 1.65	13.02 ± 1.32	9.55 ± 0.47	56.62 ± 3.94
	0.50–1.00	20.11 ± 1.14	21.87 ± 0.84	8.03 ± 0.30	50.01 ± 2.79
	0.25–0.50	18.20 ± 0.09	13.01 ± 2.06	12.05 ± 1.13	43.26 ± 3.70
50–60	1.00–2.00	32.12 ± 2.08	19.06 ± 1.91	2.43 ± 0.05	53.61 ± 1.83
	0.50–1.00	17.22 ± 0.86	21.07 ± 1.36	3.74 ± 0.52	42.03 ± 3.08
	0.25–0.50	9.04 ± 0.30	25.36 ± 1.17	3.38 ± 0.08	37.78 ± 3.84
60–70	1.00–2.00	29.38 ± 3.11	23.06 ± 1.29	3.73 ± 0.78	56.17 ± 2.08
	0.50–1.00	20.71 ± 3.14	23.02 ± 2.19	7.86 ± 0.75	51.59 ± 2.93
	0.25–0.50	12.39 ± 1.61	16.83 ± 2.71	13.31 ± 1.73	42.53 ± 3.84
70–80	1.00–2.00	21.36 ± 0.67	24.05 ± 1.58	5.76 ± 1.08	51.17 ± 2.06
	0.50–1.00	18.69 ± 1.99	25.65 ± 3.06	6.84 ± 2.68	50.18 ± 1.07
	0.25–0.50	11.51 ± 2.71	18.94 ± 0.88	17.63 ± 1.51	48.08 ± 2.17
80–90	1.00–2.00	28.14 ± 1.31	19.31 ± 2.55	3.57 ± 0.07	51.02 ± 4.18
	0.50–1.00	23.61 ± 3.17	18.64 ± 2.11	6.42 ± 1.15	48.67 ± 3.21
	0.25–0.50	18.01 ± 0.18	18.99 ± 1.98	10.14 ± 0.65	47.14 ± 3.51
90–100	1.00–2.00	28.31 ± 0.22	17.26 ± 3.05	4.63 ± 0.24	50.20 ± 4.08
	0.50–1.00	22.44 ± 0.88	18.09 ± 2.41	7.53 ± 0.60	48.06 ± 4.76
	0.25–0.50	12.29 ± 1.04	18.04 ± 0.84	16.11 ± 0.66	46.44 ± 3.43

Note: 0.25–1.00 mm aggregates, as being the most productive for black soil, were analyzed; the study was conducted three times.

coefficient of correlation of water resistance indicators of 1.00–2.00 and 0.50–1.00 mm aggregate fractions is 0.97, which is a very high correlation.

The determination of the correlation between two indicators—the coefficient of pedality (K) and water stability of soil aggregates of 1.00–2.00 mm size—has identified a high correlation coefficient of 0.76.

Discussion

Micromorphological studies have shown a high degree of soil aggregation. Upper horizons are of dark gray, almost black color, homogeneous throughout the area of the micromorphological ground slide due to the high content of organic compounds. The horizon is moist, humus-eluvial, loose, rich in roots, coarse-pored. The upper layer of the soil is solid and humus and almost completely consists of excrements of earthworms and other representatives of the soil mesofauna. The main structure-forming role belongs to dew-worms and small invertebrates. The plasma of humus is clayey, homogeneous throughout the ground slide, anisotropic, but largely hidden by organic compounds, the glow is speckled. The plant remains are fresh and slightly rotten. Humus consists of humus clusters and collomorphic fresh-brown humus. The type of humus is mull. The small areas of aggregated and spongy microstructure prevail. Visible porosity occupies a significantly large area. In the upper horizons there are a great number of large, mainly round and oval pores, which contributes to good aeration at great depth. The deeper it gets, the smaller the area of visible surface is, decreasing from 55% to 5%, the soil becomes denser and the pores gradually turn into cracks. The upper horizons are rich in isometric and slightly elongated aggregates of zoo origin (coprolites) with organic and mineral structure. Interaggregate pores partly incorporate residues of small invertebrates. Microlayers are mostly aggregated and sometimes spongy, depending on the microzone of the soil slide. Elemental microstructure is of a plasma-pulverescent type. The skeleton mainly consists of medium-rounded quartz and feldspars of isometric and slightly elongated shape. The soils are carbonate, the effervescence of $CaCO_3$ effected by HCl occurs at 42 cm depth.

The color of lower horizons varies from light brown to dark brown with a pale-yellow shade, becoming much lighter and thicker at depth. There are areas of dense opaque organic clusters of a round shape with a diffusive outline. The plasma is humus-carbonate-clayey, inhomogeneous, anisotropic. The glow is speckled throughout the ground slide. The walls of the pores incorporate anisotropic films, of mineral structure, which points to lessivage. The plant remains are scarce, mostly slightly rotten. There are much fewer organogenic compounds. The type of humus is mull having the same shapes, but in a much smaller quantity. Pores are narrow, elongated, and irregular. Visible porosity occupies a much smaller area (10–15%). Elemental microstructure is of a plasma-silty type.

The results of determining the aggregate composition of soils indicate a high aggregation of humus horizons. The coefficient of soil structure (K) reaches the greatest value (7.83 ± 0.81) in the 10–20 cm horizon, while the lowest value (1.94 ± 0.08) can be seen in the 80–90 cm horizon. The deeper the horizon is, the less the sum of 0.5–2.0 mm aggregates becomes. The greatest value of this coefficient can also be indicated for humus horizons (65.98 ± 1.63) in the 0–10 cm horizon.

The deeper the horizon is, the lower the water stability of soil aggregates becomes. The maximum value of this indicator is 90.01 ± 3.07% in the 0–10 cm horizon (1.00–2.00 mm fracture). The greater the depth is, the more the indicator of this fracture falls gradually up to 50.20 ± 4.08% at 90–100 cm depth. The water resistance of 0.50–1.00 mm fracture is also very high in the upper horizons (86.13 ± 4.11%) in the horizon of 0.50–1.00 cm and gradually decreases, the greater the depth becomes, up to 48.06% at 90–100 cm depth. The water resistance indicators of 0.25–0.50 mm fracture are the lowest and range from 56.91 ± 3.08% to 46.44 ± 3.43%.

The coefficient of correlation of water stability indicators of 1.00–2.00 and 0.50–1.00 mm aggregate fractions is 0.97, which is a very high correlation. The determination of the correlation between two indicators—the coefficient of pedality (*K*) and water resistance of soil aggregates of 1.00–2.00 mm size (as the highest value of three fractions)—has shown a high correlation coefficient of 0.76. This testifies that different qualitative indicators of soil properties synchronously decrease the deeper the soil slide is.

Earlier we investigated the properties of the ravine soils on the example of the middle third of the slope of the northern exposure (Bozhko, Bilova, 2010). The coefficient of soil pedality here turned out to be extremely high—10.53 ± 1.22 and 12.74 ± 1.79 in the 10–20 cm horizon and 30–40 cm horizon, respectively, while water stability of structural aggregates reaches 95.07 ± 2.17% and the power of the humus horizon is greater. The comparison between the coefficient of soil pedality and water stability of two ravine slopes is presented in Table 3.

T a b l e 3. The comparison between the coefficient of soil pedality and water stability of two ravine slopes.

Horizon (cm)	The slope of the southern exposure		The slope of the northern exposure	
	K	***J***	***K***	***J***
0–10	7.26 ± 0.94	90.01 ± 3.07	7.05 ± 1.06	94.67 ± 1.16
10–20	7.83 ± 0.81	81.37 ± 3.75	10.53 ± 1.22	95.07 ± 2.17
20–30	6.12 ± 0.17	74.73 ± 3.61	8.53 ± 1.51	86.75 ± 1.34
30–40	6.10 ± 0.21	63.53 ± 2.62	12.74 ± 1.79	93.62 ± 2.08
40–50	5.41 ± 0.51	56.62 ± 3.94	3.67 ± 0.74	67.94 ± 2.71
50–60	3.75 ± 0.44	53.61 ± 1.83	4.69 ± 0.85	83.15 ± 3.17
60–70	3.76 ± 0.39	56.17 ± 2.08	4.39 ± 1.01	80.37 ± 2.61
70–80	2.76 ± 0.11	51.17 ± 2.06	1.98 ± 0.08	81.55 ± 1.96
80–90	1.94 ± 0.24	51.02 ± 4.18	2.23 ± 0.37	64.16 ± 2.48
90–100	2.41 ± 0.19	50.20 ± 4.08	2.24 ± 0.71	81.32 ± 3.06

Notes: *K* – coefficient of pedality; *J* – water stability of 1.00–2.00 mm aggregates.

This difference is explained by the following factors: on the slope of the northern exposure, the type of forest vegetation is a fresh clay loam, in contrast to the southern exposure where it is a much fresher clay loam; the type of forest biogeocoenosis is a fresh lime-ash-maple wood. The light structure is half-shadowy. The density of the tree crown is 0.8. Such conditions are the most favorable for soil formation and active processes of soil humification.

Conclusion

The study of the soil structure and aggregation, water stability of aggregates, and ecological and micro-morphological features of soils of the ravine forests of Pre-Dnipro region

(Ukraine) on the example of the northern variant of the ravine forest "Kapitanivskiy" has identified a high degree of aggregation of the upper horizons of the soil slide. The structure is of a zoogenic origin. The aggregates of mainly coprolite nature contain well-rotten plant remains. The dark gray, almost black color throughout the area of the micromorphological slide is due to a large number of organic compounds, which points to active processes of humification. Finely dispersed humus includes a large number of equally spaced humus clusters. The type of humus is mull. The area of the visible surface of the pores in the upper horizons of the soil profile is fairly large (40%). Pores are round and elongated. Typically, remains of small invertebrates can be found in the pores. The deeper the soil slide is, the smaller the area of visible pores (from 55 to 5%) together with aggregation becomes. Correlating with micromorphological characteristics, the water resistance of structural aggregates reaches high values (90.01 ± 3.07%) in the upper mummified coprolite horizons of the soil slide, decreasing at depths. The coefficient of pedality in ravines reaches 7.83 ± 0.81. The sum of 0.5–2.0 mm aggregates is 65.98 ± 2.07%.

Comparing the soil properties of the southern and northern exposure of the object under study, we have come to the conclusion that the soil structure and aggregation, water stability of aggregates, and soil micromorphological indicators are much better on the slope of the northern exposure. Consequently, more favorable environmental conditions here, especially microclimatic, are the most suitable for active processes of humification and microstructure formation.

In general, the soils of the northern ravine forests of Pre-Dnipro region (Ukraine) are characterized by an active biogenic microstructure formation, which results in significant aggregation and looseness of the microstructure, which greatly increases the fertility of these soils.

References

Bartz, M.L.C., Pasini, A. & Brown G.G. (2013). Earthworms as soil quality indicators in Brazilian no-tillage systems. *Appl. Soil Ecol.,* 69, 39–48. DOI: 10.1016/j.apsoil.2013.01.011

Bekarevich, N.E. & Krechun Z.A. (1964). The water- soil structure and its determination by analysis of aggregate (in Russian). In *Methods of research in the field of soil physics* (pp. 132–164).

Belgard, A.L. (1950). *Forest vegetation of southeast of Ukrainian SSR (in Russian)*. Kiev: Lesnaja Promyshlennost'.

Belgard, A.L. (1971). *Steppe forestry (in Russian)*. Moskva: Lesnaja Promyshlennost'.

Bilova, N.A. (1997). *Ecology, micromorphology, anthropogeny forest soils of the steppe zone of Ukraine (in Russian)*. Dnepropenrovsk: DNU.

Bilova, N.A. & Travleev A.P. (1999). *Natural forest and grassland soils (in Russian)*. Dnepropenrovsk: DNU.

Bogner, C., Bauer, F., Trancón y Widemann, B., Viñan, P., Balcazar L. & Huwe B. (2014). Quantifying the morphology of flow patterns in landslide-affected and unaffected soils. *J. Hydrol.*, 511, 460–473. DOI: 10.1016/j.jhydrol.2014.01.063

Bozhko, K.M & Bilova N.A. (2010). Soil and geobotanical characteristic as well as micromorphological characteristics of soil in the ecosystem "Kapitanivskiy unhomogeneous forest in the baulk" on the example of slope of the northern exposition and the thalweg of the unhomogeneous forest (in Ukrainian). *Optimization and Protection of Ecosystems* (pp. 181–191). Simferopol: TNU.

Buza, A. K. & Divos F. (2016). Spruce tree fighting back – study of honey fungus infection. *Folia Oecologica*, 43, 204–207.

Chornyy, S.G. & Volosheniuk A.V. (2016). Evaluation of wind erosion protective efficiency No-till technology in southern Ukraine steppe conditions (in Ukrainian). *Gruntoznavstvo*, 17(3–4), 50–63. DOI: 10.15421/041613.

Dobrovolsky, G.V. (1983). *Methodological guidance on soil micromorphology (in Russian)*. Moskva.

Dmytruk, Y.M. (2016). Ecological-evolutionary analysis of lithium content in soils (in Ukrainian). *Gruntoznavstvo*, 17(1–2), 31–39. DOI: 10.15421/041603.
Epelde, L., Becerril, J.M., Alkorta, I. & Garbisu C. (2014). Adaptive long-term monitoring of soil health in metal phytostabilization: Ecological attributes and ecosystem services based on soil microbial parameters. *International Journal of Phytoremediation*, 16, 971–981. DOI: 10.1080/15226514.2013.810578.
Eshaghi Rad, J., Valadi, G. & Zargaran M.R. (2017). Effect of man-made disturbances on understory plant richness of oak forests in Iran. *Folia Oecologica*, 44, 61–68. DOI: 10.1515/foecol-2017-0008.
Gáfriková, J., Hanajík, P. & Zvarík M. (2018). Determination of organic fractions and enzymatic activity in forest spruce soil of Tatra National Park. *Ekológia (Bratislava)*, 37(4), 328–337. DOI: 10.2478/eko-2018-0024.
Gorban, V.A. (2016). To the method of studying the permittivity of soils (on an example of soils of ravine forests of the northern variant of the steppe zone of Ukraine (in Russian). *Gruntoznavstvo*, 17(3–4), 90–97. DOI: 10.15421/041616.
Júdová, J., Kanianska, R., Jaďuďová, J., Kizeková, M. & Makovníková J. (2019). The contingency of soil microorganisms and the selected soil biotic and abiotic parameters under different land-uses. *Ekológia (Bratislava)*, 38(2), 101–116. DOI: 10.2478/eko-2019-0008.
Kholodna, A.S. (2016). Soil factors of floodplain soils that limit growth of energy crops (in Ukrainian). *Gruntoznavstvo*, 17(3–4), 43–49. DOI: 10.15421/041612.
Khormali, F., Ghergherechi, S., Kehl, M. & Ayoubi S. (2012). Soil formation in loess-derived soils along a subhumid to humid climate gradient, Northeastern Iran. *Geoderma*, 179–180, 113–122. DOI: 10.1016/j.geoderma.2012.02.002.
Kotovych, O.V. (2016). Heavy metals in the ground waters of Prisamarya Dniprovske (in Ukrainian). *Gruntoznavstvo*, 17(3–4), 98–106. DOI: 10.15421/041617.
Medvedev, V.V. (2016). Methodology of effective monitoring of a soil cover (on the basis of the analysis of 25-years European experience) (in Ukrainian). *Gruntoznavstvo*, 17(3–4), 5–14. DOI: 10.15421/041609.
Miroshnychenko, N.N. & Kutz O.A. (2016). Selective absorption of heavy metals by soil and humic acids at different pH levels (in Ukrainian). *Gruntoznavstvo*, 17 (1–2), 74–82. DOI: 10.15421/041607.
Mochalova, E.F. (1956). Making thin sections of undisturbed soil with structure (in Russian). *Pochvovedenie*, 10, 6–38.
Molina, N.C., Caceres, M.R. & Pietroboni A.M. (2001). Factors affecting aggregate stability and water dispersible clay of recently cultivated semiarid soils of Argentina. *Arid Land Res. Manag.*, 15, 77–87.
Nikorych, V.A. & Chervonogrodska I.V. (2016). Colouristic criteria of S-matrix of the Precarpathians brownish-Podzolic gleyed soils (in Ukrainian). *Gruntoznavstvo*, 17(1–2), 49–63. DOI: 10.15421/041605.
Nsanganwimana, F., Marchand, L., Douay, F. & Mench M. (2014). *Arundo donax* L., a candidate for phytomanaging water and soils contaminated by trace elements and producing plant-based feedstock. A review. *International Journal of Phytoremediation*, 16, 982–1017. DOI: 10.1080/15226514.2013.810580.
Nweke, I.A. & Nnabude P.C. (2014). Aggregate size distribution and stability of aggregate fractions of fallow and cultivated soils. *Journal of Experimental Biology and Agricultural Sciences*, 1, 514–520.
Parfyonova, E.I. & Yarilova K.A. (1977). *Guide to micromorphological studies in soil science (in Russian)*. Nauka: Moskva.
Polláková, N., Šimanský, V., Ložek, O., Hanáčková, E. & Candráková E. (2015). The changes of nutrient and risk elements of top soil layers under canopy of different tree species and grassland in Arboretum Mlyňany, Slovakia. *Folia Oecologica*, 42, 29–34.
Polláková, N., Šimanský, V. & Jonczak J. (2017). Characteristics of physical properties in soil profiles under selected introduced trees in the Nature Reserve Arboretum Mlyňany, Slovakia. *Folia Oecologica*, 44, 78–86. DOI: 10.1515/foecol-2017-0010.
Ramezanpour, H. & Pourmasoumi M. (2012). Micromorphological aspects of two forest soils development derived from Igneous rocks in Lahijan (Iran). *Journal of Mountain Science*, 9, 646–655.
Revut, I.B. (1965). *Soil about yourself (modern view on the mechanical composition and structure of the soil) (in Russian)*. Nauka: Moskva.
Rui Morgado, Nuno G.C. Ferreira, Diogo N. Cardoso, Amadeu M.V.M. Soares & Susana Loureiro (2015). Abiotic-factors affect the performance of the terrestrial isopod *Porcellionides pruinosus*. *Appl. Soil Ecol.*, 95, 161–170. DOI: 10.1016/j.apsoil.2015.06.012.
Ruibo Sun, Xisheng Guo, Daozhong Wang & Haiyan Chu (2015). Effects of long-term application of chemical and organic fertilizers on the abundance of microbial communities involved in the nitrogen cycle. *Appl. Soil Ecol.*, 95, 171–178. DOI: 10.1016/j.apsoil.2015.06.010.

Samokhvalova, P.A. & Mandryk O.V. (2016). Forecasting the levels of trace elements and heavy metals content in soils of different genesis for the assessment of their environmental and productional functions (in Ukrainian). *Ecology and Noospherology*, 27(1–2), 74–79.

Smaga, I.S. (2016). Diagnostic problems of elementary soil processes and profile-differentiated soils of the Precarpathian region (in Ukrainian). *Gruntoznavstvo*, 17(1–2), 40–48. DOI: 10.15421/041604.

Tobiašová, E., Dębska, B. & Drag M. (2015). The assessment of the soil organic matter of different ecosystems according to parameters of carbon. *Folia Oecologica*, 42, 46–53.

Tužinský, L., Bublinec, E. & Tužinský M. (2017). Development of soil water regime under spruce stands. *Folia Oecologica*, 44, 46–53. DOI: 10.1515/foecol-2017-0006.

Yakovenko, V.M. (2016). Macro- and micro-morphological differentiation of humus-accumulative horizon of forest soils (in Ukrainian). *Gruntoznavstvo*, 17(3–4), 64–80. DOI: 10.15421/041614.

Yakovenko, V. (2017). Fractal properties of coarse/fine-related distribution in forest soils on colluvium. In D. Dent & Y. Dmytruk (Eds.), *Soil science working for a living* (pp. 29– 42). Switzerland: Springer International Publishing. DOI: 10.1007/978-3-319-45417-7.

Zamesova, T.A. (2016). Influens of mole rats burrow activity to restore the soils proteolytic activity in terms of their man-made pollution (in Ukrainian). *Gruntoznavstvo*, 17(3–4), 107–111. DOI: 10.15421/041618.

Zhukov, O., Kunah, O., Dubinina, Y., Zhukova, Y. & Ganzha D. (2019). The effect of soil on spatial variation of the herbaceous layer modulated by overstorey in an eastern European poplar-willow forest. *Ekológia (Bratislava)*, 38 (3), 253–272. DOI: 10.2478/eko-2019-0020.

Zúñiga, M.C., Feijoo M.A., Quintero, H., Aldana, N.J. & Carvajal A.F. (2013). Farmers' perceptions of earth worms and their role in soil. *Appl. Soil Ecol.*, 69, 61–68. DOI: 10.1016/j.apsoil.2013.03.001.

Zverkovsky, V. M. & Zubkova O.S. (2016). Dynamics of mine rocks and artificial soils agrochemical characteristics under the impact of long-term reclamation (in Ukrainian). *Gruntoznavstvo*, 17(1–2), 83–89. DOI: 10.15421/041608.

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GEODIVERSITY AND LANDSCAPE SERVICES IN THE REGION OF OGULINSKO-PLAŠĆANSKA ZAVALA, CROATIA

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Abstract

Butorac V., Buzjak N.: Geodiversity and landscape services in the region of Ogulinsko-Plašćanska Zavala, Croatia. Ekológia (Bratislava), Vol. 39, No. 2, p. 130–144, 2020.

In terms of spatial planning and environment protection procedures in Croatia, geomorphological features as a component of geodiversity are mostly considered marginally. They are considered locally in the scope of certain operations such as urban development, mining, or activities that are being assessed in the environment impact assessment procedures or spatial and strategical planning. Regarding the protection and the planning processes connected with it, geomorphological features should be considered in the right scale and with all of its values and services that are being provided to the environment on a landscape scale. In this paper, geodiversity and its role in landscape evolution will be connected and explained with the example of subgeomorphological region Ogulinsko-plašćanska Zavala, in the mountainous Dinaric karst part of Croatia. As it has been a region with long human and nature interaction, and a region with preserved natural and seminatural landscapes, it is a suitable area for such an analysis. In order to properly assess the geomorphological features as one of the determinants of landscape, the basis for environment impact assessment procedures and spatial planning procedures, geoecological analysis of geodiversity and landscape services occurrence and spatial distribution is carried out.

Key words: landscape typology, geodiversity, relative relief assessment method.

Introduction

Landscape as a term can be defined in different ways that are dependent on the aspect of a study. In this study, landscape is analyzed from physical geography aspect. It is the closest to the definition of landscape as a physical space, part (at various scales) of the Earth's surface, which is shaped by natural conditions and formed by human influences to a different extent (Bastian et al., 2014). Natural basis of landscapes is determined by its primary structure. Primary landscape structure is defined by the abiotic elements of geosystems, for example, geology, relief, soil, water, and air. It is the primary basis for other landscape structures (Miklós et al., 2019). Collective term for overall diversity of landscapes, landforms, and processes that form and reshape them on the Earth's surface and in the lithosphere (that includes their features, relationships, and systems) is defined as *geodiversity*. Its elements are geological, geomorphological, and pedological diversity (Gray, 2005; Serrano, Ruiz-Flaño, 2007; Gray, 2008, 2013; Gray et al., 2013; Brilha et al., 2018; Reynard, Brilha, 2018; Coratza et al., 2018; Zwolinski et al., 2018).

Landscape types are units that are relatively homogeneous in their composition. These are generic in nature since these may occur in different areas in different parts of the country and share broadly similar combinations of geology, topography, drainage, landforms, vegetation, historical land use, and settlement patterns (Swanwick, 2002).

Although it recognizes the importance of the landscape natural elements as a basis for spatial development, the spatial planning sector in Croatia does not currently sufficiently utilize geographical elements and holistic approaches such as ecosystem services and landscape services in the planning process (Marohnić-Kuzmanović et al., 2017). Recent concept is adapted to the strategic level of landscape planning and protection (Obad Šćitaroci et al., 2014). It provides good overview of the landscape state, as well as an overview of the methods and strategic guidelines used for landscape planning and protection. It does not sufficiently consider the primary structure of the landscape, which is not further elaborated on lower order components, while the secondary and tertiary landscape structures are elaborated in greater detail. An approach based on the cultural components of the landscape was emphasized and, unfortunately, did not go far from the landscape regionalization developed for spatial development by geographer I. Bralić in the mid 1990s (Bralić, 1999).

On the other side, the usual approach in the Croatian nature protection sector is based on the biological point of view or, rarely, ecosystem services approach. It recognizes habitat types (determined by vegetation areas) as basic spatial units, initially based on EUNIS habitat classification (EUNIS, 2019; HAOP, 2017). Besides positive aspects, one of the main problems with ecosystem services approach (or newer *Nature's Benefits to People* concept; Kadykalo et al., 2019) is the focus on the benefits for human society, economic framework or biodiversity, often not giving correct attention to the other ecosystems' components. Therefore, it can be used only as an additional method in landscape analysis (Birkhofer et al., 2015; Lele et al., 2013; Portman, 2013).

As an alternative to the concept of ecosystem units and ecosystem services, the concept of landscape units and *landscape services* (or functions) in landscape analysis is much more applicable due to the more holistic perspective. Termorshuizen and Opdam (2009) stressed the role of landscapes as "spatial human-ecological systems that deliver a wide range of *functions*, that are or can be valued by humans for economic, sociocultural, and ecological reasons". Developing the landscape services approach, Bastian et al. (2014) provided a wider look at the geographical space and its components. Due to the features and modifiers that shape *landscape types*, they are more appropriate spatial units than ecosystems in spatial planning and environment protection. Landscape services are defined similar to ecosystem services, but represent upgrade due to the more holistic perspective. They also account for the contributions of landscapes to human well-being (Bastian et al., 2014), but are also applicable to other landscape/ecosystem elements. Landscape services are mostly classified in three themes: provisioning, regulation and socio-cultural (Vallés-Planells et al., 2014).

As abiotic nature is greatly taking part in landscape types, one aim of this paper is to analyze spatial distribution and features of geodiversity, and its role in landscape typology. The other aim is to analyze primary landscape structure and to examine possibilities of landscape services' quantification with the use of relative relief assessment method. For this study provisioning, socio-cultural and regulation services were selected. This approach is focused on the integration of primary landscape structure or physical geographical aspect in combination with values of landscape through utilitarian value, which also includes some indicators of biodiversity. This kind of approach could bring new insights about landscape structure and values besides the usual approach of land cover characteristics analysis (Pătru-Stupariu et al., 2017; Zaušková, 2014; Mkrtchian, 2013).

Study area

Study area of Ogulinsko-plašćanska Zavala (Fig. 1) covers 300.24 km^2 and it is a part of the Mountainous Croatia in Dinaric karst mountain belt (Bognar, 2001; Bognar et al., 2012). It was named after two main settlements – Ogulin and Plaški. It is tectonically relatively low-

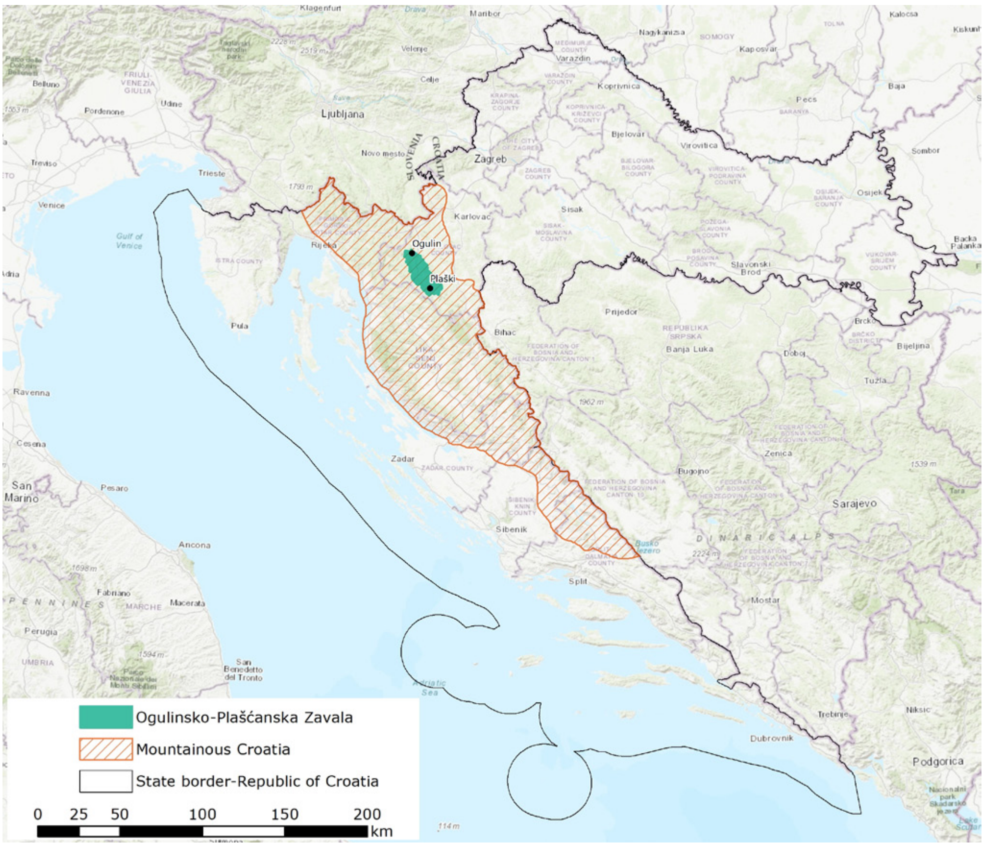

Fig. 1. Location of study area.

ered depression of Dinaric direction (Bočić et al., 2016) modelled by various geomorphological processes.

The spatial distribution and alteration of limestones and dolomites resulted in local differences in geomorphological processes and landscape properties (Fig. 2). Two main morphogenetic relief types determine the landscape: karst and fluviokarst. The karst was developed in areas built of Cretaceous limestones. It is characterized by dynamic landscape of sharper contours due to the steeper slopes and higher relative relief values. There are forest landscapes on thin soils and bare rocky surface in the mountains. The manmade grasslands are characteristic for karst plateaus. The diagnostic landscape elements are numerous karst phenomena: closed depressions on the surface (dolines and uvalas), subterranean phenomena (caves and shafts) and predominantly underground karst water circulation. Fluviokarst is a morphogenetic relief type typical for the dolomite and contact karst zones where relief developed by the combination of karst, fluvial and slope processes. There are also karst phenomena (dolines, caves, poljes, and blind valleys with ponors), underground water circulation, but also better developed surface stream network with often thicker fluvial deposits important for agriculture. This is the area of mosaic cultural and semi-

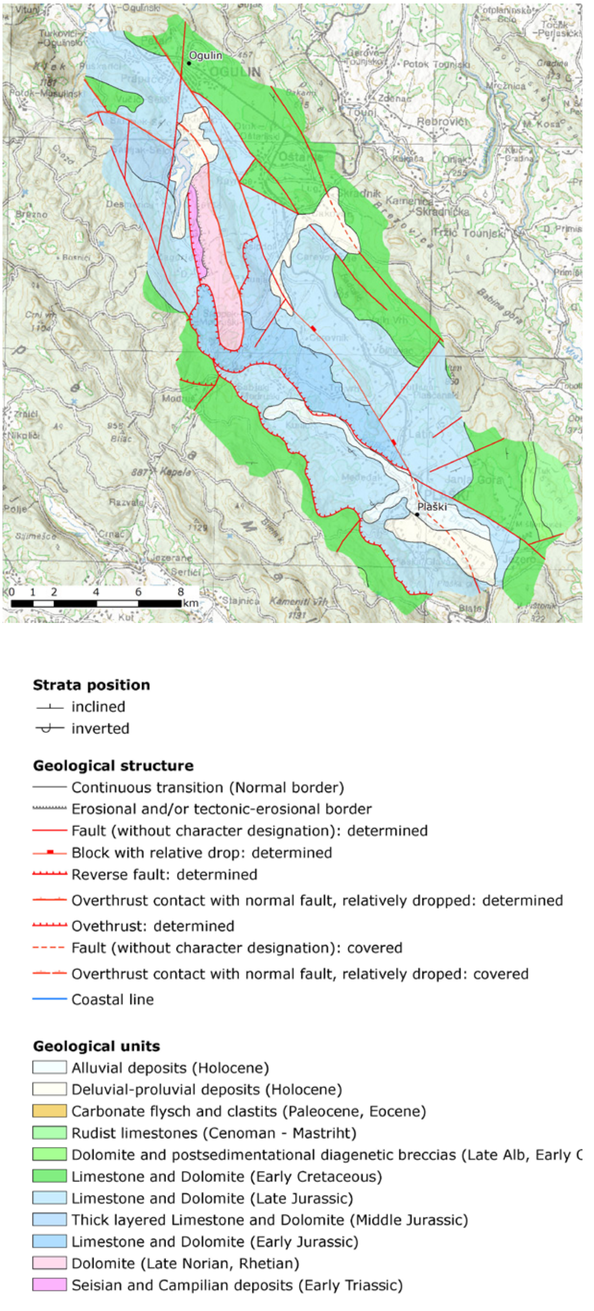

Fig. 2. Geological map of study area (after Velić, Vlahović, 2009).

natural landscapes with settlements, arable lands and forest patches. Valleys and karst poljes bottoms are covered with the younger Holocene alluvial and slope deposits.

The simultaneous occurrence of typical karst phenomena with big karst springs (Zagorska Mrežnica, Bistrac, Zagorska Peć, Rupečica) and surface hydrological network typical for non-karst areas are a result of geological conditions (Bahun, 1970). The largest stream Dobra is a losing fluviokarst river that flows into the research area from the NW. A losing river is a stream in karst in which water infiltrates underground because the groundwater table is below the bottom of the stream channel (Bonacci, Andrić, 2010a). River Dobra receives water from the well karstified area of the Velika Kapela Mt. Its valley developed along the part of an upper impounding zone, along the contact zone of well-karstified limestones, and less permeable dolomites that act as a hydrogeological barrier for karst drainage, causing specific valley geomorphology, hydrology, the occurrence of numerous springs and small streams contributing Dobra River from the SW (Bahun, 1968, 1970; Bonacci, Andrić, 2010b; Žganec, 2012). Dobra is also the sinking river with ponor developed at the contact between Jurassic dolomite zone and Cretaceous limestone zone. Ponor is located at the border of Ogulinsko polje in the city of Ogulin and continues in the 16.4 km long cave system Đula-Medvedica (Kuhta, Novosel, 2001). The springs, streams and valleys were important factors of settlement development and contribute to the landscape diversity of the area. The valleys attracted population with fertile soil and they were the cores of landscape transformation from mostly forests and wetlands to agricultural landscape.

Material and methods

To proceed the qualitative-quantitative assessment of landscape services and geodiversity, the most common geoecological method in Republic of Croatia, *relative relief assessment* method, was chosen. Its aim is to assess values of the landscape or any other spatial unit for certain landscape evaluation, economic activity or nature protection tasks (Bognar, 1990; Bognar, Bognar, 2010; Buzjak, 2008; Mamut, 2010a,b). A qualitative characterization of geodiversity consists of elements' analysis and explanation on assigned values. This qualitative approach includes proposals that are based on expert views. Geodiversity values are labelled or rated but always in a non-numerical form and tend to be highly subjective (Pereira, Pereira, 2010). Following geospatial methods were performed in ESRI ArcMap 10.3.

Delineation of study area was done on the ridge and thalweg map extracted from DEM (10 x10 m) with Spatial Analyst extension. The borderline is extracted from continuous ridges surrounding hypsometrically lower area.

Delineation of landscape types and subtypes was made based on primary and secondary criteria. Primary criterion is relative relief (Lozić, 1995). It was calculated from DEM (Focal statistics) and classified into 6 classes of which the first three are distinctive for *planated relief*, and the last three for *dissected relief*. With this first level of delineation, some of the borders were too generalized, therefore additional steps have been taken. On the northern and southern part of the study area, the boundary between two landscape types was set on elevation contour 320 m, similar to Bahun (1970). Subtypes were afterwards delineated by continuous ridges and thalwegs, which are visible structural borderlines between landforms. First level of delineation produced two *macrolandscape* types: *dissected relief* and *planated relief macro-landscape types*. Major difference between them are the overall geomorphological processes, which are influencing energy fluxes between natural processes and human activities, and those are denudation and accumulation in different proportions. The second level, *subtypes*, were defined by their location in study area (boundary or central), size (micro, meso or macro), landform type and morphogenetic features (karstic, fluviokarstic, and karst polje as a polygenetic landform type). Third level of landscape typology delineation was done between landscape subtypes, which are different and divided in morphologic and morphometric aspect. *Micro-types* are named after the names of settlements, peaks or landforms.

With the use of relative relief evaluation method provisioning, regulation and socio-cultural landscape services were also assessed. Input data were generated from DEM, topographic map 1:25k and geological map 1:100k, Non-Forest Terrestrial Habitat Map of Republic of Croatia (Bardi et al., 2016), and cave archive of the Speleological Section of Croatian

T a b l e 1. Landscape services valorization table.

Services	Service name	Indicator	Categories	Points
Provisioning services (100)	Freshwater (50)	Springs Kernel density (25)	0-1	8.4
			1-2	16.7
			2-3.85	25
		Drainage network density (25)	0.1-1	8.4
			1-2	16.7
			2-3.6	25
	Construction material (50)	Carbonates (50)	Presence	50
Regulation services (100)	Rock cycle (33.3)	Fault density (6.6)	0-0.3	8.4
			0.3-0.6	16.7
			0.6-1	25
		Dolines Kernel density (6.6)	0-20	0.825
			20-40	1.625
			40-60	2.45
			60-80	3.3
			80-100	4.125
			100-120	4.95
			120-140	5.775
			140-160	6.6
		Springs Kernel density (6.6)	0-1	2.2
			1-2	4.4
			2-3.85	6.6
		Caves (6.6)	Presence	6.6
		Ponors (6.6)	Presence	6.6
	Water regulation (33.3)	Karst polje (16.65)	Presence	16.65
		Drainage network (16.65)	Presence	16.65
	Conserving biodiversity (33.3)	Naturalness of vegetation (16.65)	Presence	16.65
		Caves (16.65)	Presence	16.65
Socio-cultural services (100)	Aesthetic values (50)	Vertical relief dissection (7.14)	0-5	1.4
			5-100	2.8
			100-300	4.2
			300-500	5.7
			500-800	7.14
		Naturalness of vegetation (7.14)	Presence	7.14
		Caves (7.14)	Presence	7.14
		Ponors (7.14)	Presence	7.14
		Dolines Kernel density (7.14)	0-20	0.89
			20-40	1.8
			40-60	2.69
			60-80	3.58
			80-100	4.47
			100-120	5.36
			120-140	6.25
			140-160	7.14
		Springs Kernel density (7.14)	0-1	2.9
			1-2	4.7
			2-3.85	7.14
	Recreation (50)	Naturalness of vegetation (12.5)	Presence	12.5
		Caves (12.5)	Presence	12.5
		Drainage network density (12.5)	0.1-1	4.16
			1-2	8.34
			2-3.6	12.5
		Protected areas (12.5)	Presence	12.5

Mountaineering Society "Željezničar", Zagreb. First step in landscape services assessment was inventory of specified features (Table 1). Criteria determination and scoring was done in MS Excel, and afterwards applied in ESRI ArcMap 10.3. Final landscape services assessment was proceeded with Spatial Analyst tools (Reclassify and Cell Statistics). In Table 1, three types of landscape services and their belonging indicators are listed together with their descriptions.

Geodiversity assessment was proceeded in order to quantify the occurrence of geodiversity elements. As perquisite, digital geomorphological analysis and geomorphological phenomena inventory was done. After that, vectorization and geostatistical analysis of collected data and determination of assessment criteria for geodiversity was performed. Criteria determination and scoring was done in MS Excel, and applied in ESRI ArcMap 10.3. Final geodiversity assessment was proceeded with Spatial Analyst tools (Reclassify and Cell Statistics). Criteria for geodiversity assessment was intrinsic value.

Results and discussion

Landscape typology was done according to the primary landscape structure. It consists of morphogenetic relief types with similar morphometric characteristics and assemblages of landforms. Delineation product is 2 macro-landscape types, 6 subtypes, and 13 micro-types (Fig. 3).

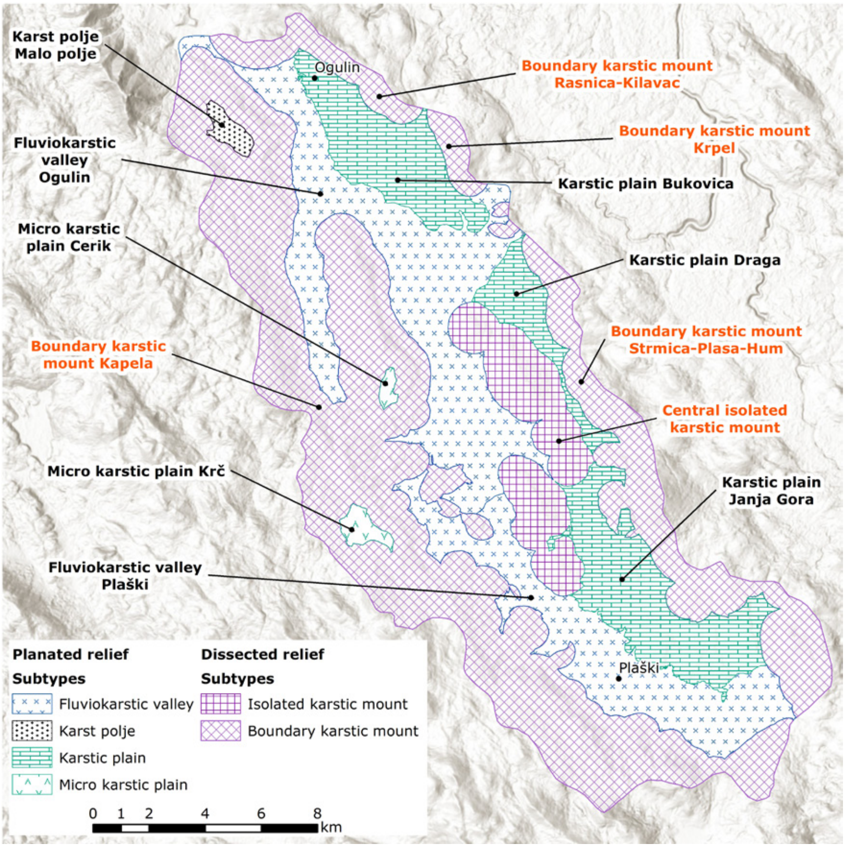

Fig. 3. Landscape typology.

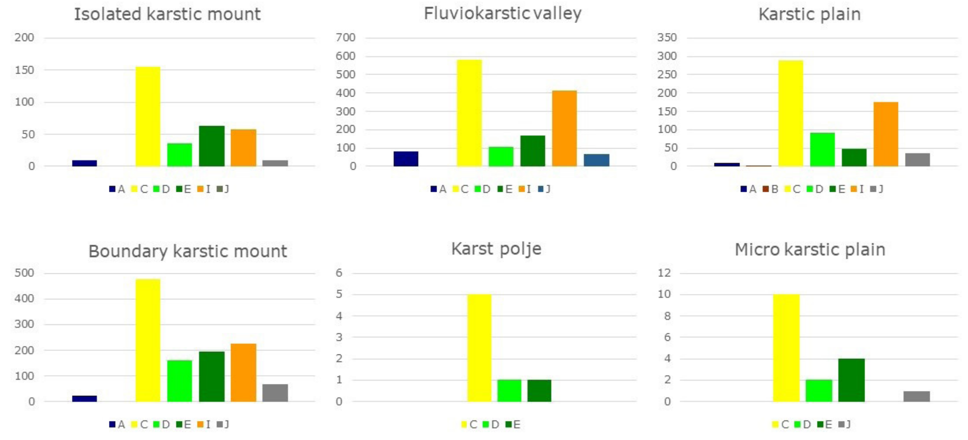

Fig. 4. Number of habitat types per landscape subtype.
Notes: A – Terrestrial surface waters and marshy habitats, B – Terrestrial areas without vegetation or poorly covered with vegetation, C – Grasslands, meadows, peat bogs and high grasses, D – Shrubs, E – Forest, I – Cultivated non-forest areas, weeds and ruderal vegetation, J – Built and industrial habitats.

Delineated landscape subtypes have some common characteristics regarding the habitat composition, which is closely connected to the land cover and abiotic components of environment. Natural habitat is a unique functional unit of terrestrial or water ecosystem determined by geographic, biotic, and abiotic features, independent regardless of whether it is natural or seminatural. All equivalent habitats are one habitat type (Croatian Nature Protection Act, Official Gazette of RC, No. 15/18, 14/19, Zagreb). Habitat types map is a good starting point for landscape analysis, its naturalness and changes measured by fragmentation (Bočić et al., 2018). Good indicator of landscape fragmentation is the number of habitat types patches in one landscape subtype and their ratio. In geographical term, landscape fragmentation is the division of larger homogeneous entities into smaller entities of different shapes, composition and structure. Due to the changes, those new forms are changing the composition, structure and function of the initial landscape (e.g., when the forest area is deforested and the so-altered area is occupied by agriculture areas, transport infrastructure, and settlements; Bočić et al., 2018; Buzjak et al., 2018). According to Non-Forest Terrestrial Habitat Map of Croatia, there are 7 habitat types present in the study area. Considering the absolute number of habitat type patches, *fluviokarstic valley* subtype has the largest number of patches. The smallest number of patches has landscape subtype *karst polje*, but this number is the result of the small area of this landscape subtype, which is the result of geomorphological conditions. Total area of *karst polje* is only 1.5 km^2, while *fluviokarstic valley* total area is 76.10 km^2. All subtypes have numerous patches of grasslands habitat type (C) (Fig. 4). The smallest landscape subtypes, *karst polje* and *micro karstic plain*, have only 3–4 habitat types present. Presence of numerous grassland patches is the evidence of anthropogenic changes of natural landscape since forests are the primary ecosystem type

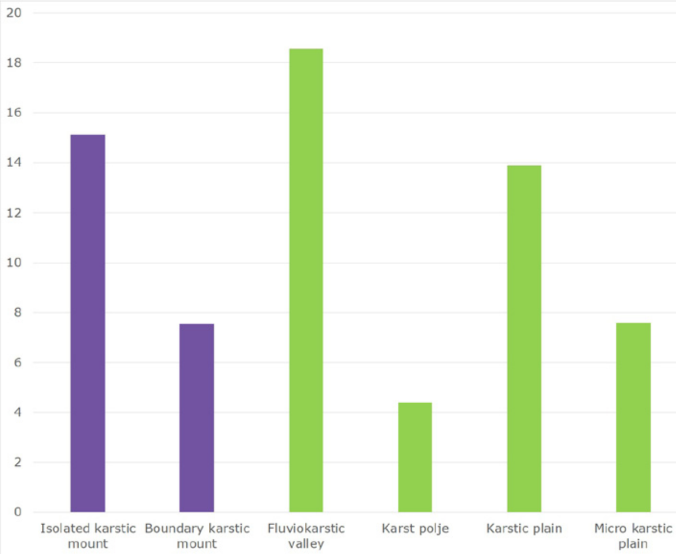

Fig. 5. Landscape Fragmentation Index (LFI).

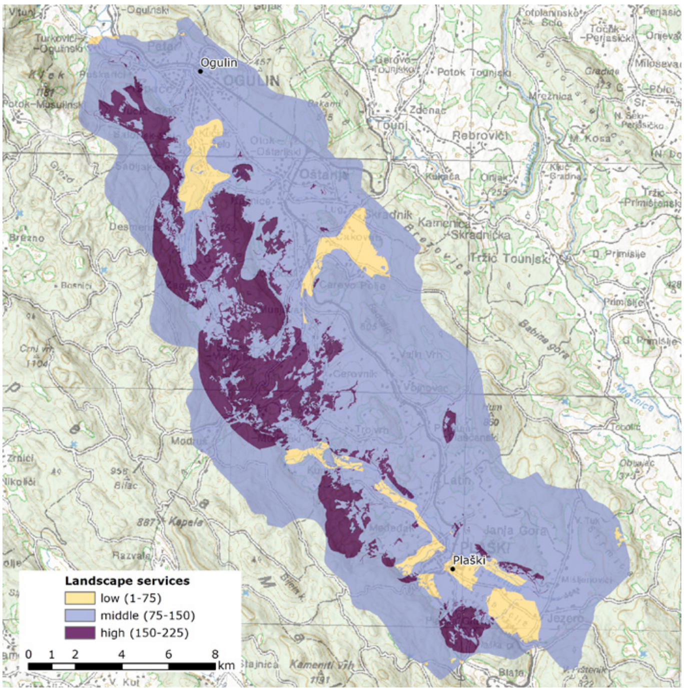

Fig. 6. Landscape services value in Ogulinsko-plašćanska Zavala.

for Croatia and this area (Trinajstić, 1998). Grasslands in Dinaric karst area relate to forest clearance for fuel and grazing (Kaligarič et al., 2006).

To examine the relation of the number of patches and area, a *Landscape Fragmentation Index* (*LFI*) was created. The LFI is expressed as the number of patches and total landscape subtype area ratio. The lowest value is recorded for *karst polje* and *boundary karstic mount* landscape subtypes, and the highest value is recorded for *fluviokarstic valley* and *isolated karstic mount* (Fig. 5). Low values of LFI are indicating low level of fragmentation, while higher values are indicating high level of fragmentation, which is connected to anthropogenic influenced landscape changes (urbanization, traffic network expansion, agriculture). As previously mentioned, such an index is a good indicator for landscape and habitat change characterized by landscape and habitat fragmentation.

The cumulative map shows landscape services values across study area (Fig. 6). Landscape services are classified into three categories. Low value class occupies 18.5% of the study area, middle class 77.7% and high value 16.2%. Areas with low value are distributed insular and are parts of macro-landscape type *planated relief*. The highest values are part of macro-landscape type *dissected relief*. High value of landscape services is present in landscape subtype *boundary karstic mount* with 24% of area with high landscape services value.

Considering Fig. 7, role of different landscape services in the whole landscape services value can be determined, as well as their spatial distribution. The high values coincide with the transition zones between the different landscape units. Provisioning services have the highest values in mountain units and in the transition zones from mountain to valley areas due to forests and the appearance of large karst springs. Socio-cultural services show a similar trend, with the highest values in the zones of high aesthetic and recreational value with diverse relief, karst phenomena and natural forests. For its regulating services the most interesting is the fluviokarst area, due to the drainage network, and transitional zones to karst with large caves and important biodiversity (Gottstein Matočec, 2002). This kind of information serves as a good basis for decision making in the environment impact assessment procedures or planning. From the other side, there is no solid basis to conclude that only "hotspot" areas are the ones in need of protection. For better justification and delineation of areas in high demand of protection, additional criteria should be applied, such as rareness of certain landscape service. Also, risk assessment of landscape services would serve as good additional information.

The same methodology was used to valorize landscape services and geodiversity occurrence, which enhances the possibility of comparison. Geodiversity assessment valorization criteria are listed in Table 2.

Geodiversity value is divided into 3 categories with equal intervals. The mean value of geodiversity is 43.8, which fits the class of middle geodiversity value. First class of low geodiversity value occupies only 0.26% of study area, which is why this class is barely visible in Fig. 8. Middle geodiversity value class occupies 52.1% of total area, while high geodiversity

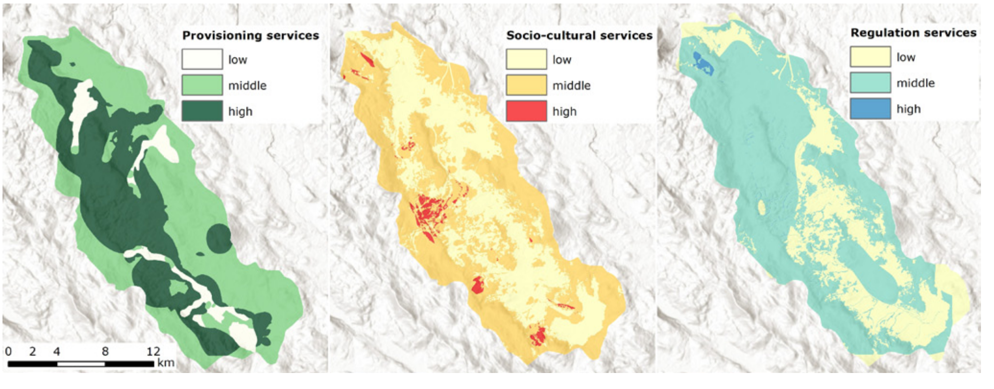

Fig. 7. Provisioning, Socio-cultural and Regulation services values in Ogulinsko-plašćanska Zavala.

T a b l e 2. Geodiversity valorization table.

Criteria	Criteria feature		Criteria feature categories	Points
Scientific value (100)	Lithology (25)	Carbonates	Presence	12.5
		Quaternary deposits	Presence	25
	Structure (25)	Fault density	0-0.3	8.3
			0.3-0.6	16.7
			0.6-1	25
	Landforms (25)	Dolines kernel density (5)	0-20	0.625
			20-40	1.25
			40-60	1.875
			60-80	2.5
			80-100	3.125
			100-120	3.75
			120-140	4.375
			140-160	5
		Spring caves (5)	Presence	5
		Caves (5)	Presence	5
		Springs Kernel density (5)	0-1	1.8
			1-2	3.4
			2-3.85	5
		Ponors (5)	Presence	5
	Morphometry (25)	Slope (8.3)	0-2	1.4
			2-5	2.78
			5-12	4.16
			12-32	5.54
			32-55	6.92
			>55	8.3
		Dissected relief (8.3)	0-5	1.66
			5-100	3.32
			100-300	4.98
			300-500	6.64
			500-800	8.3
		Hypsometry (8.3)	46-300	2.76
			300-600	5.54
			600-965	8.3

value class occupies 47.7%. As in the case of landscape services, the western mountainous part of the study area has higher geodiversity value. The highest values are recorded in the landscape subtype of *boundary karstic mount* (48% of total subtype area). The high values are also present in subtype of *fluviokarstic valley* (57% of total subtype area). Since whole study area was assessed by DEM as primary data source, landscape subtype *fluviokarstic valley* has a big percentage of high geodiversity value areas even though it is densely populated and consequently built. Spatial distribution and shape of geodiversity classes (Fig. 8) is at the most similar to spatial distribution and shape of provisioning services classes (Fig. 7).

Through this kind of geodiversity and abiotic values assessment, it is possible to detect loss of these values due to the spread of settlements and agriculture. It is also possible to detect areas in need of a protection.

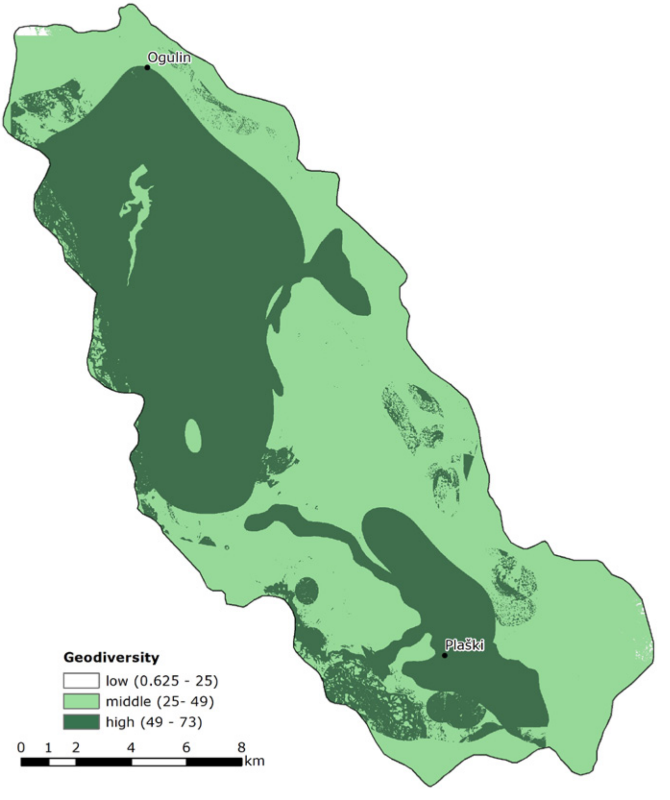

Fig. 8. Geodiversity value in Ogulinsko-plašćanska Zavala.

The primary structure of landscape was analyzed with the comparison of two indicators (landscape services and geodiversity), so it is possible to detect a role of abiotic part of environment in the determination of landscape types and values. But besides that, naturalness of vegetation in combination with geodiversity was very useful (bio)indicator for recreational and aesthetic value of socio-cultural landscape services. It was noted that areas with higher ratio of natural vegetation are located in the areas of highly dissected relief on limestones with high dolines density. Dolines are typical and most common karst features, so easy to measure and evaluate (Pahrenik, 2012). They also indicate areas with limitations for anthropogenic activities (like agriculture), mostly interesting in forestry, especially where limestone karst with bare surface or thin soils predominate. Dolines shape, depth and slopes inclination can also represent obstacle for activities both in rural and urban areas. On the other side, combined with vegetation, they can be used as indicator for more natural areas and landscapes with high level of aesthetic and recreational values due to the preserved nature. Dolines also indicate areas of high karst porosity where pollution can be quickly transferred to aquifers. In the case of spatial planning, these terrains are not suitable for activities with high risk potential that must be introduced with special care or completely avoided (waste disposal; use, storage and traffic of chemicals; sewage management, etc.).

Conclusion

Since it is fundamental for all other landscape structures (biogene and anthropogenic), it is necessary that primary (abiotic) landscape structure analysis represents the first step in landscape typology and assessment. In this study, geodiversity and landscape services were analyzed due to its importance for primary and other landscape structures. The relative relief evaluation method was chosen for quantitative assessment of both elements. When the similar methodology is used for analysis, it makes it easier to examine how they affect one another and what kind of interdependence are they in. For this study, provisioning, sociocultural, and regulation landscape services were selected.

Delineation of landscape types based on morphogenetic relief types produced landscape units suitable for further analysis. Their present state was analyzed according to the anthropogenic impacts measured by habitat fragmentation. Low values of Landscape Fragmentation Index were recorded in areas with limited sources for human activities, especially agriculture as a traditional economy for this area. Landscape types show conformity with landscape services and geodiversity values distribution. The dissected relief units have significantly high values, while planated relief units have lower values of landscape services and geodiversity. Concerning the analyzed landscape services, the most important are provisioning services since the high value category occupies the largest area.

Because of the conformity of indicators (geodiversity and landscape), landscape types delineated based on primary landscape structure represent good spatial units for planning, environment protection and sustainable management. The primary structure can be considered as the "original structure", or the "zero state" of the landscape. It is the base for other structures (secondary and tertiary), and in some way conditions, directs, and limits them. In the process of spatial planning or environmental protection, it is efficient to use clearly defined spatial units of homogeneous features relevant to other components. Their identification requires the analysis of synergic inner (inside primary structure) and outer (with secondary or other structures) relationships of the relevant interacting landscape elements (substrate, georelief, water, soil, and potential vegetation) and their properties.

References

Bahun, S. (1968). Geološka osnova hidrogeoloških odnosa krškog područja između Slunja i Vrbovskog. *Geološki Vjesnik,* 21, 19–82.

Bahun, S. (1970). Geološka osnova krške zavale Ogulin-Plaški. *Krš Jugoslavije,* 7(1), 1–20.

Bardi, A., Papini, P., Quaglino, E., Biondi, E., Topić, J., Milović, M., Pandža, M., Kaligarič, M., Oriolo, G., Roland, V., Batina, A. & Kirin T. (2016). *Karta prirodnih i poluprirodnih ne-šumskih kopnenih i slatkovodnih staništa Republike Hrvatske*. Zagreb: HAOP.

Bastian, O., Grunewald, K., Syrbe R.U., Walz, U. & Wende W. (2014). Landscape services: the concept and its practical relevance. *Landsc. Ecol.,* 29, 1463–1479. DOI: 10.1007/s10980-014-0064-5.

Birkhofer, K., Diehl, E., Andersson, J., Ekroos, J., Früh-Müller, A., Machnikowski, F., Mader, V.L., Nilsson, L., Sasaki, K., Rundlöf, M., Wolters, V. & Smith H.G. (2015). Ecosystem services – current challenges and opportunities for ecological research. *Frontiers in Ecology and Evolution*, 2, 1–12. DOI: 10.3389/fevo.2014.00087.

Bočić, N., Pahernik, M. & Maradin M. (2016). Temeljna geomorfološka obilježja Karlovačke županije. *Prirodoslovlje,* 16(1–2), 153–172.

Bočić, N., Buzjak, N., Čanjevac, I., Lukić, A., Opačić, V.T., Prelogović, V., Šulc, I., Zupanc, I. & Jašinski D. (2018). *Analiza i vrednovanje razvojnih potencijala i ograničenja područja urbane aglomeracije Zagreb*. Zagreb: Grad Zagreb, Gradski Ured za Strategjsko Planiranje i Razvoj Grada.

Bognar, A. (1990). Geomorfološke i inženjersko-geomorfološke osobine otoka Hvara i ekološko vrednovanje reljefa. *Hrvatski Geografski Glasnik*, 52, 49–65.

Bognar, A. (2001). Geomorfološka regionalizacija Republike Hrvatske. *Acta Geographica Croatica,* 34(1), 7–29.

Bognar, A. & Bognar H.I. (2010). Geoekološko vrednovanje reljefa Republike Hrvatske. In *Geoekologija - 21. vijek, teorijski i aplikativni zadaci* (pp. 44–65). Nikšić: Faculty of Philosophy Nikšić.

Bognar, A., Faivre, S., Buzjak, N., Pahernik, M. & Bočić N. (2012). Recent landform evolution in the dinaric and Pannonian Regions of Croatia. In D. Lóczy, M. Stankoviansky & A. Kotarba (Eds.), *Recent landform evolution* (pp. 313–334). Heidelberg, London, New York: Springer.

Bonacci, O. & Andrić I. (2010a). Hidrološka analiza krške rijeke Dobre. *Hrvatske Vode*, 18, 127–138.

Bonacci, O. & Andrić I. (2010b). Impact of an inter-basin water transfer and reservoir operation on a karst open streamflow hydrological regime: an example from the Dinaric karst (Croatia). *Hydrological Processes*, 24(26), 3852–3863. DOI: 10.1002/hyp.7817.

Bralić, I. (1999). Krajobrazno diferenciranje i vrednovanje s obzirom na prirodna obilježja). In N. Furlan-Zimmerman & M. Salaj (Eds.), *Krajolik – Sadržajna i metodska podloga Krajobrazne osnove Hrvatske* (pp. 101–109). Zagreb: Ministry of Spatial Planning, Construction and Housing & Faculty of Agriculture, University of Zagreb.

Brilha, J., Gray, M., Pereira, D.I. & Pereira P. (2018). Geodiversity: An integrative review as a contribution to the sustainable management of the whole of nature. *Environmental Science and Policy*, 86, 19–28. DOI: 10.1016/j.envsci.2018.05.001.

Buzjak, N. (2008). Geoekološko vrednovanje speleoloških pojava Žumberačke gore. *Hrvatski Geografski Glasnik*, 70(2), 73–89. DOI: 10.21861/hgg.2008.70.02.04.

Buzjak, N., Čanjevac, I., Vučković, I., Martinić, I. & Valožić L. (2018). Geoekološka analiza parka prirode i okolice Vranskog jezera u Dalmaciji. In *Hidrologija u službi zaštite i korištenja voda te smanjenja poplavnih rizika* (pp. 25–33). Brela: Hrvatsko Hidrološko Društvo.

Coratza, P., Reynad, E. & Zwolinski Z. (2018). Geodiversity and geoheritage: Crossing disciplines and approaches. *Geoheritage*, 10(4), 525–526. DOI: 10.1007/s12371-018-0333-9.

Croatian Nature Protection Act, Official Gazette of RC, No. 15/18, 14/19. Zagreb.

EUNIS (2019). *EUNIS habitat classification*. https://www.eea.europa.eu/data-and-maps/data/eunis-habitat-classification

Gottstein Matočec, S., Bakran-Petricioli, T., Bedek, J., Bukovec, D., Buzjak, S., Franičević, M., Jalžić, B., Kerovec, M., Kletečki, E., Kralj, J., Kružić, P., Kučinić, M., Kuhta, M., Matočec, N., Ozimec, R., Rađa, T., Štamol, V., Ternjej, I. & Tvrtković N. (2002). An overview of the cave and interstitial biota of Croatia. *Natura Croatica*, 11(Suppl. 1), 1–112.

Gray, M. (2005). Geodiversity and geoconservation: what, why, and how? *George Wright Forum*, 22, 4–12.

Gray, M. (2008). Geodiversity: developing the paradigm. *Proceedings Geologists Association,* 119, 287–298. DOI: 10.1016/S0016-7878(08)80307-0.

Gray, M. (2013). *Geodiversity. Valuing and conserving abiotic nature*. Chichester: Wiley.

Gray, M., Gordon, J.E. & Brown E.J. (2013). Geodiversity and the ecosystem approach: the contribution of geoscience in delivering integrated environmental management. *Proceedings Geologists Association,* 124, 659–673. DOI: 10.1016/j.pgeola.2013.01.003.

HAOP (2017). *Karta staništa*. http://www.haop.hr/hr/tematska-podrucja/prirodne-vrijednosti-stanje-i-ocuvanje/stanista-i-ekosustavi/stanista/karta

Kadykalo, A.N., López-Rodriguez, M.D., Ainscough, J., Droste, N., Ryu, H., Ávila-Flores, G., Le Clec'h, S., Muñoz, M.C., Nilsson, L., Rana, S., Sarkar, P., Sevecke, K.J. & Harmáčkov Z.V. (2019). Disentangling 'ecosystem services' and 'nature'scontributions to people. *Ecosystems and People*, 15(1), 269–287. DOI: 10.1080/26395916.2019.1669713.

Kaligarič, M., Culiberg, M. & Kramberger B. (2006). Recent vegetation history of the North Adriatic grasslands: Expansion and decay of an anthropogenic habitat. *Folia Geobot.*, 41, 241–258. DOI: 10.1007/BF02904940.

Kuhta, M. & Novosel A. (2001). Hydrogeology and cave explorations of the lost River Dobra: A case study of underground flow in the Dinaric karst. In *13th International Congress of Speleology* (pp. 181–186), 15-22 July, 2001. Brasilia: Union Internationale de Spéléologie.

Lele, S., Springate-Baginski, O., Lakerveld, R., Deb, D. & Dash P. (2013). Ecosystem services: Origins, contributions, pitfalls, and alternatives. *Conservation & Society*, 11, 343–358. DOI: 10.4103/0972-4923.125752.

Lozić, S. (1995). Vertikalna raščlanjensot reljefa kopnenog dijela Republike Hrvatske. *Acta Geographica Croatica*, 30, 17–28.

Mamut, M. (2010a). Primjena metode relativnog vrednovanja reljefa na primjeru otoka Rave (Hrvatska). *Naše More: Znanstveni Časopis za More i Pomorstvo*, 57, 260–271.

Mamut, M. (2010b). Geoekološko vrednovanje reljefa otoka Pašmana. *Geoadria,* 15(2), 241–267.

Marohnić-Kuzmanović, V., Matković, I. & Korlaet A. (Eds.) (2017). *Strategija prostornog razvoja Republike Hrvatske*. Zagreb: Hrvatski Zavod za Prostorni Razvoj.

Miklós, L., Kočická, E., Izakovičová, Z., Kočický, D., Špinerová, A., Diviaková, A. & Miklósová V. (2019): *Landscape as a geosystem*. Cham: Springer.

Mkrtchian, A. (2013). The relations between lands surface morphometry and spectral characteristics of ecosystems in the Ukrainian Carpathians. *Ekológia (Bratislava),* 32(1), 87–94. DOI: 10.2478/eko-2013-0007.

Obad Šćitaroci, M., Dumbović Bilušić, B., Bojanić Obad Šćitaroci, B. & Božić N. (2014). *Krajolik - čimbenik strategije prostornog uređenja*. Zagreb: Hrvatski Zavod za Prostorni Razvoj.

Pahernik, M. (2012). Prostorna gustoća ponikava na području Republike Hrvatske. *Hrvatski Geografski Glasnik,* 74(2), 5–26. DOI: 10.21861/HGG.2012.74.02.01.

Pătru-Stupariua, I., Stupariua, M.I., Stoicescua, I., Peringerd, A., Buttlerd, E. & Fürstg C. (2017). Integrating geo-biodiversity features in the analysis of landscape patterns. *Ecological Indicators,* 80, 363–375. DOI: 10.1016/j.ecolind.2017.05.010.

Pereira, P. & Pereira D.I. (2010). Methodological guidelines for geomorphosite assessment. *Géomorphologie: Relief, processes, environment*, 2, 215–222. DOI: 10.4000/geomorphologie.7942.

Portman, M.E. (2013). Ecosystem services in practice: Challenges to real world implementation of ecosystem services across multiple landscapes - A critical review. *Applied Geography*, 45, 185–192. DOI: 10.1016/j.apgeog.2013.09.011.

Reynard, E. & Brilha J. (2018). *Geoheritage: Assessment, protection and management*. Amsterdam: Elsevier.

Serrano, E. & Ruiz-Flaño P. (2007). Geodiversity. A theoretical and applied concept. *Geographica Helvetica*, 62(3), 140–147. DOI: 10.5194/gh-62-140-2007.

Swanwick, C. (2002). *Landscape character assessment, guidance for England and Scotland*. Edinburgh: The Countryside Agency, Scottish Natural Heritage.

Termorshuizen, J.W. & Opdam P. (2009). Landscape services as a bridge between landscape ecology and sustainable development. *Landsc. Ecol.*, 24(8), 1037–1052. DOI: 10.1007/s10980-008-9314-8.

Trinajstić, I. (1998). Fitogeografsko raščlanjenje klimazonalne šumske vegetacije Hrvatske. *Šumarski List,* 9(10), 407–421.

Vallés-Planells, M., Galiana, F. & Van Etvelde V. (2014). A Classification of landscape services to support local landscape planning. *Ecology and Society*, 19(1), 44. DOI: 10.5751/ES-06251-190144.

Velić, I. & Vlahović I. (2009). *Tumač geološke karte Republike Hrvatske 1:300 000*. Zagreb: Hrvatski Geološki Institut.

Zaušková, L. (2014). Landscape ecological interpretation of landscape survey results for optimal land use. *Ekológia (Bratislava),* 33(3), 252–258. DOI: 10.2478/eko-2014-0024.

Zwolinski, Z., Najwer, A. & Giardino M. (2018). Methods for assessing geodiversity. In E. Reynard & J. Brilha (Eds.), *Geoheritage: Assessment, protection and management* (pp. 27–52). Amsterdam: Elsevier. DOI: 10.1016/B978-0-12-809531-7.00002-2.

Žganec, K. (2012). The effects of water diversion and climate change on hydrological alteration and temperature regime of karst rivers in central Croatia. *Environ. Monit. Assess.*, 84, 5705–5723. DOI: 10.1007/s10661-011-2375-1.

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USING THE METHODOLOGICAL PROCEDURES FOR WATER EROSION RISK AREAS IDENTIFICATION FOR SUSTAINABLE LAND USE

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Abstract

Petlušová V., Petluš P., Tobiašová E., Hreško J.: Using the methodological procedures for water erosion risk areas identification for sustainable land use. Ekológia (Bratislava), Vol. 39, No. 2, p. 145–158, 2020.

The countries of the European Union have joined, inter alia, soil protection in the Common Agricultural Policy (hereinafter referred to as CAP). Accelerated soil erosion is a problem resulting from inappropriate land management, which affects both the presence of organic matter and the soil structure. The tool for elimination of negative impacts on soil can be its sustainable use. This requires the use of an accurate system to improve its condition. The first step should be problem identification and localisation. The research is aimed at the identification of water erosion risk areas by using selected methodological procedures. The research area was located at the intensively used hilly land of the Southwestern Slovakia. The digitisation of the manual interpretation of erosion risk areas with the use of aerial photos, erosion modelling, chemical analysis of soil organic matter (SOM) and analysis of soil structure were used. Verification was implemented via the field research with the use of the soil probes. Methods affirmed significant presence of the water erosion in the area. Efficient identification of erosional processes is possible via combination of presented methods by taking into consideration geological, geomorphological, pedological and geographical conditions and the use of the area over a longer period of time. The results of using methods that ensure accurate and effective localisation of erosion surfaces can be used for sustainable land use and its conservation.

Key words: digitization of the erosion surface, water erosion modelling, visual identification of the erosion, carbon sequestration and movement, hilly lowland.

Introduction

Sustainable management of natural resources has become an essential attribute in eliminating effects of climate change, preserving biodiversity and improving environment. Soil as a natural resource used primarily for agricultural purposes undergoes a process that threatens its quality and functions. It is endangered by erosion, compaction, loss of nutrients, increased acidity, low organic matter content and areal land take up for non-agricultural purposes. It leads to disruption or disappearance of ecosystems and important elements of the country. The application of sustainable and soil conservation innovative technologies of soil management can be ensured by lowering down the impact of negative effects.

The Common Agricultural Policy (CAP) has introduced the promotion of sustainable land use at the EU level. It supports soil management research and innovation. The aim is to identify and evaluate perspective land use systems that improve soils' condition and take into account the socio-economical aspect.

Following the principles and objectives of the CAP, the research is focused on the identification of water erosion using various methods. Testing of methods took place in the lowland type of hilly land.

In Slovakia, hills have suitable climatic and soil conditions for agricultural use. Gradual intensification of land use in Slovak hills that is related to the development of agricultural activities leads not only to the change of the landscape structure but mainly to the development of soil degradation. It promotes the acceleration of processes associated with surface run-off by formation and development of water erosion.

Accurate identification of areas endangered by water erosion is one of the fundamental attributes entering into sustainable land management proposals. Šarapatka and Niggli (2008) stated that if it needs to meet the requirements of sustainable land management, more soil drainage than the amount of soil that will be created should not be allowed to maintain the land for future generations. Soil drainage brings many negative phenomena. Bielek (2017) stated that the most important potentials of soil functions, especially biomass production, water filtration and hydrological rainfall cycle in the area, are reduced directly where the erosion took place, reducing the overall utility value of the soil habitat. Outside of the area where erosion takes place, there are threats such as unwanted supply of soil matter, pollutants, nutrients and agrochemicals and sediments damaging ecosystems. Thus, the erosive processes represent a significant negative factor threatening especially the surface soil horizon. Its removal is the ultimate and irreversible state that leads to decline in agricultural land yields and significant economic losses, which may not only have economic but also social consequences. Solution of this problem requires an active research on water erosion. Zachar (1970, 1982), Fulajtár and Janský (2001), Stankoviansky (2000, 2001), Kenderessy (2012) and Petlušová et al. (2016, 2017) dealt with the research of erosion in conditions of Slovakia. The development of water erosion is conditioned by morphological and morphometric features of the relief. Falťan et al. (2017), Panagos et al. (2015), Zeleňáková and Jakubíková (2010), Kirkby et al. (2002) and Styk (2002) highlighted the importance of the slope angle and land use. The importance of the erosion-accumulation processes is also highlighted by the extent of elaboration of this issue in many foreign scientific works that use innovative information systems. One of the dominant systems playing an important role in determining erosion-accumulation processes is remote sensing of the Earth (hereinafter referred to as RSE) (Biswas et al., 1999; Bouaziz et al., 2011; Al-Abadi et al., 2016). Research on erosion processes is progressively improving. New possibilities and procedures for identifying water erosion are being sought, which could be generalised and applied in practice. However, the choice of methods depends on the purpose of the research and may be thematically and territorially designed. The high variability of erosion research methods causes that most of works have a specific focus (pedological, geographical, hydrological, etc.). Signs of water erosion are not only quantitative but also qualitative. The identification of these changes based on the selected soil organic matter (SOM) parameters and soil structure reflects the manifestations of this form of erosion in the landscape as well as changes in the soil production capacity affected by water erosion (Pintaldi et al., 2018; Wang et al., 2018; Abera, Assen, 2019). The removal of particles from the humus surface horizon, the richest in organic matter, not only reduces the thickness of this active soil profile (Conforti et al., 2013) but also deteriorates other soil properties (Šarapatka et al., 2018) because they are closely related to the SOM. In this context, it is necessary to consider not only the reduction of organic carbon content but also its distribution in soil aggregates, which is closely related to carbon sequestration (Li et al., 2017). Soil structure and SOM together with soil texture are part of the soil erodibility factor, which is a part of the universal equation for the calculation of the average long-term soil loss (Wischmeier, Smith, 1978).

The aim is to present several procedures for the optimal identification of water erosion processes. The results can enter into sustainable land use proposals based on the precise and effective localisation of erosion risk areas that require a specific approach when they are used.

Material and methods

The model area is the intensively agriculturally used area of the SE part of Podunajská nížina lowland, which is formed by the spur of Hronská pahorkatina hills, in the cadastral areas of Ľubá and Belá villages. The current character of land use is the result of the intensification of agriculture with a predominant abundance of large-scale arable field. Predominant soil types are brown earth soils (Orthic Luvisols), regosols (Regosols) and chernozems. The area is part of Podunajská pahorkatina hills. It has the character of medium to slightly rugged hills with an altitude of 110–250 m asl. The selection of the area was conditioned by the presence of significant erosion processes.

Following methods were used to test methodologies for identifying water erosion processes:

Digitisation of erosion processes manifestations using aerial photos with the aim of finding area extension of water erosion

Spatial expansion of erosion and accumulation areas was realised by visual evaluation and interpretation of aerial photos, which are a proven possibility of their application in the field. Identification of erosion processes as colour differences was realised visually based on the aerial photos from six time periods (1949, 1970, 2006, 2011, 2014 and 2015). The images are from different seasons, thus partially eliminating the effect of the vegetation cover (visual expression of erosion partially overlapping seasonally). A surface assessment of the quantitative representation of erosion areas was obtained by vectorisation of light amoebic formations on aerial images. Significant areas where erosion took place in the past and is still going on or where erosion occurred only in the past or now were identified. They were named as actual and potential erosion risk areas (Fig. 1).

Modelling of water erosion

Modelling was realised as a calculation of the potential average annual loss of soil and the determination of the material removal from the slope. The digital terrain model 10×10 m was used, derived from the basic contour maps of 1:10 000. The rainfall erosivity factor (R factor) was identified from the climate map (Lapin et al., 2002) and a database of selected R factor values (Ilavská et al., 2005). To determine the soil erodibility factor (K factor), a database of selected K factor values was used (Ilavská et al., 2005). The calculation of the topographical factor, combined slope length and slope inclination (LS factor), was created in GRASS GIS (module r.watershed). The crop management factor (C factor) was used according

Fig. 1. Areas realistically and potentially endangered by water erosion (resource: orthophoto map © EUROSENSE).

to Malíšek (1992). The conservation practice factor (P factor) was determined accordingly (Ilavská et al., 2005).

For the determination of the 'average' annual loss of soil, the so-called universal equation for long-term loss of water by water erosion – USLE by Wischmeier and Smith (1978) – was used. The value of permissible loss of land serves to determine the erosion risk degree of the land. It is defined as a maximum intensity of soil erosion that allows to maintain sufficient level of the soil fertility on a long-term basis, which is economically available (Janeček et al., 2012). The categorisation of calculated data was based on four categories of average annual soil loss: no to weak soil loss (0–4 t/ha/year), medium soil loss (4–10 t/ha/year), high soil loss (10–30 t/ha/year) and extreme soil loss (>30 t/ha/year). They are listed in Act No. 220/2004 Coll. on the limit values for the erodibility categories of agricultural soils.

Modelling the drainage direction of the material

The determination of the direction of material removal from the slope was based on the methodology of Miklós et al. (1997). The methodology was adapted considering the assumption of transport of soil particles along the slope and not their accumulation. The relief shapes calculated in the geographic information system (hereinafter referred to as GIS) environment based on the digital relief model were used. The digital relief model was derived from the basic contour maps of the SR at a scale of 1:10 000. For vertical and horizontal ruggedness, the following categories have been determined: concave surfaces, sunken with slow-drainage, flat surfaces, non-curved and convex surfaces and convex surfaces with accelerated drainage. By the synthesis of vertical and horizontal curvature, the basic forms of the relief have been found to be critical in determining run-off ratios and for our needs to determine the potential material drainage from the slope. The basic forms of the relief together with the slope of the area were used in the calculation of material removal from slopes.

Identification and verification of water erosion with the use of soil probes

An Edelman auger with drill capacity of up to 500 cm with a diameter of 50 mm was used for the identification. The thickness of the humus horizon, the presence of soil horizons, the depth of turning the soil over, the character of the soil-forming substrate and the thickness of the accumulated material were verified. On the basis of the ascertained data, soils endangered by erosion (light surfaces) or soil accumulated in concave slope areas were identified as a manifestation of the transport of soil particles along the slope. There were 51 soil probes realised. Probes were located irregularly throughout the territory on slopes with expected erosion or accumulation (Fig. 2).

Soil erosion in the context of soil organic matter and soil structure analysis

In assessing the effects of water erosion through SOM and soil structure, several of their parameters, such as total organic carbon (TOC), labile (C_L) and non-labile (C_{NL}) carbon, humic substances (HS) and fractional composition of soil aggregates, were used. Soil structure vulnerability coefficient (K_v) (Valla et al., 2000), clogging index (I_c) (Lal, Shukla, 2004) or critical content of SOM (S_t) (Valla et al., 2000) were also used, on which effects of erosion-accumulation processes were fully manifested. At the same time, carbon sequestration was analysed. One of the main mechanisms of carbon sequestration is its increase in carbon-rich macro-aggregates (Six et al., 2002), which also implies their favourable size distribution. SOM analysis was performed on soil samples collected from three parts of the slope: the upper convex part, the slope and the accumulation concave part. On one slope (Fig. 2), soil samples were also taken in a line, from top to bottom along the slope. The sampling points were 20 m apart.

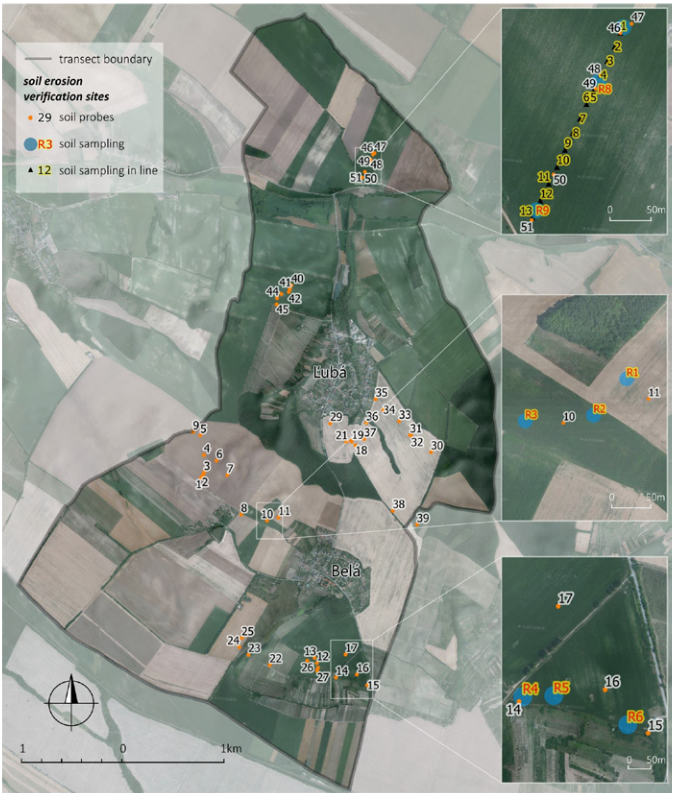

Fig. 2. Soil probes and sampling sites (resource: orthophoto map © EUROSENSE).

Results

Digitisation of erosion processes manifestations using aerial photos with the aim of finding area extension of water erosion

The area manifestation of water erosion in the period 1949–2016 increased by 130.56 ha (8.88%). The erosion areas have increased by changing land use because of the change of narrow strip fields to large-strip fields. Areas realistically and potentially endangered by water erosion represented 408.44 ha (27.78%) in total. These areas (Fig. 3) can be used in other procedures for the detection of water erosional processes.

Demarcated erosion surfaces were used in the process of partial syntheses, which express their relationships with the relief parameters (slope, exposure and relief shapes) and the type of land use change, and also with the values of soil removal or with material removal directions and so on (Petlušová et al., 2016). The slope is given as demonstration of research results. Areas where slope significantly affects or does not affect the spread of erosion have been identified via the combination of realistically endangered surfaces and slope. The slope was evaluated according to five categories. In the 0–1° category, actual erosion was abundant on 25.44 ha (16.9%). These are areas that are used to grow spring and winter cereals, oilseeds, leguminous seeds and the like. In the 1–3° category, erosion was on 66.44 ha (17.70%). In terms of land use, these sites were mainly used as arable land and also as permanent grassland. In the slope category 3–7°, erosion was on

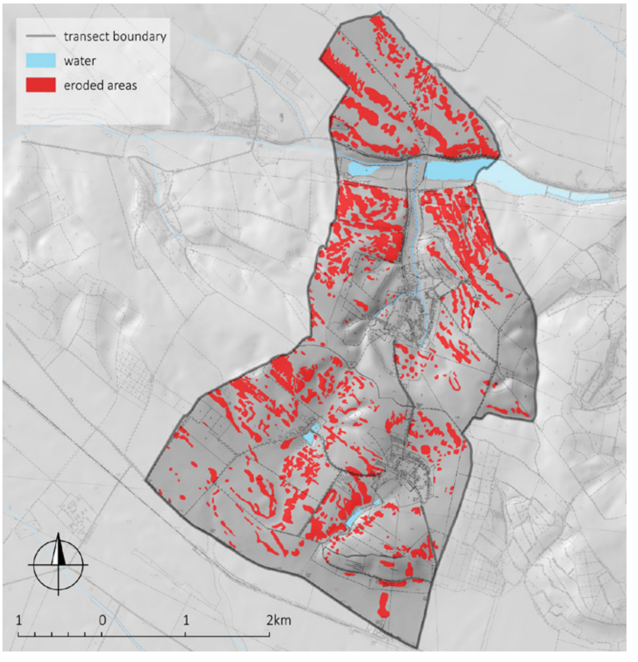

Fig. 3. Surfaces endangered by water erosion (resource: DTM 10x10, Esprit s. r. o; Basic map of SR 1: 10 000).

218.90 ha (30.27%). Major part of area comprised arable soil. Vineyards are considered to be a significant element. In the 7–12° category, erosion was found on 80.69 ha (39.83%). Suitability of soil types enables to use this area as arable soil despite the fact that the risk of erosion processes occurrence is high. Similarly, it is in the slope category of 12° or more, which was abundant on 4.22 ha (22.40%). The results show that the slope inclination determines the occurrence of areas with a realistic presence of soil erosion.

Modelling of water erosion

Water erosion modelling results in spatial expression of areas with potential average annual soil loss. Significant manifestations of erosion were found in moderately rugged hilly parts of the area with a slope of higher than 3°, but mostly where the slope of the area was 7–12°. In these parts, categories 2 and 3 (medium and high soil loss) prevail according to the data obtained in the soil loss calculations (Fig. 4). The area has long been used as large-scale intensively used arable fields. In one site, in category 3 with high soil loss, the forest is currently there.

Three categories of soil removal were identified: weak, medium strength and strong. On the greater part of the area, there are abundant categories of medium strong to strong drainage, which significantly determines the drainage of soil particles and their accumulation in the concave parts of the area. Research has shown that soil is subject to erosion processes. During the monitored period, the surface was intensively farmed. Rapeseed and winter wheat were grown, and on the neighbouring soil complex with medium soil loss, maize was grown.

Identification and verification of water erosion with the use of soil probes

Soil erosion identification was carried out with 51 drilled soil probes. Verification of erosional processes with the use of soil probes shows that predominantly soil types, regosols, eroded brown earths and eroded chernozems, indicate the presence of erosion in at least three quarters of cases (Table 1).

Soil probes also pointed out that in the lower accumulation slope parts and on the convex slope forms with a lower slope, the alluvial deposits (coluviosol-Colluvisols) were more than 2 m deep, indicating the downward movement of soil particles where they have been deposited.

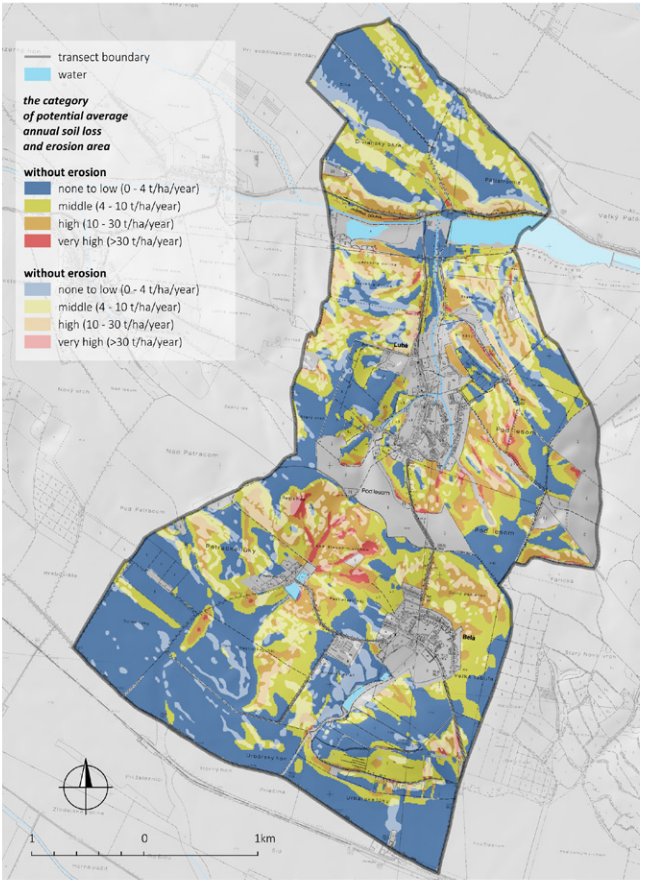

Fig. 4. Potential average annual soil loss (resource: DTM 10x10, Esprit s. r. o; Basic map of SR 1: 10 000; orthophoto map © EUROSENSE).

Soil erosion in the context of soil organic matter and soil structure analysis

TOC content in both soil (Table 2) and soil aggregates (Table 3) was increasing in the direction from the top of the slope down, but humus quality (Table 2) had opposite tendency.

In the case of C_L (Table 3) and C_{NL} (Table 3) contents in water-resistant macro-aggregates (WSA), the situation was similar, thus increasing its content down the slope direction. At the same time, organic carbon contents had a decreasing tendency from the larger fraction with its highest content to the smallest fraction of WSA with the lowest content. On the basis of carbon sequestration, the proportion of dry-sieved macro-aggregates (DSA; 1–7 mm) increased down the slope direction, and on the contrary, the content of the smaller (<1 mm) and larger (>7 mm) DSA fractions decreased (Table 3), which again points out not only the deterioration of the eroded top soil profile properties but, at the same time, similar the previous parameters, can also identify the erosion of the affected surfaces.

By linking the parameters that are part of the erodibility factor (SOM, soil structure and soil texture), further parameters that correspond to the manifestations of water erosion were obtained. Values of the soil structure vulnerability coefficient (K_v) (Valla et al., 2000) and index of crusting (I_c) (Lal, Shukla, 2004) were decreasing, and on the contrary, critical SOM content (S_t) (Valla et al., 2000) increased downward the slope (Table 4). On the basis of these

T a b l e 1. Soil types with erosion manifestation (E) and with no erosion manifestation (N).

Probe No.	**Soil Type Classification**	**Erosion Manifestation**
1	Cultivated Calcaric Regosols on neogene sediments	E
2	Cultivated Calcaric Regosols on neogene sediments	E
3	Cultivated Calcaric Regosols on neogene sediments	E
4	Eroded Orthic Luvisols on neogene sediments	E
5	Eroded Chernozems on neogene sediments	E
6	Calcaric Regosols on neogene sediments	E
7	Colluvisols	E
8	Eroded Chernozems on loess	E
9	Cultivated Calcaric Chernozems on loess	N
10	Colluvisols	E
11	Regosols on neogene sands	E
12	Cultivated Orthic Luvisols on loess loam	N
13	Regosols on weathered neogene gravels	E
14	Decalcified Haplic Chernozems on loess	E
15	Luvi-haplic Cernozems	E
16	Cultivated Calcaric Regosols on loess	E
17	Cultivated Calcaro Haplic Chernozems	N
18	Regosols on loess	E
19	Regosols on loess	E
20	Eroded Orthic Luvisols (rubified) on loess	E
21	Colluvisols	E
22	Colluvisols	E
23	Cultivated Calcaric Regosols on neogene sediments	E
24	Cultivated Calcaric Regosols on neogene sediments	E
25	Eroded Chernozems on loess	E
26	Cultivated Chernozems on loess	N
27	Eroded Chernozems on loess	E
28	Arenic Chernozems on neogene sands	N
29	Calcaric Regosols on loess	E
30	Haplic Chernozems on loess	N
31	Cultivated regosols on loess	E
32	Eroded Chernozems on loess	E
33	Eroded rubified soil (not classified)	N
34	Eroded Chernozems on loess	E
35	Colluvisols	E
36	Rubified soil on loess (not classified)	N
37	Haplic Chernozems on loess	N
38	Eroded Chernozems on loess	E
39	Haplic Chernozems on loess	N
40	Eroded Regosols	E
41	Cultivated regosols on loess	E
42	Eroded Regosols	E
43	Regosols on loess	E
44	Haplic Chernozems	N
45	Luvi-haplic Chernozems on loess	N
46	Eroded Chernozems on loess	E
47	Eroded Chernozems on loess	E
48	Regosols on loess	E
49	Eroded Chernozems on loess	E
50	Eroded Chernozems on loess	E
51	Colluvisols on neogene gravels	E

Resource: Terrain Research (2015–2017).

T a b l e 2. Selected quantitative and qualitative parameters of soil organic matter.

Part of the Slope/ Parameters	Total Organic Carbon (TOC)	Humic Substance Carbon (HSC)	Humic Acid Carbon (C_{HA})	Fulvic Acid Carbon (C_{FA})	C_{HA}/C_{FA} Ratio
	(mg kg^{-1})	(%)	(%)	(%)	
R1	7530	38.11	26.16	11.95	1.43
R2	10 810	39.59	26.64	12.95	1.15
R3	14 570	43.17	25.26	17.91	1.13
R4	16 980	37.99	23.79	14.19	1.70
R5	12 570	44.74	26.01	18.70	1.26
R6	18 900	70.06	40.10	30.95	1.89
R7	16 810	36.41	24.21	12.20	1.50
R8	9220	35.03	21.15	13.88	0.95
R9	13 480	37.09	21.14	15.95	1.01

Notes: R1, R4 and R7 – upper convex part; R2, R5 and R8 – slope; R3, R6 and R9 – accumulative concave part.

results, the least favourable state could be again considered on the top slope parts and basically with the downward trend.

The loss of soil structure and its tendency to erosion (S_t <5%) was manifested in all cases and had a decreasing tendency. The tendency of SOM in the line is shown in Table 5.

On the slope, the quality and quantity of soil organic matter are changing. Organic carbon is shifted to the bottom of the slope. This indicates soil erosion, which affects other important soil components and functions.

Discussion

Using selected methods for identifying water erosion has shown that water erosion processes can be identified by several methods. However, they cannot be generalised and unequivocally recommended for any assessed area. When selecting methods, it is necessary to start with the specific natural and anthropogenic conditions of a given area. The presented methods were used on the example of loess lowland hill land with the occurrence of Chernozems and haplic Luvisols. The use of several methods for detecting erosion processes increases the objectivity of claims about currently ongoing soil removal processes.

The advantage of using the digitisation of surface erosion manifestations using aerial photos is the possibility of creating a reference layer by vectorisation in the ArcGIS environment, over which spatial units are created by overlay method. The spatial units are also the basic operational units for further analyses, including detailed field research of erosion areas. The disadvantage is that the light amoebic formations in the images do not necessarily have to be erosion surfaces in all cases, and therefore, verification in the field is needed. It was not possible to accurately locate and identify erosional processes on areas with agricultural cultures during the research. More specifically, it was in the large-scale ploughed fields and vineyards. Large-scale fields were not covered with veg-

T a b l e 3. Selected carbon parameters in fractions of water-resistant macro-aggregates and their percent abundance

Part of the Slope/ Fraction Size	**>5 mm**	**>3–5 mm**	**>2–3 mm**	**>1–2 mm**	**>0.5–1 mm**	**>0.25–0.5 mm**	**<0.25 mm**
	(mg kg^{-1})						
	Total Organic Carbon (TOC)						
R1	10.990	8.880	8.860	11.270	10.980	8.090	5.580
R2	13.750	13.020	13.570	13.080	11.800	11.360	10.270
R3	21.010	17.580	18.380	19.460	17.950	16.040	15.210
R4	15.900	15.630	16.730	17.280	15.580	13.730	12.330
R5	13.960	14.280	14.430	14.730	14.180	11.650	9.250
R6	18.690	19.940	19.130	20.290	20.110	22.190	16.400
R7	16.050	17.240	17.000	17.240	17.640	16.880	13.630
R8	17.230	15.720	14.140	13.800	14.570	14.610	11.370
R9	9.540	11.190	10.970	12.150	11.900	10.590	7.950
	Labile Organic Carbon oxidised with $KMnO_4$ (C_L)						
R1	778	691	695	731	682	465	277
R2	1.203	1.186	1.070	1.124	1.118	911	730
R3	3.270	1.593	2.019	1.653	1.463	1.683	987
R4	1.622.2	1.656.1	1.480.3	1.646.5	1.763	1607.6	1.362.7
R5	949.9	1.009.1	902.9	1.029.5	950.1	936.1	670.6
R6	1.061.5	1.176.1	1.144.1	1.623.2	907.1	1784.5	1127
R7	1.354	1.711	1.781	1.935	1.754	2.108	2.150
R8	679	924	922	969	1.094	1.116	1.148
R9	617	734	923	1.102	874	1.027	1.092
	Non-labile Organic Carbon (NLC)						
R1	10.212	8.189	8.165	10.539	10.298	7.625	5.303
R2	12.547	11.834	12.500	11.956	10.682	10.449	9.540
R3	17.740	15,987	16.361	17.807	16.487	14.357	14.223
R4	14.277.8	13.973.9	15.249.7	15.633.5	13.817.0	12.122.4	10.967.3
R5	13.010.1	13.270.9	13.527.1	13.700.5	13.229.9	10.713.9	8.579.4
R6	17.628.5	18.763.9	17.985.9	18.666.8	19.202.9	20.405.5	15.273.0
R7	14.696	15.529	15.219	15.305	15.886	14.772	11.480
R8	16.551	14.796	13.218	12.831	13.476	13.494	10.222
R9	8.923	10.456	10.047	11.048	11.026	9.563	6.858
	Percentage abundance of dry-sieved macro-aggregates fractions						
Part of the Slope/ Fraction Size	**>7 mm**	**>5–7 mm**	**>3–5 mm**	**>1–3 mm**	**>0.5–1 mm**	**>0.25–0.5 mm**	**<0.25 mm**
	%						
R1	13.08	11.49	14.58	20.70	17.12	9.65	13.39
R2	12.36	16.55	18.23	22.82	13.74	7.27	9.04
R3	9.86	17.03	21.16	28.70	13.18	4.55	5.52
R4	47.03	11.84	12.32	14.89	7.99	2.95	2.99
R5	27.05	11.28	13.90	19.76	14.93	6.49	6.58
R6	25.74	7.69	8.54	9.61	5.11	3.20	40.12
R7	0.68	3.70	5.34	6.46	11.60	23.06	49.16
R8	0.80	1.32	3.20	3.48	6.74	22.34	62.12
R9	2.42	4.86	9.54	21.40	20.00	21.00	20.78

Notes: R1, R4 and R7 – upper convex part; R2, R5 and R8 – slope; R3, R6 and R9 – accumulative concave part.

T a b l e 4. Chosen parameters of the soil structure.

Part of the Slope/ Parameters	Soil Structure Vulnerability Coefficient (K_v)	Index of Crusting (I_c)	Critical Soil Organic Matter Content (S_t) (%)
R1	1.441	1.477	1.706
R2	1.343	1.011	2.911
R3	1.204	0.689	3.913
R4	2.319	3.94	0.819
R5	0.915	2.557	1.484
R6	1.054	3.809	1.007
R7	2.695	3.537	0.881
R8	3.346	2.019	1.849
R9	1.644	2.765	1.283

Notes: R1, R4 and R7 – upper convex part; R2, R5 and R8 – slope; R3, R6 and R9 – accumulative concave part.

T a b l e 5. Total organic carbon and selected quantitative and qualitative parameters of soil organic matter.

	Total Organic Carbon	Cold Water Extractable Organic Carbon	Hot Water Extractable Organic Carbon	Labile Organic Carbon	Humic Substance Carbon	Humic Acid Carbon (C_{HA})	Fulvic Acid Carbon (C_{FA})	C_{HA}/C_{FA} Ratio
		(mg kg^{-1})			(%)			
RL1	1.366	252.25	416.05	976	36.38	19.84	16.54	1.03
RL2	1.130	343.98	554.58	885	37.52	17.26	20.27	0.72
RL3	0.933	217.86	311.46	764	38.69	19.08	19.61	0.78
RL4	0.738	263.72	427.52	649	35.77	23.58	12.20	1.68
RL5	0.963	447.18	587.58	850	36.14	20.25	15.89	1.67
RL6	1.202	470.11	1218.91	825	36.19	21.71	14.48	1.70
RL7	1.217	412.78	822.28	986	36.89	18.57	18.32	1.60
RL8	1.337	217.86	592.26	1074	36.35	18.70	17.65	1.53
RL9	1.651	194.93	674.63	1008	35.61	21.26	14.35	1.09
RL10	1.378	114.66	301.86	1211	39.62	26.99	12.63	1.34
RL11	1.456	149.06	617.06	1301	24.24	23.42	13.32	1.37
RL12	1.592	149.06	500.06	1231	34.48	19.91	14.57	1.27
RL13	1.676	137.59	535.39	1289	32.76	19.93	12.83	1.64

Note: RL1 to RL13 are soil sampling localities on slopes.

etation during the mapping, and the newly established intensively used bleak vineyards uncovered erosional areas.

Modelling of water erosion is an effective method used in research. It is an effective way of protecting the soil by which the user regulates the management to minimise soil mass loss. Model results may not match the actual occurrence of erosion in space. This contention was confirmed in the synthesis of erosion endangered areas and the result of modelling of potential average annual soil loss (Fig. 4). The models also do not feature precise expression of the movement of the material along the slope, and they do not determine the exact location of the soil particles in the plot. It is not appropriate to use them for a shorter period than the season, respectively, to determine the soil loss from a single rainfall event or sudden snow heating.

Identification and verification of water erosion using soil probes is a relatively effective way of identifying real erosional processes. The advantage is also the understanding of landscape ecological, geomorphological connections and relationships in the landscape, which should be included in the proposals and principles for sustainable land use. However, using the method requires expertise and experience in field soil research. The method has its limitations, which result from real conditions in a given part of the growing season and conditions of soil state (optimal soil moisture, vegetation height, etc.).

When identifying erosional processes in the context of SOM and soil structure analysis, unlike other methods, not only quantitative but also qualitative changes in soil erosion endangered soil properties are determined. The difficulty of the method is that it requires precise analyses in laboratory conditions and field sampling is bound by the optimal current soil moisture.

It is not possible to determine which of the methods of soil loss detection used is more significant. The methods used are complementary. Identification of spatial distribution of erosion areas and modelling are helpful in quantitatively define erosion areas. The use of soil probes and analysis of SOM are helpful in verify qualitative changes in soil during the process of water erosion.

The contribution of the research is also the identification of erosional processes in relation to the dynamics of land use. On the basis of this, it can be assumed whether the development or the elimination of erosional processes or their elimination can potentially occur in the area. The importance of using a combination of various methods for research of erosional processes in the landscape is a suitable tool in a precise agriculture. Accurate localisation of endangered surface by erosion with the possibility of proposing an adequate soil erosion protection corresponds to the principles of the CAP. Farmers must respect Good Agricultural and Environmental Conditions (GAEC), which are linked to direct income support. Soil GAECs adopted for the European Union 2014–2020 planning period, designed to prevent soil erosion through appropriate measures, include standards that should be respected. However, the opposite has been shown in practice. Despite the fact that crops are being reduced in agriculture; soil, landscape, environment and other properties are damaged; and profit reduction, standards are rarely abided. The accurate detection of endangered surface by erosion will allow the farmer to invest less costs in soil protection than when trying to provide protection on a whole land block, which often leads to inefficient loss of energy and funds. It leads to an increase in the cost of the whole company. The costs of technical landscaping, infrastructure repair and water treatment are increasing.

Acknowledgements

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References

Abera, W. & Assen M. (2019). Dynamics of selected soil quality indicators in response to land use/cover and elevation variations in Wanka watershed, northwestern Ethiopian highlands. *Ekológia (Bratislava)*, 38(2), 126–139, 2019. DOI: 10.2478/eko-2019-0010.

Act No. 220/2004 Coll. on the protection and use of agricultural land and on the amendment of Act No. 245/2003 Coll. on Integrated Pollution Prevention and Control and on amendments to certain acts.

Al-Abadi, A.M.A., Ghalib, H.B. & Al-QurnawI W.S. (2016). Estimation of soil erosion in northern Kirkuk governorate, Iraq using RUSLE, Remote sensing and GIS. *Carpathian Journal of Earth and Environmental Sciences*, 11(1), 153–166.

Bielek, P. (2017). *Soil science (in Slovak.)* Nitra: SPU.

Biswas, S., Sudhakar, S. & Desai V.R. (1999). Prioritization of sub-watersheds based on morphometric analysis of drainage basin: a remote sensing and GIS approach. *Journal of the Indian Society of Remote Sensing*, 27(3), 155–166. DOI: 10.1007/BF02991569.

Bouaziz, M., Leidig, M. & Gloaguen R. (2011). Optimal parameter selection for qualitative regional erosion risk monitoring: A remote sensing study of SE Ethiopia. *Geoscience Frontiers,* 2(2), 237–142. DOI: 10.1016/j.gsf.2011.03.004.

Conforti, M., Buttafuoco, G., Leone, A.P., Aucelli, P.P.C., Robustelli, G. & Scarciglia F. (2013). Studying the relationship between water-induced soil erosion and soil organic matter using Vis–NIR spectroscopy and geomorphological analysis: A case study in southern Italy. *Catena*, 110, 44–58. DOI: 10.1016/j.catena.2013.06.013.

Falťan, V., Krajčírovičová, L., Petrovič, F. & Khun M. (2017). Detailed geoecological research of terroir with the focus on georelief and soil – a case study of Krátke Kesy vineyards. *Ekológia (Bratislava)*, 36(3), 214–225. DOI: 10.1515/eko-2017-0018.

Fulajtár, E. & Janský L. (2001). *Soil water erosion and erosion protection (in Slovak).* Bratislava: VÚPaOP.

Ilavská, B., Jambor, P. & Lazúr R. (2005). *Identification of endangered to soil quality by water and wind erosion and proposed measures (in Slovak)*. Bratislava: VÚPaOP.

Janeček, M., Dostál, T., Dufková, K.J., Dumbrovský, M., Hůla, J., Kadlec, V., Konečná, J., Kovář, P., Krása, J., Kubátová, E., Kobzová, D., Kudrnáčová, M., Novotný, I., Podhrázská, J., Pražan, J., Procházková, E., Středová, H., Toman, F., Vopravil, J. & Vlasák J. (2012). *Agricultural land erosion protection. Methodics (in Czech).* Praha: Česká zemědělská univerzita.

Kenderessy, P. (2012). Soil loss assessment in an agricultural landscape and its utilization in landscape planning. *Ekológia (Bratislava)*, 31(3), 309–321. DOI: 10.4149/ekol_2012_03_309.

Kirkby, M., Bracken, L. & Reaney S. (2002). The influence of land use, soils and topography on the delivery of hillslope runoff to channels in SE Spain. *Earth Surface Processes and Landforms,* 27, 1459– 1473. DOI: 10.1002/esp.441.

Lal, R. & Shukla M.K. (2004). *Principles of soil physics*. New York: CRC Press.

Lapin, M., Faško, P., Melo, M., Šťastný, P. & Tomlain, J. (2002). Climatic regions (in Slovak). In *Atlas krajiny SR* (pp. 95). Bratislava: MŽP SR, Banská Bystrica: SAŽP.

Li, Z., Liu, CH., Dong, Y., Chang, X., Nie, X., Liu, L., Xiao, H., Lu, Y. & Zeng G. (2017). Response of soil organic carbon and nitrogen stocks to soil erosion and land use types in the Loess hilly–gully region of China. *Soil Tillage Res.*, 166, 1–9. DOI: 10.1016/j.still.2016.10.004.

Malíšek, A. (1992). Optimal slope length depending on water erosion (in Slovak). In *Vedecké práce* (pp. 201–220). Bratislava: VÚPaOP.

Miklós, L., Krcho, J., Hrnčiarová, T., Matečný, I. & Kozová M. (1997). *Morphometric relief indicators and their usefulness of interpretation for planning processes (in Slovak)*. Bratislava: ÚKE SAV.

Panagos, P., Borrelli, P., Poesen, J., Ballabio, C., Lugato, E., Meusburger, K., Montanarella, L. & Alewell C. (2015). The new assessment of soil loss by water erosion in Europe. *Environmental Science & Policy*, 54, 438–447. DOI: 10.1016/j.envsci.2015.08.010.

Petlušová, V., Petluš, P. & Hreško J. (2016). *Identification of water erosion processes in agricultural land (in Slovak).* Nitra: UKF.

Petlušová, V., Petluš, P. & Hreško J. (2017). The effect of land use changes on soil erosion in the areas of Lubá and Belá (Hronská pahorkatina upland). *Geografický Časopis,* 69(3), 245–262.

Pintaldi, E., D'amico, M.E., Stanchi, S., Catoni, M., Freppaz, M. & Bonifacio E. (2018). Humus forms affect soil susceptibility to water erosion in the Western Italian Alps. *Applied Soil Ecology*, 123, 478–483. DOI: 10.1016/j.apsoil.2017.04.007.

Six, J., Callewaert, P., Lenders, S., De Gryze, S., Morris, S.J., Gregorich, E.G., Paul, E.A. & Paustian K. (2002). Measuring and understanding carbon storage in afforested soils by physical fractionation. *Soil Sci. Soc. Am. J.*, 66(6), 1981–1987. DOI: 10.2136/sssaj2002.1981.

Stankoviansky, M. (2000). Differentiated geomorphic effect of gully erosion due to large scale land use changes. In D. Balteanu, M. Ielenicz & N. Popescu (Eds.), *Geomorphology of the Carpatho-Balcan Region*. Bucurest: Drobeta Turnu Severin.

Stankoviansky, M. (2001). Tillage erosion and its geomorphic effect with special regard to the Myjava-White Carpathian kopanitse area (in Slovak). *Geografický Časopis*, 53(2), 95–110.

Styk, J. (2002). Monitoring the impact of water erosion on changes in selected soil properties (in Slovak). In J. Kobza (Ed.), *Monitoring pôd SR. Aktuálny stav a vývoj monitoringu pôd. Výsledky z Čiastkového monitorovacieho systému – Pôda za obdobie 1997–2001*. Bratislava: VÚPaOP.

Šarapatka, B. & Niggli U. (2008). *Agriculture and landscape: The paths to alignment (in Czech)*. Olomouc: Univerzita Palackého.

Šarapatka, B., Cap, L. & Bila P. (2018). The varying effect of water erosion on chemical and biochemical soil properties in different parts of Chernozem slopes. *Geoderma,* 314, 20–26. DOI: 10.1016/j.geoderma.2017.10.037.

Valla, M., Kozák, J. & Ondráček V. (2000). Vulnerability of aggregates separated from selected anthorsols developed on reclaimed dumpsites. *Rostlinná Výroba,* 46, 563–568.

Wang, Y., Ran, L., Fang, N. & Shi Z. (2018). Aggregate stability and associated organic carbon and nitrogen as affected by soil erosion and vegetation rehabilitation on the Loess Plateau. *Catena,* 167, 257–265. DOI: 10.1016/j.catena.2018.05.005.

Wischmeier, W.H. & Smith D.D. (1978). *Predicting rainfall erosion losses*. Maryland: SEA USDA Hystaville.

Zachar, D. (1970). *Soil erosion (in Slovak)*. Bratislava: SAV.

Zachar, D. (1982): *Soil erosion*. New York, Amsterdam, Oxford: Elsevier.

Zeleňáková, M. & Jakubíková A. (2010). Modeling of erosion and transport processes. *Ekológia (Bratislava)*, 29(1), 87–98. DOI: 10.4149/ekol_2010_01_87.

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LAND COVER AND LAND USE CHANGE-DRIVEN DYNAMICS OF SOIL ORGANIC CARBON IN NORTH-EAST SLOVAKIAN CROPLANDS AND GRASSLANDS BETWEEN 1970 AND 2013

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Abstract

Skalský R., Koco Š., Barančíková G., Tarasovičová Z., Halas J., Koleda P., Makovníková J., Gutteková M., Tobiášová E., Gömöryová E., Takáč J.: Land cover and land use change-driven dynamics of soil organic carbon in North-East Slovakian croplands and grasslands between 1970 and 2013. Ekológia (Bratislava), Vol. 39, No. 2, p. 159–173, 2020.

Soil organic carbon (SOC) in agricultural land forms part of the global terrestrial carbon cycle and it affects atmospheric carbon dioxide balance. SOC is sensitive to local agricultural management practices that sum up into regional SOC storage dynamics. Understanding regional carbon emission and sequestration trends is, therefore, important in formulating and implementing climate change adaptation and mitigation policies. In this study, the estimation of SOC stock and regional storage dynamics in the *Ondavská Vrchovina* region (North-Eastern Slovakia) cropland and grassland topsoil between 1970 and 2013 was performed with the RothC model and gridded spatial data on weather, initial SOC stock and historical land cover and land use changes. Initial SOC stock in the 0.3-m topsoil layer was estimated at 38.4 t ha^{-1} in 1970. The 2013 simulated value was 49.2 t ha^{-1}, and the 1993–2013 simulated SOC stock values were within the measured data range. The total SOC storage in the study area, cropland and grassland areas, was 4.21 Mt in 1970 and 5.16 Mt in 2013, and this 0.95 Mt net SOC gain was attributed to interconversions of cropland and grassland areas between 1970 and 2013, which caused different organic carbon inputs to the soil during the simulation period with a strong effect on SOC stock temporal dynamics.

Key words: soil organic carbon stock, regional soil organic carbon storage, soil organic carbon modelling, RothC, legacy soil data.

Introduction

Soil organic carbon (SOC) is one of the most important soil constituents, and it affects many soil and ecosystem processes. These include soil physical structure, its fertility and water-holding capacity (Vopravil et al., 2014; Campbell, Paustian, 2015). Moreover, Banwart et al. (2015) considered that SOC has a key role in all soil ecosystem services because not only it is sensitive to climate change, as recorded by Smith et al. (2005), but also it has both direct and indirect impacts from agricultural management practices (Janzen, 2006) and land use changes (Minasny et al., 2014; Vopravil et al., 2014).

The direction of SOC quantity and quality changes in the most anthropogenic affected ecosystems is controlled by their historical and actual use, and it is estimated that soil cultivation, and especially conversion of grassland to croplands, leads to significant SOC losses. In the overall global balance, Janzen (2006) confirmed that this can be as high as 50Pg carbon emitted because of land conversion and Guo and Gifford's (2002) meta-analysis estimated that SOC stock loss in converting grassland to cropland is up to 59%.

Crop and soil management are additional important factors influencing SOC stock in agricultural areas because these are intimately connected with particular land use. The low input of organic carbon from crop residues and insufficient supply of high-quality organic fertilisers can increase SOC mineralisation and decrease its overall stock in intensively managed croplands (Capriel, 2013). Conversely, many agricultural practices can offset the negative impacts of intensive cultivation and provide opportunity for long-term sequestration of organic carbon in intensively used cropland soils (Janzen, 2006; Lehtinen, 2014; Barančíková et al., 2014). These include the application of manure or compost, optimal crop rotation, minimum soil tillage or a lack of till technology, proper water management and soil protection technology, which includes mixing crop residues with soil and the subsequent application of organic fertilisers.

The combination of on-site effects of local management strategies and farmers' decisions on topsoil SOC dynamics translates into regional SOC storage dynamics, and understanding SOC dynamics on the larger regional, national and global scales is the most important in estimating the carbon emission and sequestration trends associated with agricultural practices (Smith et al., 2005). This knowledge focuses on SOC management, and it provides possible input for evaluating existing policies and formulating new ones that adapt to climate change and mitigate its impacts with agricultural practices (Frank et al., 2015).

Although information on the large-scale SOC stock temporal dynamics driven by land cover land use changes is limited by available data with appropriate spatial and temporal resolution, the process-based models combined with spatially explicit quantitative data on topsoil SOC, weather and land use prove effective in overcoming this problem (as in Barančíková et al., 2010; Alvaro-Fuentes et al., 2011; Gottshalk et al., 2012; Barančíková et al., 2012; Ma et al., 2016; Kaczynski et al., 2017).

For this purpose, the RothC turnover model (Coleman, Jenkinson, 2014) is most commonly used in SOC management studies because it can be run on single-site basis or integrated with geographical data sets to provide spatially explicit estimates for regions, countries and the world (Falloon, Smith, 2012; Campbell, Paustian, 2015; Gottshalk et al., 2012; Wang et al., 2016; van Wesemael et al., 2010; Barančíková et al., 2013).

This model was successfully used in reconstructing the 1970–2010 Slovak crop and grasslands topsoil SOC stock development trajectory and also for estimating the current SOC levels on a national scale (Barančíková et al., 2010, 2012, 2013). However, these authors record that the results provide only approximate SOC stock estimates because of limitations in the spatial resolution of gridded data (10x10 km) on organic carbon inputs from management, the monthly weather records and the initial 1970's SOC stock estimated from the soil map and profile data from the Slovak National agricultural soils inventory used to run the RothC model. This research also neglects marginal cropland and grassland areas and does not consider land cover change during the simulation period (Barančíková et al., 2012, 2013).

This study focuses on the *Ondavská Vrchovina* highland region located in the West Carpathian mountain range in North-East Slovakia. The study region has experienced significant land cover and land use changes over the past 50 years, including conversions of grasslands and croplands; and it is, therefore, a good example of these changes in Slovak sub-mountain and inner-mountain basin areas. Here, we used the RothC model to estimate the 1970–2013 SOC stock and regional SOC storage dynamics in cropland and grassland in *Ondavská Vrchovina* region between 1970 and 2013 as affected by cropland and grassland conversions and associated crop management.

Materials and methods

Study area

The 3,129 km^2 *Ondavská Vrchovina* highlands form part of the West Carpathian Mountains in North-eastern Slovakia (Fig. 1). The study area altitude ranges from 114 m in the southernmost part to 1,090 m in the north, with a

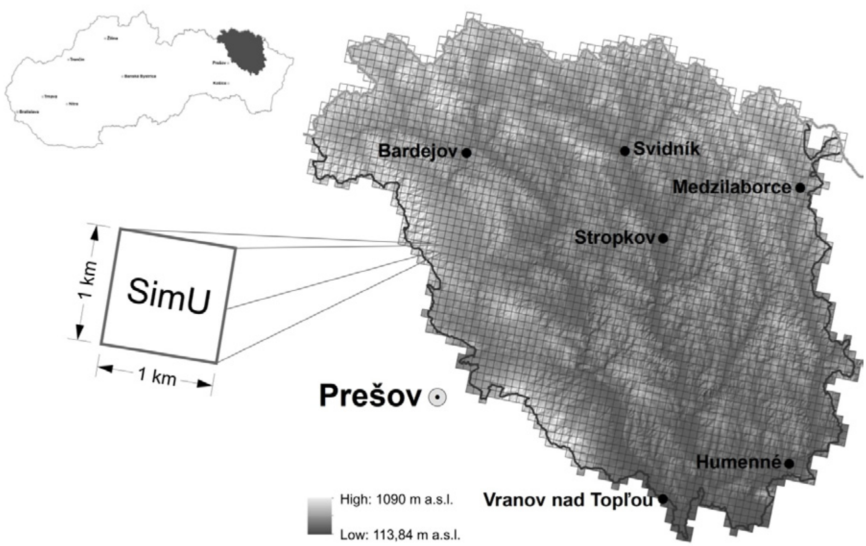

Fig. 1. Location and topography of *Ondavská Vrchovina* highland region, and spatial simulation units (SimU) for running RothC model.

mean of 360 m. The mean annual temperature ranges from 7 to 8 °C. July is the warmest with 22.8–23.8 °C, January is the coldest with −3.6 to 4.3 °C, and the annual rainfall ranges from 600–650 mm in lower regions to 650–700 mm in higher mountain areas. *Dystric* and *Eutric Cambisols* developed from tertiary sedimentary sandstones and siltstones dominate the mountain positions, whereas the lower mountain foot has gently sloping topography with *Stagnic Cambisols*, *Stagnic Luvisols* and *Stagnosols* from quaternary unconsolidated slope sediments. *Fluvisols* are then located in the floodplains. The texture of soils is mostly medium, locally coarse or fine. Stone content is higher in *Cambisols* and lower or absent in *Luvisols* and *Stagnosols*. The study area comprises broadleaf and mixed forests in the higher mountains areas and grass and cropland at lower elevations.

RothC model

The Rothc-26.3 model (Coleman, Jenkinson, 2014) determines the turnover of organic carbon in non-waterlogged soils. The SOC turnover process is affected by weather, plant cover, soil type and organic carbon input. The model used a monthly time step to calculate carbon turnover in the soil, and it can operate on annual to century time scales. The model splits total SOC pool into four active compartments (decomposable plant material, resistant plant material, microbial biomass and hummified organic matter) and a small amount of inert organic matter.

The model requires data on weather (monthly rainfall [mm], monthly open-pan evaporation [mm], average monthly mean air temperature [°C]), soil (clay content [%], content of inert organic carbon [IOM, %], soil-layer depth [cm]), land use and management (soil cover presence, monthly input of plant residues [t C ha^{-1}] and the monthly input of farmyard manure [t C ha^{-1}]).

Weather data

This was obtained from three WMO (World Meteorological Organization) international meteorological stations located directly in the study area (WMO 11976 – *Stropkov-Tisinec* and WMO 11977 – *Medzilaborce*) or close to it (WMO 11993 – *Kamenica nad Cirochou*). Daily temperature records (°C) and sum of rainfall (mm) for each year of the simulation period (1970–2013) were then aggregated to the monthly mean values. This was complemented by daily potential evapotranspiration [mm] estimated using the *Penman-Monteith* method and summated to monthly values to replace the open-pan evaporation data required by the RothC model.

Land cover and land use data

There is no consistent study area land cover product available for the entire 1970–2013 period, so multiple data resources were used to reconstruct land cover dynamics. Polygon data from the national Land Parcel Identification System (LPIS) was the basic source used to identify agricultural parcel borders and cropland and grassland areas in 2003 and 2013.

The Landsat 4, 5 and 7 imageries at 30-m spatial resolution were interpreted in LPIS agricultural area borders, and this yielded crop and grassland area estimates for 1986 and 1994. Content from the 1965 military topography maps in 1:10,000 scale was digitised, and the interpreted map legend provided initial 1970 crop and grassland areas.

Historical land use data for Slovakia are mostly limited to national agricultural statistics for harvested crop areas, crop yields and the number of animals and their manure products at national, regional (NUTS3) and district (NUTS4) levels. District-level statistics on harvested crop areas were used to reconstruct annual crop shares for the period before 2003. LPIS data and farmer's declarations for each registered agricultural parcel provided the first official crop area information after 2003. The annual district-level statistics on crop yields, the annual crop share data and national carbon conversion coefficients recommended for SOC balanced crop production in Slovakia by Bielek and Jurčová (2010) enabled organic carbon inputs calculation from plant residues [t C ha^{-1}] separately for the cropland and grassland areas. The annual farmyard manure inputs were calculated from the district-level statistics on farmyard manure application rates and the number of animals per unit area. More precise estimates of animal distribution were possible after 2003 from the National Veterinary GIS farm-level data on animal number and type. Cropland- and grassland-specific organic carbon inputs from farmyard manure [t C ha^{-1}] were then estimated using the carbon conversion coefficients by Bielek and Jurčová (2010).

Initial SOC stock

The dominant soil typological unit of soil type, soil texture class and stone content was assigned to each agricultural parcel for both cropland and grassland from 1:10,000 polygon soil maps (Němeček et al., 1967; Hraško, Bedrna, 1970), and topsoil SOC concentrations [%] were allocated from the closest measured soil profile (Linkeš et al., 1988) for the same soil typological unit and land cover class (Table 1).

T a b l e 1. Topsoil (0–30 cm) SOC concentration [%] in pre-stratified measured soil profiles used as an input for calculating initial topsoil SOC stock for the pilot area.

Soil/land cover class	**Min**	**Max**	**Mean**	**Standard deviation**	**Median**	**Quartile**		**Profile count**
						Lower	**Upper**	
HM	0.45	1.73	0.85	0.21	0.84	0.71	0.96	222
RA	0.48	1.95	0.99	0.35	0.91	0.74	1.18	62
KM_C	0.40	3.47	1.02	0.36	0.94	0.76	1.19	554
KM_G	0.43	3.02	1.12	0.47	1.03	0.79	1.30	209
FM_C	0.44	3.05	1.22	0.44	1.15	0.94	1.42	159
FM_G	0.48	3.06	1.39	0.50	1.32	1.02	1.66	84
CA_C	0.71	2.59	1.44	0.44	1.35	1.20	1.68	18

Notes: HM – Albic/Stagnic Luvisols; RA – Rendzic Phaeozem; CA_C – Endogleyic Phaeozem/Mollic Fluvisol on cropland; KM_C – Cambisols on cropland; KM_G – Cambisols on grassland; FM_C – Fluvisols on cropland; FM_G – Fluvisols on grassland.

The topsoil clay content for each agricultural parcel was calculated as the weighted mean from the 20-m spatial resolution raster of topsoil clay content previously derived from measured soil profile data (Balkovič et al., 2010). The stone content was then taken directly from the soil map. However, the bulk density values (t ha^{-1}) equired for SOC stock calculation were not available in the original data, and these had to be estimated by regional pedotransfer function for Slovakia (Makovníková, Širáň, 2011):

$$BD_{ij} = 1.52644 + 0.0000517149 * CLAY_{ij} - 0.107002 * COX_{ij} \qquad (1)$$

where *BD* is the soil bulk density (g cm^{-3}), CLAY is the topsoil clay content (%), *COX* is the topsoil SOC concentration (%), *i* is the i-th agricultural parcel and *j* is either cropland or grassland. The SOC stock for each parcel was then calculated as

$$SOC_{ij} = (BD_{ij} * COX_{ij} * (30 * (1 - \frac{SC_{ij}}{100})) \qquad (2)$$

where *SOC* is the SOC stock in 0–30 cm depth (t ha^{-1}), *BD* is the soil bulk density (g cm^{-3}), *COX* is the topsoil SOC concentration (%), *SC* is the topsoil stone content (%), *i* is the i-th agricultural parcel and *j* is either cropland or grassland.

All estimated SOC stock values were relevant for 1970; and the area-weighted mean was calculated from data on the individual parcels to provide initial SOC stock values for cropland and grassland areas within the 1x1 km spatial resolution grid cells (Fig. 1).

The RothC model requires initial SOC stock to be pooled in five compartments before SOC balance is calculated. Therefore, simulations with in-built model equilibrium mode were performed to obtain the SOC content in the different compartments, and the model was run by applying pedotransfer estimation (Falloon et al., 1998) of inert SOC compartment values necessary to run the model:

$$IOM = 0.049 * SOCj^{1.139} \qquad (3)$$

where *IOM* is the inert organic carbon (t ha^{-1}), *SOC* is the initial SOC stock (t ha^{-1}) estimated for 1970 and *j* is either cropland or grassland.

In addition to compartmental SOC content, the RothC model also provides balanced organic carbon input values to the soil, which can keep initial SOC stock values unchanged in constant climate over thousands of years. These inputs arise as outputs from the model equilibrium runs.

Model setup and SOC stock simulations

Simulation units (SimU) for running the RothC model were derived from individual cells of the 1x1 km spatial resolution grid for the entire study area (Fig. 1). These SimU provided geographical reference for inputs and outputs of the RothC model and grassland and cropland areas (ha) for 1970, 1986, 1994, 2003 and 2013; and the inputs included monthly weather and land-cover-specific initial SOC stock, organic carbon input.

The precise timing of individual land conversion events in the pilot area between 1970 and 2013 was assumed to be unknown, and therefore, a set of theoretical scenarios on cropland to grassland and/or grassland to cropland conversions was prepared for each SimU to obtain theoretically possible SOC stock change trajectories. The scenarios were set to provide for both land cover change occurring at approximately 10-year intervals, thus corresponding to available land cover data from the pilot area and also instances of no assumed land conversion (Table 2).

The SOC stock dynamics for each SimU and theoretical land cover change scenario in Table 2 were simulated between 1970 and 2013 with either (1) annual organic carbon inputs from plant residues and farmyard manure reconstructed for the corresponding simulation period or (2) balanced organic carbon inputs to the soil from the equilibrium RothC runs. These balanced inputs provided reference SOC stock estimates without assuming impacts from land use.

Land-cover-specific SOC stock (t ha^{-1}) for each SimU and each year was calculated as the weighted average using the SimU scenario area as the weight. Table 2 highlights that the land cover class was assigned with an estimated area for each scenario and year within the SimU based on land cover data available for the corresponding time period. This enabled total SOC storage (Mt) to be summated from SimU and land-cover-specific SOC storage (t) calculated annually from SOC stock multiplied by the cropland or grassland area in a given year.

T a b l e 2. Theoretical scenarios of cropland to grassland and grassland to cropland conversions within the individual SimU and land cover class which the scenario represents in the respective time intervals.

Land cover conversion		**Land cover class**				
Type	**Occurred in**	**1970**	**1971–1980**	**1980–1990**	**1990–2000**	**2000–2013**
C (no change)	-	C	C	C	C	C
C to G	1971	C	G	G	G	G
C to G	1980	C	C	G	G	G
C to G	1990	C	C	C	G	G
C to G	2000	C	C	C	C	G
G (no change)	-	G	G	G	G	G
G to C	1971	G	C	C	C	C
G to C	1980	G	G	C	C	C
G to C	1990	G	G	G	C	C
G to C	2000	G	G	G	G	C

Notes: C – cropland; G – grassland.

Results

Historical development of land cover and land use

The agricultural area in the *Ondavská Vrchovina* highland region in 2013 was 34% of the total area (107,506 ha). This area was assumed as constant throughout the simulation period, and

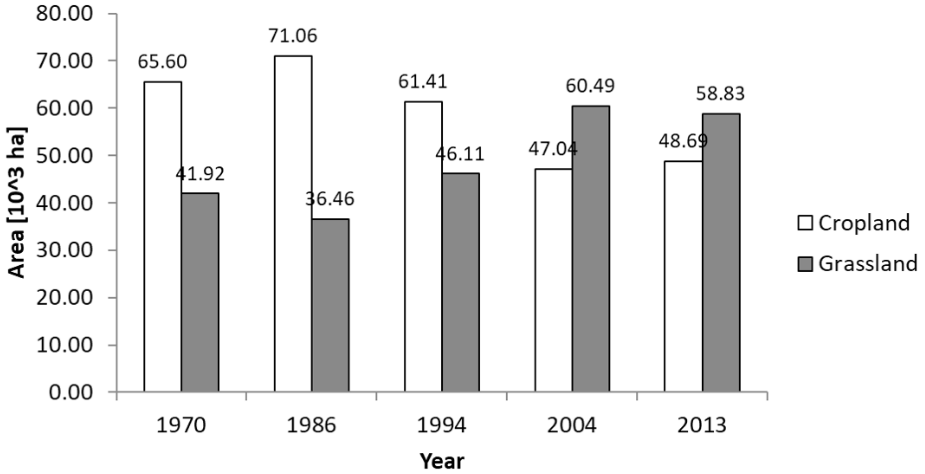

Fig. 2. Historical development of cropland and grassland areas in the *Ondavská Vrchovina* highland region between 1970 and 2013.

therefore, only the impact of cropland and grassland conversions on total SOC storage was estimated. The combination of cropland and grassland, which was 61 and 39%, respectively, in 1970, underwent maximum change in 1986 to 66 and 34%, respectively. The cropland percentage then decreased continuously to 57% of the total area by 1994, to a minimum of 44% in 2003 and slightly increased to 45% by 2013 (Fig. 2); grasslands correspondingly increased to 43, 56 and 55% at these times.

Table 3 lists the average organic carbon inputs from plant residue and farmyard manure estimated from existing data from the *Ondavská Vrchovina* highland region between 1970 and 2013. Average inputs, however, varied between the cropland and grassland.

The average organic carbon input to cropland was approximately 2.3 t C ha^{-1}, with farmyard manure supplying 24% of this amount. This remained balanced between 1970 and 2000,

T a b l e 3. Input of plant residua and farmyard manure as estimated for the Ondavská Vrchovina highland region between 1970 and 2013.

Decade	**Cropland (t C ha^{-1} yr^{-1})**			**Grassland (t C ha^{-1} yr^{-1})**		
	C_PR	**C_FYM**	**C_PR+ C_FYM**	**C_PR**	**C_FYM**	**C_PR+ C_FYM**
1970–1980	1.74	0.54	2.28	2.88	1.07	3.95
1981–1990	1.64	0.54	2.18	2.88	1.17	4.05
1991–2000	1.75	0.53	2.28	2.09	0.66	2.75
2001–2013	2.62	0.57	3.19	1.77	1.07	2.84

Notes: C_PR – organic carbon from plant residua; C_FYM – organic carbon from farmyard manure.

but whilst the average organic carbon input increased by approximately one-third between 2001 and 2013, the farmyard manure portion decreased to 18%.

Meanwhile, the average amount of organic carbon input to the grassland was approximately 4 t C ha^{-1} between 1970 and 1990, and farmyard manure was approximately 28% of this amount. The average organic carbon input then decreased from 1990 to 2013 to approximately 2.8 t C ha^{-1}, but the plant residue to farmyard manure ratio remained balanced, with the farmyard manure carbon input at 31%.

Initial SOC stock and its temporal dynamics between 1970–2013

The average initial 0- to 30-cm topsoil SOC stock in the *Ondavská Vrchovina* region was 38.4 t ha^{-1} in 1970, with 38.4 t ha^{-1} for croplands and 39.0 t ha^{-1} for grasslands. This initial value was 30% lower than the national average of 54.7 t ha^{-1} around 1970 (Barančíková et al., 2010); and it was mainly due to low 1% average topsoil SOC concentrations in the areas' dominant *Cambisol* and *Luvisol* soil types, with even lower values for *Luvisols* (Table 2).

The average SOC stock in agricultural land estimated for 2013 was 49.2 t ha^{-1}. Figure 3 highlights that this was 28% higher than the initial 1970 stock and it continually increased over the entire simulation period, except for a minor decrease around 2000.

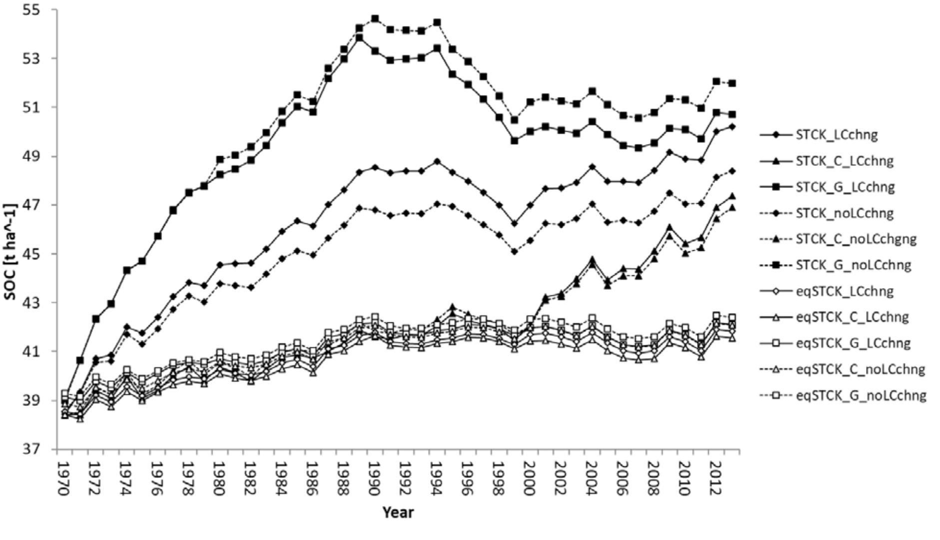

Fig. 3. Temporal development of SOC stock in *Ondavská Vrchovina* highland region between 1970 and 2013 under two different organic carbon inputs levels and with and without assuming land cover change in SOC simulations. Notes: STCK, SOC stock (ha^{-1}) as averaged across cropland and grassland areas together; STCKC, SOC stock (t ha^{-1}) as averaged across cropland only; STCKG, SOC stock (t ha^{-1}) as averaged across grassland only; LCchng, land cover change assumed; noLCchng, no land cover change assumed; eq, simulations with the organic carbon inputs at equilibrium.

The cropland SOC stock continuously grew between 1970 and 2013 with more intensive topsoil SOC accumulation simulated in the final period of 2003–2013. Its 2013 average of 47.4 t ha^{-1} SOC was 23% higher than that at the beginning of the simulation period. Meanwhile, the grasslands SOC stock increased in the initial period of 1970–1986, culminating in 54.0 t ha^{-1} SOC in 1990. It then decreased rapidly to the 1986 level by 2003, and this value remained balanced over the next 10 years, so that the final 2013 49.4 t ha^{-1} SOC averaged 27% more than initial SOC stock.

Land cover and land use change impact on SOC stock

The impact of land cover change on SOC stock dynamics was analysed on the assumption that initial 1970 cropland and grassland areas remained constant over the entire simulation period and no land cover change occurred between 1970 and 2013 (Fig. 3).

However, whilst grasslands SOC stock values without land cover change were slightly higher than those with change over this period, the situation was different in croplands. There, the SOC stock values without land cover change were slightly higher before 2000 and then slightly lower than SOC stock estimated with land cover change (Fig. 3).

The difference in grassland SOC stock values in scenarios with and without land cover change is attributed to cropland to grassland conversions, mostly after 1990 when cropland areas with lower SOC stock commenced vast change to grasslands (Fig. 2). This was generally amplified in all agricultural land, where larger cropland areas negatively affected average SOC stock throughout the entire simulation period, with an average of 2.5% less than the SOC stock estimated with land cover change (Fig. 3).

The combined impact of amount and type of organic carbon inputs on SOC stock dynamics was analysed by comparing the simulation results from reconstructed annual organic carbon inputs with those plant residues established in the RothC equilibrium runs (Fig. 3, Table 3). The latter results provide organic carbon input, which is well balanced with initial SOC stock, and this can maintain unchanged SOC stock for long periods under constant climate conditions. It was estimated at 2.23 t C ha^{-1} yr^{-1} and included no farmyard manure.

Meanwhile, the combined croplands and grasslands average SOC stock relative to initial difference between the two was 14%. The maximum of 22% was reached around 1990, and the results with plant residue at equilibrium were lower in both instances (Fig. 3, Table 4).

T a b l e 4. Relative to initial [%] differences in SOC stock (t ha^{-1}) during the simulation period (1970–2013) between the simulations with reconstructed organic carbon inputs (t C ha^{-1} yr^{-1}) from Table 3 and balanced organic carbon input at equilibrium (2.23 t C ha^{-1} yr^{-1} for both the cropland and grassland).

	Cropland + Grassland	**Cropland**	**Grassland**
Mean	14.2	3.6	20.8
Median	15.4	1.4	21.3
Minimum	0.0	0.0	0.0
Lower quartile	11.5	0.8	20.0
Upper quartile	17.9	5.9	25.2
Maximum	21.8	15.2	30.0

The average difference between the two in grasslands was even higher, averaging 20.8% and culminating 30% by 1990. Table 3 highlights that the reconstructed historical amount of organic carbon inputs to cropland was very low before 2000, with only the 2.25 t C ha^{-1} yr^{-1} on an average, with almost negligible difference between the two carbon input scenarios. This increased by almost one-third after 2000 with the greatest difference in 2013 when the SOC stock under organic carbon input at equilibrium was 15.2% lower (Fig. 3, Table 3).

Regional SOC storage between 1970 and 2013

The initial 1970 regional 0- to 30-cm topsoil SOC storage in the *Ondavská Vrchovina* highland region for cropland, grassland and combined cropland and grassland was estimated at 2.57, 1.64 and 4.21 Mt, respectively (Fig. 4). However, by 2013, these amounts altered to 2.34 Mt (9% decrease), 2.67 Mt (63% increase) and 5.16 Mt (22.5% increase). Thus, the croplands and grasslands regional SOC storage ratio was reversed over the simulation period, with cropland to grassland ratios of 62.4 to 37.5% in 1970 and 46.0 to 54.0% in 2013. This is attributed to the historical development of cropland and grassland areas in this period (Fig. 2).

An analysis also highlights land cover conversions in the study period. There was 25.83% decrease in cropland area and 40.65% increase in grasslands, and 8.9% net cropland SOC storage decrease and 74.4% increase in the grasslands. This establishes that land cover change was not solely responsible for SOC storage dynamics. Moreover, Figure 4 shows that the assumption of no land cover change caused only marginal difference in the estimated total SOC storage in the study region from 5.16 to 5.18 Mt, but the individual trajectories for crop-

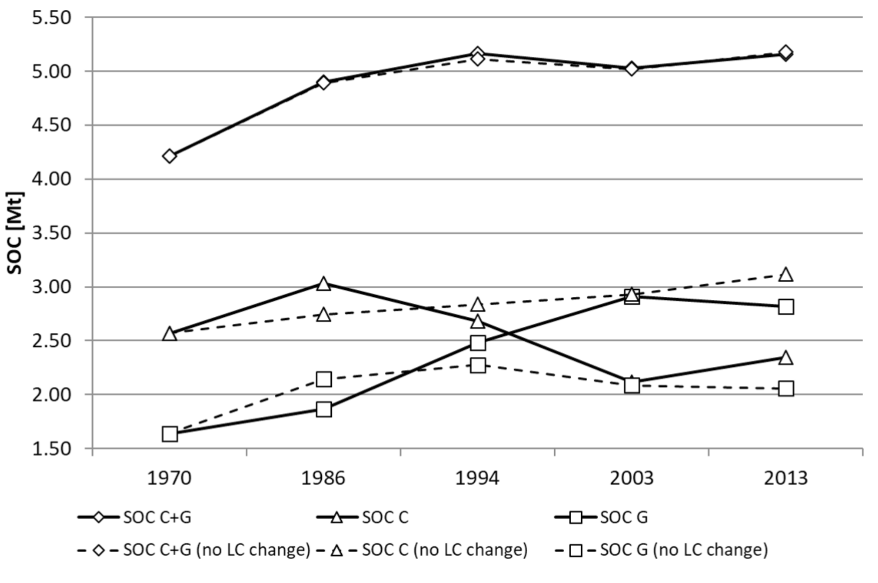

Fig. 4. Development of integral SOC stock (Mt) during the period 1970–2013 with and without conversions between cropland and grassland assumed (LC, land cover; C, cropland; G, grassland).

land and grassland were different. These clearly exemplify the organic carbon inputs to the soil (Table 3), providing 21.4 and 25.5% increase in initial regional SOC storage in croplands and grasslands, respectively.

Although croplands continuously decreased in area, their organic carbon input continued to grow, thus buffering SOC losses. In contrast, the adverse effects of decreased management intensity after 1994 induced less grassland topsoil SOC accumulation than it would be expected by its increased area.

Discussion

Reliability of the RothC model outputs for the study area

The RothC model has responded appropriately to soil management in long-term experiments in Central and Western European conditions and determined realistic estimates (Ludwig et al., 2007; Leifeld et al., 2009). We, therefore, investigated if this held true for the *Ondavská Vrchovina* highland region by comparing the simulated values for cropland, grassland and combined cropland and grassland with the time series of measured topsoil SOC data from 11 permanent soil monitoring sites in the region (Bielek et al., 2005; Kobza, 2015).

Figure 5 depicts that the 1993 measured mean SOC stock was lower than the simulated average values but simulated values were still within the range of observations. An agreement between measured and simulated SOC stock improved towards the end of the simulation period, with simulated SOC stock values in the upper measured quartile. The SOC accumulation between 1970 and 2007 in Slovakia was also simulated at the national scale and validated with measured SOC data by Barančíková et al. (2010).

The 1971–2013 topsoil SOC stock increase under business-as-usual management of soils with low and medium 25–50 t ha^{-1} SOC

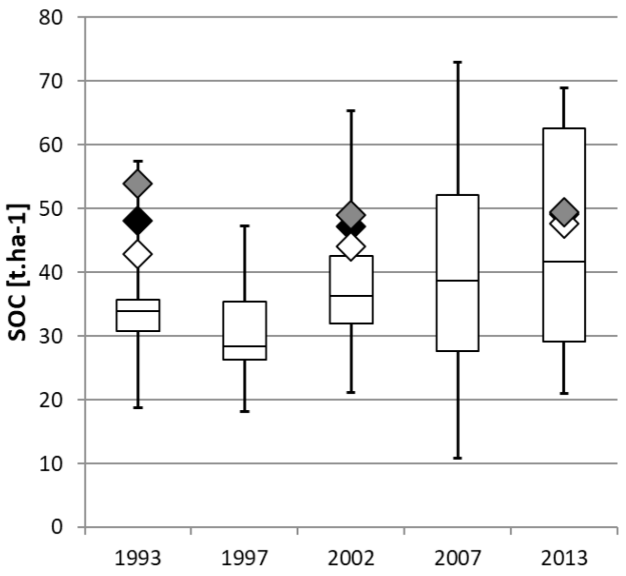

Fig. 5. Mean SOC stock (t ha^{-1}) in cropland, grassland and cropland and grassland together plotted against the measured SOC data from the *Ondavská Vrchovina* highland region.
Notes: Boxes, median; upper and lower quartile, whiskers,10/90th percentile of measured data; diamonds, arithmetic mean of simulated values for cropland (white), grassland (grey), and cropland and grassland together (black).

content was also simulated for similar climate conditions in southwest Poland (Kaczynski et al., 2017). In contrast, Smith et al. (2005) simulated a slightly decreasing trend in 1990–2010 in average crop and grassland topsoil SOC stock under an unchanged European management regime.

Drivers of land cover and land use change

The observed temporal dynamics of cropland and grassland areas in the *Ondavská Vrchovina* region (Fig. 2) are attributed to political and socio-economic changes in Slovak in 1970–2013. These comprised collectivisation and intensification of agricultural production between 1970 and 1990 followed by the sudden decrease in agricultural production after 1990 because of abandoned cropland and less animal husbandry. There was then a slight increase in the production intensity when Slovakia joined the European Union in 2004 (Blažík et al., 2011; Tarasovičová et al., 2013; Skokanová et al., 2016). Similar decreasing trend in agricultural area development was also observed by Masný et al. (2017) in mountain region of Central Slovakia between 1949 and 2006 occurring mostly at terrain positions less favourable for agricultural production. Significant conversions of small-parcel cropland to grassland because of collectivisation after 1970 were also recorded in other sub-mountain regions of Slovakia (Fazekašová et al., 2013; Mojses, Petrovič, 2013). The increasing trend of organic carbon inputs to the cropland in the study area after 1990 is attributed to higher production levels of the most important crops. This was promoted by the 1980's shift from fodder and potato production to biomass-rich technical crops; especially the rapeseed increase in North-eastern Slovakia after 2000 (Tarasovičová et al., 2013).

SOC dynamics in response to land use change

Agricultural soil SOC content is mainly influenced by organic carbon input from plant residue and manure (Schulp, Verbung, 2009; van Wesemael et al., 2010; Wang et al., 2013; Wang et al., 2016), and close correlation between 1970–2007 topsoil SOC content and organic carbon input was also identified in the study by Barančíkova et al. (2013).

In 1970–1990, the *Ondavská Vrchovina* highland region grasslands received substantially higher organic carbon from plant residue and farmyard manure than croplands (Table 3), and this is reflected in the SOC stock development trajectories for both areas before 1994 (Fig. 3). Grassland organic carbon inputs then suddenly decreased in response to 1990 political and socio-economical changes inducing less grazing animals and lower grassland management intensity (Blažík et al., 2011; Skokanová et al., 2016). Crop production was also affected, with cropland organic carbon input maintaining increasing trend after1990 and with substantial increase in 2003–2013.

This is explained by higher production of richer crops and the slight increase in stabled animals supplying more farmyard manure to the croplands. The major effect, however, was due to the altered percentages of main crops; the shift from fodder and potato production to the biomass-rich rapeseed in North-eastern Slovakia (Tarasovičová et al., 2013).

The simulations of balanced plant residue input at equilibrium which resulted in 10% increase in average SOC stock in 1970–2013 identifies possible climate impact on SOC stock dynamics. There was slight 1.5% increase in mean annual temperatures, but no change in average rainfall in the pilot area (data not shown). This resulted in less soil humidity and also likely promoted greater SOC accumulation.

Conclusion

Herein, the RothC model simulated 1970–2013 SOC stock and storage temporal dynamics of the cropland and grassland topsoil in the *Ondavská Vrchovina* highland region. The inputs for the model were organised in a 1x1 km spatial resolution grid and comprised national-scale soil and weather data and the historical record of this area's land cover and land use changes.

Between 1970 and 2013, the study area was subject to major land cover and land use change, mainly because of cropland and grassland inter-conversions and the initial 61% cropland in 1970 was reduced to 45% in 2013. The average SOC stock in 0.3-m deep topsoil increased from 38.4 to 49.2 t ha^{-1} during that period, and the average values were 47.6 t ha^{-1} in croplands and 49.4 t ha^{-1} in grasslands. The RothC topsoil SOC stock simulated values were within the range of 1993–2013 SOC measurements.

In conclusion, the total *Ondavská Vrchovina* highland regional SOC storage was 4.21 Mt in 1970 and 5.16 Mt in 2013. This 22% estimated gain documents the importance of land cover conversions and also the impact of organic carbon inputs to the soil from cropland and grassland management regimes. These land cover changes and soil organic carbon inputs are, therefore, most important in determining the regional SOC stock balance and estimates of total regional SOC storage. Finally, they also establish that precise reconstruction of land cover and land use dynamics is an essential prerequisite in regional SOC storage inventories where current SOC data is unavailable and contemporary regional SOC stock and storage must be accurately estimated from historical soil data.

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References

Alvaro-Fuentes, J., Easter, M., Cantero-Martinez, C. & Paustian K. (2011). Modelling soil organic carbon stocks and their changes in the northeast of Spain. *Eur. J. Soil Sci.,* 62, 685–695. DOI: 10.1111/j.1365-2389.2011.01390.x.

Balkovič J., Skalský, R. & Nováková M. (2010). Spatial model of sand and clay distribution in the topsoil of agricultural land of Slovakia. In R. Bujnovský (Ed.), *Vedecké práce VÚPOP 32* (pp. 5−13). Bratislava: VÚPaOP.

Banwart, S.A., Black, H., Cai, Z., Gicheru, P.T., Joosten, H., Victoria, R.L., Milne, E., Noellemeyer, E. & Pascual U. (2015). The global challenge for soil carbon. In S.A. Banwart, C. Noellemeyer C. & E. Milne (Eds.), *Soil carbon. Science, management and policy for multiple benefits* (pp. 1−9). Oxfordshire: CABI.

Barančíková, G., Halás, J., Gutteková, M., Makovníková, J., Nováková, M., Skalský, R. & Tarasovičová Z. (2010). Application of RothC model to predict soil organic carbon stock on agricultural soils of Slovakia. *Soil and Water Research,* 5, 1−9. DOI: 10.17221/23/2009-SWR.

Barančíková, G., Makovníková, J., Skalský, R., Tarasovičová, Z., Nováková, M., Halas, J., Gutteková, M. & Koco Š. (2012). Simulation of soil organic carbon changes in Slovak arable land and their environmental aspects. *Soil and Water Research,* 7, 45−51. DOI: 10.17221/38/2011-SWR.

Barančíková, G., Makovníková, J., Skalský, R., Tarasovičová, Z., Nováková, M., Halas, J., Koco, Š. & Gutteková M. (2013). Changes in organic carbon pool in agricultural soils and its different development in individual agro-climatic regions of Slovakia. *Agriculture (Poľnhospodárstvo),* 59, 9–20. DOI: 10.2478/agri-2013-0002.

Barančíková, G., Skalský, R., Koco, Š., Halas, J., Tarasovičová, Z. & Nováková M. (2014). Farm-level modelling of soil organic carbon sequestration under climate and land use change. In G. Halldórsson, F. Bampa, A.B. Thorsteinsdóttir, B.D. Sigurdsson, L. Montanarella & A. Arnalds (Eds.), *Soil carbon sequestration for climate food security and ecosystem services* (pp. 100–106). Proceedings of the International Conference 27-29 May 2013, Reykjavík, Iceland. JRC Scientific and Policy Reports. European Union JRC 88412.

Bielek P., Čurlík P., Fulajtár E., Houšková B., Ilavská B. & Kobza J. (2005). Soil survey and managing of soil data in Slovakia. In R.J.A. Jones, B. Houšková, P. Bullock & L. Montanarella (Eds.), *Soil resources of Europe* (pp. 317–329). European Soil Bureau Research Report No. 9.

Bielek, P. & Jurčová O. (2010). *Methodology of soil organic matter balance and determination the need for organic fertilization of agricultural soils (in Slovak).* Bratislava: VÚPaOP.

Blažík, T., Falťan, V., Tarasovičová, Z. & Saksa M. (2011). Land use changes in chosen districts of various productive agricultural regions in the context of transformational process. *Geografický Časopis,* 63, 301–323.

Campbell, E. & Paustian K. (2015). Current developments in soil organic matter modelling and the expansion of model applications: a review. *Environmental Research Letters,* 10, 123004. DOI: 10.1088/1748-9326/10/12/123004.

Capriel, P. (2013). Trends in organic carbon and nitrogen contents in agricultural soils in Bavaria (south Germany) between 1986 and 2007. *Eur. J. Soil Sci.,* 64, 445–454. DOI: 10.1111/ejss.12054.

Coleman, K. & Jenkinson D.S. (2014). *ROTHC-26.3 A model for the turnover of carbon in soil. Model description and windows users guide.* Harpenden Herts: Rothamsted Research. https://www.rothamsted.ac.uk/sites/default/files/RothC_guide_DOS.pdf

Falloon, P.D., Smith, P., Smith, J.U., Szabo, J., Coleman, K. & Marshall S. (1998). Regional estimates of carbon sequestration potential: linking the Rothamsted Carbon Model to GIS databases. *Biol. Fertil. Soils,* 27, 236–241. DOI: 10.1007/s003740050426.

Falloon, P. & Smith P. (2012). Modelling soil carbon dynamics. In W.L. Kutsch, M. Bahn & A. Heinemeyer (Eds.), *Soil carbon dynamics: An integrated methodology* (pp. 221–244). Cambridge: Cambridge University Press.

Fazekašová, D., Boltižiar, M., Bobuľská, L., Kotorová, D., Hecl, J. & Krnáčová Z. (2013). Development of soil parameters and changing landscape structure in conditions of cold mountain climate (case study Liptovská Teplička). *Ekológia (Bratislava)*, 32, 197–210. DOI: 10.2478/eko-2013-0017.

Frank, S., Schmid, E., Havlik, P., Schneider, U., Boettcher, H., Balkovič, J. & Obersteiner M. (2015). The dynamic soil organic carbon mitigation potential of European cropland. *Global Environmental Change,* 35, 269–278. DOI: 10.1016/j.gloenvcha.2015.08.004.

Gottschalk, P., Smith, J.U., Wattenbach, M., Bellarby, J., Stehfest, E., Arnell, N., Osborn, T.J., Jones, C. & Smith P. (2012). How will organic carbon stocks in mineral soils evolve under future climate? Global projections using RothC for a range of climate change scenarios. *Biogeosciences,* 9, 3151–3171. DOI: 10.5194/bg-9-3151-2012.

Guo, L.B. & Gifford R.M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology,* 8, 345–360. DOI: 10.1046/j.1354-1013.2002.00486.x.

Hraško, J. & Bedrna Z. (1970). Soil mapping on the territory of Slovakia. In J. Hraško (Ed.), *Vedecké práce výskumného ústavu pôdoznalectva a výživy rastlín v Bratislave č. 4* (pp. 13–19). Bratislava: Príroda.

Janzen, H.H. (2006). The soil carbon dilemma: Shall we hoard it or use it?. *Soil Biol. Biochem.,* 38, 419–424. DOI: 10.1016/j.soilbio.2005.10.008.

Kaczynski, R., Siebielec, G., Hanegraaf, M.C. & Korevaar H. (2017). Modelling soil carbon trends for agriculture development scenarios at regional level. *Geoderma,* 286, 104–115. DOI: 10.1016/j.geoderma.2016.10.026.

Kobza, J. (2015). Permanent soil monitoring system as a basic tool for protection of soils and sustainable land use in Slovakia. *Earth and Environmental Science,* 25, 012011. DOI: 10.1088/1755-1315/25/1/012011.

Lehtinen, T., Schlatter, N., Baumgarten, A., Bechini, L., Kruger, J., Grignani, C., Zavattaro, L., Costamegna, C. & Spiegel H. (2014). Effect of crop residua incorporation on soil organic carbon and greenhouse gas emissions in European agricultural soils. *Soil Use Manag.*, 30, 524–538. DOI: 10.1111/sum.12151.

Leifeld, J., Reiser, R. & Oberholzer H.R. (2009). Consequences of conventional versus organic farming on soil carbon: results from a 27 year field experiment. *Agron. J.,* 101, 1204–1218. DOI: 10.2134/agronj2009.0002.

Linkeš, V., Gromová, A., Lupták, D., Pestún, V. & Poliak P. (1988). *Informačný systém o pôde*. Bratislava: Príroda.
Ludwig, B., Schultz, E., Rethemeyer, J., Merbach, I. & Flessa H. (2007). Predictive modeling of C dynamics in the long-term fertilization experiment at Bad Lauchstadt with the Rothamsted carbon model. *Eur. J. Soil Sci.,* 58, 1155–1163. DOI: 10.1111/j.1365-2389.2007.00907.x.
Ma, K., Liu, J., Balkovič, J., Skalský, R., Azevedo, L. & Kraxner F. (2016). Changes in soil organic carbon stocks of wetlands on China's Zoige plateau from 1980 to 2010. *Ecol. Model.,* 327, 18–28. DOI: 10.1016/j.ecolmodel.2016.01.009.
Makovníková, J. & Širáň M. (2011). *Modelovanie rovnovážnej objemovej hmotnosti pôdy*. Bratislava: VÚPaOP.
Masný, M., Weis, K. & Boltižiar M. (2017). Agricultural abandonment in chosen terrain context – case study from the Poľana UNESCO Biosphere Reserve (Central Slovakia). *Ekológia (Bratislava)*, 36, 339–351. DOI: 10.1515/eko-2017-0027.
Minasny, B., McBrandan, A.B., Malone, B.P., Lacoste, M. & Walter C. (2014). Quantitatively predicting soil carbon across landscapes. In A. Hartemink & K. McSweeney (Eds.), *Soil carbon* (pp. 45–57). Heidelberg: Springer Cham. DOI: 10.1007/978-3-319-04084-4.
Mojses, M. & Petrovič F. (2013). Land use changes of historical structures in the agricultural landscape at the local level — Hriňová case study. *Ekológia (Bratislava)*, 32, 1-12. DOI: 10.2478/eko-2013-0001.
Němeček, J., Damaška, J., Hraško, J., Bedrna, Z., Zuska, V., Tomášek, M. & Kalenda M. (1967). *Průzkum zemědělských půd ČSSR – souborná metodika* (1. díl). Praha: Ministerstvo zemědělství a výživy.
Schulp, C.J.E. & Verbung P.H. (2009). Effect of land use history and site factors on spatial variation of soil organic carbon across a physiographic region. *Agric. Ecosyst. Environ.,* 133, 86–97. DOI: 10.1016/j.agee.2009.05.005.
Skokanová, H., Falťan, V. & Havlíček V. (2016). Driving forces of main landscape change processes from past 200 years in Central Europe – differences between old democratic and post-socialist countries. *Ekológia (Bratislava)*, 35, 50–65. DOI: 10.1515/eko-2016-0004.
Smith, J., Smith, P., Wattenbach, M., Zaehle, S., Hiederer, R., Jones, R.J.A., Montanarella, L., Rounsevell, M.D.A., Reginster, I. & Ewert F. (2005). Projected changes in mineral soil carbon of European croplands and grasslands, 1990–2080. *Global Change Biology,* 11, 2141–2152. DOI: 10.1111/j.1365-2486.2005.001075.x.
Tarasovičová, Z., Saksa, M., Blažík, T. & Falťan V. (2013). Changes in agricultural land use in the context of ongoing transformational processes in Slovakia. *Agriculture (Poľnohospodárstvo),* 59, 49–64. DOI: 10.2478/agri-2013-0005.
van Wesemael, B., Paustian, K., Meersmans, J., Goidts, E., Barančíková, G. & Easter M. (2010). Agricultural management explains historic changes in regional soil carbon stocks. *PNAS,* 107(33), 14926–14930. DOI: 10.1073/pnas.1002592107.
Vopravil, J., Podrázský, V., Khel, T., Holubík, O. & Vacek S. (2014). Effect of afforestation of agricultural soils and tree species composition on soil physical characteristics changes. *Ekológia (Bratislava)*, 33, 67–80. DOI: 10.2478/eko-2014-0008.
Wang, G., Luo, Z., Han, P., Chen, H. & Xu J. (2016). Critical carbon input to maintain current soil organic carbon stocks in global wheat systems. *Scientific Reports,* 6. DOI: 10.1038/srep19327.
Wang, J., Lu, S., Xu, M., Zhu, P., Huang, S., Zhang, W., Peng, C., Chen, X. & Wu L. (2013). Soil organic carbon sequestration under different fertilizer regimes in north and northeast China: RothC simulation. *Soil Use Manag.,* 29, 182–190. DOI: 10.1111/sum.12032.

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ASSESSMENTS OF BIODIVERSITY AND HABITAT SERVICES IN CITIES – EXEMPLIFIED BY DRESDEN (GERMANY) AND LIBEREC (CZECH REPUBLIC)

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Abstract

Bastian O., Cudlín P., Pechanec V., Brzoska P., Štěrbová L., Včeláková R., Purkyt J., Grunewald K.: Assessments of biodiversity and habitat services in cities – exemplified by Dresden (Germany) and Liberec (Czech Republic). Ekológia (Bratislava), Vol. 39, No. 2, p. 174–189, 2020.

The choice of suitable biodiversity assessment methods for practical purposes in city planning and decision-making is still a challenging problem. Despite the availability of a wide variety of methods for almost all dimensions of diversity (mainly species and habitat diversity, including spatial aspects), few of them have entered the practical 'daily' work. In the example of in-depth examinations in German and Czech cities (e.g., Dresden and Liberec), it was found that the most frequently applied analyses are those of protected species and habitats in urban nature conservation in general, and particularly in city development planning to derive avoidance, protection and compensation measures. Preference analyses (questionnaires, structured interviews) are becoming increasingly popular. Economic calculations of habitat values and the valuation of ecosystem services are still in their infancy. We will present methods that are presently being applied or could be included in a practical methodological toolkit to analyse and valuate biodiversity in urban nature conservation, city planning and decision-making.

Key words: economic valuation, land use, preference analyses, protected species, vegetation.

Introduction

Plants, animals, and microorganisms including fungi are the basis of all ecosystems and the services they provide. Not only pristine and rural areas but also urban and peri-urban areas can host a rich biodiversity and deliver a wide range of benefits to sustain and improve human livelihood and the quality of life through ecosystem services (TEEB, 2011). Due to the permanently growing proportion of people living in cities, biodiversity is continuously gaining importance. On the one hand, the diversity of species and genetic variants is an inherent component of ecosystems; it can be a regulator of ecosystem processes but also a prerequisite for various ecosystem services it supports. On the other hand, biodiversity itself can be a good (Mace et al., 2012) or the result of ecosystem services (particularly the habitat service, i.e., the capacity or the service to provide suitable living spaces and living conditions for plants and animals), which can be classified as a regulating, supporting or intermediate service. In the sense of the international CICES classification of ES (Haines-Young, Potschin, 2018), habitat service(s) belong to the 'group' of 'Lifecycle maintenance, habitat and gene pool protection', which is part of the division of 'Maintenance of physical, chemical, biological conditions'.

Cities can be important sites of local and regional biodiversity. Within their boundaries, the richness of land cover types and intensities of land use create a wide array of different habitats and microhabitats, and highly varied habitat mosaic configurations. Biodiversity in cities is adapted to human settlements in different ways. There are species that occur predominantly or exclusively in cities or species that occur both in urban areas and in the wider landscape, but also species that avoid urban spaces and habitats (Werner, Zahner, 2009).

Ecological characteristics of cities relevant for urban biodiversity include the existence of (e.g., Werner, 2016; Bastian, Xiao, 2018):

- dry and heat islands,
- small structures, small-scale spatial patterns of site conditions and land use,
- wilderness on fallow land,
- frequently disturbed habitats,
- a high share of thermophilic and non-native species,
- refuge and substitute habitats, stepping stones.

It is very important to assess the present state of biodiversity in cities in order to draw the appropriate conclusions for planning and management, on the one hand because of the (intrinsic) value of biodiversity underpinned by legal obligations, on the other hand as the sustainable supply of many ecosystem goods and services largely depends on maintaining biodiversity at a favourable conservation status.

More or less all spheres of biodiversity (UNEP, 1992) are relevant in city planning and management, especially:

- species diversity (number of species),
- diversity of ecosystems, biocenoses, habitats, and landscapes,
- for particular questions, also genetical diversity (within-species diversity), for example, varieties of fruits.

The choice of suitable biodiversity and habitat service assessment methods for practical purposes in city planning and decision-making is still a challenging problem. Data availability, personnel and monetary resources are crucial as well as the specific tasks to be performed, for example, nature conservation (protection of species, habitats, areas), compensation and replacement measures, land use and landscape planning. A wide variety of methods and approaches have been described in the literature or used in case studies (User's manual for CBI, 2012; Grunewald, Bastian, 2017). But few methods are actually applied under practical conditions in city administrations.

The aim of the study was to find which role methods to analyse and valuate biodiversity in its different dimensions play in urban nature conservation, city planning and decision-making. In this paper, we present a bundle of methods that are presently being applied or could be included in a practical methodological toolkit without major difficulties, particularly in cities. We refer to data availability, feasibility to work with these methods, and expressiveness of the results. We show advantages and problems of methods, and we discuss necessities, opportunities and limitations for broadening the toolkit.

The investigation was performed in the framework of the EU project BIDELIN (The value of ecosystem services, biodiversity and blue-green infrastructures in cities, exemplified by Dresden, Liberec and Děčín) (2017–2019), supported by the European Union, the Free State of Saxony (Germany) and the Czech Republic (Project No. 100282320) (IÖR, 2019). Therefore, we focus on experiences, data and applications from Germany and the Czech Republic, particularly from the cities of Dresden and Liberec.

Dresden is the capital city of the German federal state of Saxony, which borders on the Czech Republic and Poland. The city lies in a marked widening of the *Elbe* river valley. Its total population is c. 563,000 inhabitants. The territory stretches over several physical landscapes: a hilly loess region, and foothills of lower mountain ranges. The *Elbe* river with its broad floodplain mostly covered by semi-natural meadows crosses the entire city from southeast to northwest. Though surface sealing and intensive land use have changed the natural conditions drastically, the inner city is greened by parks and avenues but also by many small habitats of partly rare and threatened plants and animals.

Liberec is one of the biggest and most important cities of Northern Bohemia. The city lies at the *Lužická Nisa* at the foot of the *Ještědský hřbet,* covered partially by the Nature Park *Ještěd* and the Protected National Area *Jizerské hory* mountains. Commercial spruce forests, some beech forests, and semi-natural meadows occur on its cadastral territory. However, in the city centre, there are only a few parks and other urban greenery. Its population is 102,000 inhabitants.

Methods

In order to identify methods that are presently being applied or could be included in a practical methodological toolkit to analyse and valuate biodiversity in urban nature conservation, city planning and decision-making, the following working steps were involved in the investigation:

- literature analysis referring to biodiversity assessment methods in cities,
- assessment of the practical experiences in city administration and planning, particularly in the cities of Dresden and Liberec,
- evaluation of the suitability of existing methods to assess biodiversity and ecosystem services in cities,
- application of these methods in the framework of the BIDELIN project to assess ecosystem services in both cities included in the study,
- identification of research gaps, challenges, and limitations.

We classified the methods according to their complexity, from the direct counting and mapping of species and ecosystems to more complex, integrated approaches such as the degree of naturalness and the habitat value, and we stressed preference analyses and monetary methods like the Habitat Valuation Method and valuation based on restoration costs.

Thus, we included methods belonging to the three broad perspectives on valuing biodiversity that cover ecological, socio-cultural and economic benefits of biodiversity as distinguished in the Millennium Ecosystem Assessment (MA, 2005) and TEEB (2010). This is also in line with the three main approaches of valuing biodiversity in environmental management proposed by Laurila-Pant et al. (2015), where the first approach is to value biodiversity in terms of the services provided for society, while the second approach is to assess socio-cultural values; the last approach adopts a biological viewpoint.

Results

Ecological indicator approaches

Counting and mapping of rare/protected species. Recording and mapping of animal and plant species is a very common tool, the application of which is driven mainly by legal means (national nature conservation acts, European regulations, e.g., Natura 2000). A special focus is placed on species being protected and/or endangered (red list species).

According to German legislation, it is not allowed to kill or destroy native animals and plants without reasonable cause. There are also categories of special or strict protection: Among animals, all native bird species (except, e.g., feral domestic pigeons) and bats, including their habitats, are protected, many mammals, all reptiles, amphibians, but also fishes and insects are included. That means, it is not allowed to pursue or even kill them. A similar situation obtains in the Czech Republic, where all autochthonous species of plants and animals are protected if they do not occur on land to be managed in a standard manner.

Development planning is obliged to take care of protected species. Such plans have to lay down avoidance measures (planning alternatives, construction field preparation outside the breeding season) and compensation. Measures to ensure the Continuous Ecological Functionality (CEF) are obligatory; they must be implemented completely before the construction works are started, for example, creation of substitute habitats for birds, sand lizards, rare beetles and butterflies (EEC, 2007).

There are various data sources for species, mainly faunistical and floristical maps and publications. In many cases, however, the data are incomplete, not up to date or at an inappropriate scale. Thus, it is often necessary to engage consultants to analyse the occurrence of (particularly European) protected species in a planned development area and to draw conclusions for protection, avoidance and compensation measures.

An important data source for biodiversity issues (among them species and habitats) in both countries is the landscape plan, which is defined as the expert planning of nature conservation and landscape management (Haaren, Albert, 2011; Riedel et al., 2016). Vice versa, the landscape plan needs to be fed with biodiversity data (Bastian, 1998). To a large extent, the landscape plan covers the spatial dimension, which is very important for biodiversity conservation and management.

The landscape plan is an important document that gives an overview of habitat/biotope issues and provides a framework for conservation and management. Both landscape plan of Dresden and urban plan of Liberec are similar in the basic content. But there are several specific features, partly indicated below. For example, the landscape plan of Dresden (Landschaftsplan Dresden, 2018) refers to

- animals, plants, especially rare and protected species,
- historical woodlots (valuable for specialized animal species),
- highly valuable woods,
- biotope compound systems, habitat connectivity: core areas of habitat connection and species diversity, corridors and stepping stones (similar to the USES concept of the Czech Republic and Slovakia) (Diviaková et al., 2018),

- core areas for protected building-inhabiting animal species,
- conservation measures for amphibians along roads,
- precautionary assessment of species populations before the implementation of measures designed in the landscape plan,
- maintenance and improvement of habitats of ground-nesting bird species on arable fields and meadows,
- deficits and threats to protected areas, species, habitats and connectivity (e.g., barriers).

In the urban plan of Liberec (City plan Liberec, 2019), special emphasis is placed on

- significant elements of ecological stability,
- urban greenery,
- scattered greenery in the landscape,
- peripheral forest landscape,
- territorial system of ecological stability (USES),
- landscape character.

Both landscape plans are similar concerning nature protection in general, but the Dresden plan puts greater emphasis on protecting biodiversity, while the Liberec plan focuses primarily on supporting environmental stability.

Vegetation

Vegetation can be regarded as the most conspicuous biotic component of a terrestrial area. We distinguish between current vegetation and actual potential natural vegetation (Tüxen, 1956). The actual vegetation is shaped by natural conditions and land use. The potential natural vegetation is defined as the vegetation representing the abiotic site conditions (including all essential irreversible changes) as a manifestation of the abiotic environment. The comparison of the actual vegetation and the potential natural vegetation enables the assessment of the actual changes in the vegetation, particularly the extent of human impact. An indicator for this difference is the naturalness of vegetation (e.g., after Schlüter, 1982, 10 degrees), or reciprocally the hemeroby (Table 1).

For the differentiation of nature in cities, the degrees of hemeroby, synanthropy and naturalness are suitable indicators (Table 1).

In view of naturalness or human influence, Kowarik, Körner (2005) distinguished four basic forms of nature found in cities:

- Nature 1 'old wilderness': remnants of pristine nature,
- Nature 2 'traditional cultural landscape': continuity of former agricultural or forested land,
- Nature 3 'functional greening': urban parks, green areas and gardens,
- Nature 4 'urban wilderness': new elements from natural colonization processes, particularly distinct on urban wastelands.

With the aim of achieving a comprehensive evaluation of urban nature, an additional form Nature 5 'artificial and unnatural elements' (e.g., buildings, green roofs) was developed within the framework of the project. This form of nature encompasses heavily sealed or built-up areas, which may also be partially greened, but can still be regarded as predominantly

T a b l e 1. Hemeroby of vegetation (modified after Blume, Sukopp, 1976; Bastian, Schreiber, 1999).

Hemeroby	**Anthropogenic vegetation changes**	**Ecosystem types (examples)**
Ahemerobic	None	Remnants of pristine nature
Oligohemerobic	Rather low/few	Near-natural forests, bogs
Mesohemerobic	Medium	Semi-natural or more artificial forests, heaths, dry and rough meadows, extensively used meadows and pastures
β-euhemerobic	Strong	Intensively used grassland and forests, perennial ruderal vegetation, segetal communities (on arable fields)
α-euhemerobic	Very strong	Special cultures (orchards, vineyards, ornamental lawns), annual pioneer and ruderal vegetation
Polyhemerobic	Extremely strong	Waste deposits, spoil heaps, paved paths, gravelled railroad tracks
Metahemerobic	Vegetation totally removed	Poisoned ecosystems, totally built-up/sealed areas (buildings, tar surface)

artificial in the overall urban context. Although this type of urban nature is not nature in the 'classical' sense, it should nevertheless be regarded as a potential ecosystem, and thus, a potential provider of habitats for different species, e.g., plants. Thus, areas are also included that would not have been considered according to the four basic forms of nature according to Kowarik, Körner (2005).

By means of the extended classification of urban areas with regard to their form of nature, as shown in Fig. 1, as an example for Dresden, areas can be identified in which there is urban planning potential with regard to preserving nature (Nature 1 and 2). These kinds of area can be identified, for example, within the areas of the urban forest 'Dresdner Heide' or along the Elbe river (see Fig. 1). Additionally, this classification shows parts of the city where upgrading measures could be necessary (Nature 4 and 5). Furthermore, the spatial distribution of urban nature forms shows where an increased need for horticultural maintenance (Nature 3) and an increased demand for the provision of habitats for a possible promotion of biodiversity (Nature 4 and 5) can be expected. The latter occur especially in the city centre of Dresden (see Fig. 1). Thus, the extended urban nature approach according to Kowarik, Körner (2005) is interesting for assessing the existing nature forms in a city and for identifying areas with special need regarding the promotion and preservation of urban biodiversity.

Classification of land cover data (biotope and land use mapping)

Land cover maps showing type and intensity of land use combined with biotope types are widely distributed and frequently applied. Land cover data include indicators that can be used to characterize human impact on the territory and correspondingly the degree of naturalness. This data type can also serve to roughly characterize vegetation and can indicate the site conditions crucial for organisms, for example, estimate the intensity of human influence on the microclimatic conditions of the vegetation cover and on the habitat structures for wild plants and animals. Moreover, land cover data imply causal ecological networks and running processes. Therefore, one can obtain the frequency and distribution of all habitat types

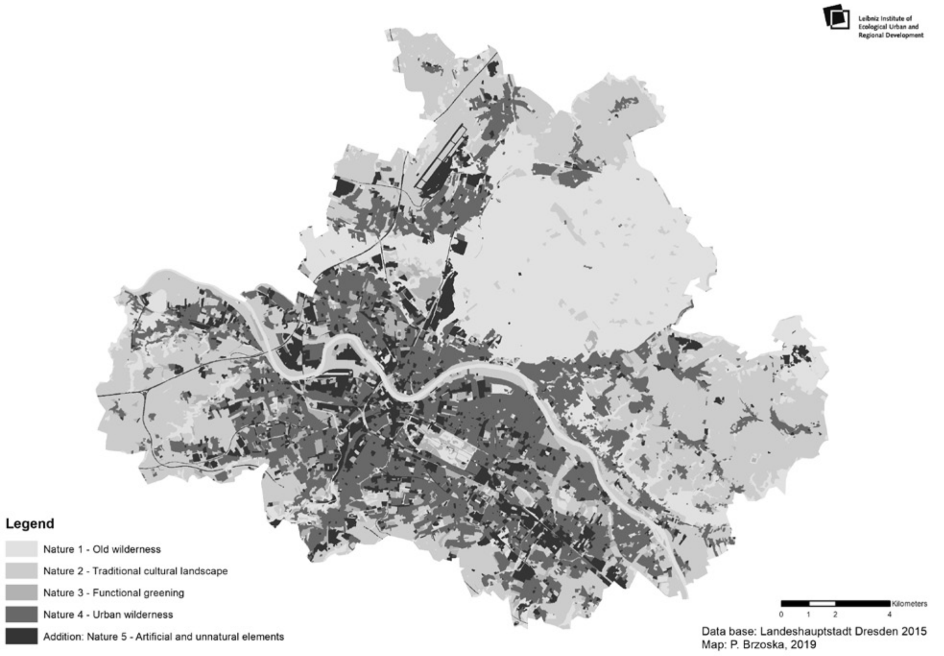

Fig. 1. Five basic forms of nature in cities (after Kowarik, Körner, 2005, modified).

T a b l e 2. Classification of habitat types (role for biodiversity and habitat function) – 5 degrees (1 – the highest; 5 – the lowest) (according to Bastian, Schreiber, 1999).

Habitat/land use type	**Habitat value**
Deciduous forest	1
Humid/wet forest	1
Mixed forest	1.5
Coniferous forest	2
First afforestation	2
Tree row, alley	2.5
Hedge	2
Orchard meadow	1
Dry, nutrient-poor grassland	1
Wet grassland, moist ruderal area	2
Grassland, dry ruderal area	2.5
Fields, arable land	4
River	2.5
Riparian vegetation	1
Pond	1
Sports, leisure area	3.5
Industrial site	4.5
Open building area	3
Dense urban development	5
Road	5

of an area as well as information on their naturalness.

The landscape plan of Dresden includes habitats and land cover maps that are based on Colour Infrared Photographs (CIR). These data identify about 100 different habitat types, which can be further differentiated by structural and additional features (primary scale 1:5,000, generalized 1:50,000), and can be combined with 19 comprehensive habitats and land use complexes. By classifying the habitat or land cover types with regard to their suitability for being habitats of flora and fauna (and for hosting a rich biodiversity), it is possible to identify habitat values (see Table 2).

The classification of the habitat and land cover types in the city of Dresden shows the amount and the distribution of areas that exhibit high to low habitat val-

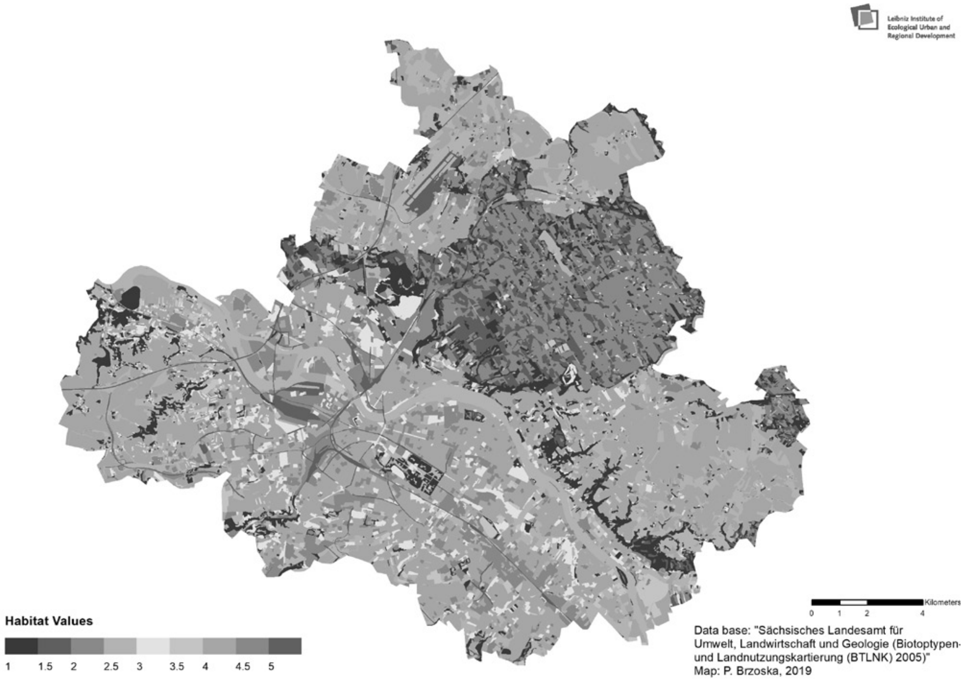

Fig. 2. Values of habitat and land cover types in the city of Dresden (green colour and low numbers: highest values, orange and red colour and high numbers: lowest values; see Table 2).

ues, and therefore, indicates their significance for the protection and preservation of biodiversity in the city (Fig. 2). Areas with the highest habitat values are in the northern part of Dresden (dark and light green areas). These areas include mainly large forest territories, such as the *Dresdner Heide*, as well as agricultural regions in the outskirts. But small patches within urban parks, as indicated in the *Großer Garten* in the city centre, may also exhibit high biotope values and can be seen as important near-natural habitats. The lowest habitat values (dark red areas) are located in the central parts of Dresden with high soil sealing and very high human influence. But in general, most of the city area is classified with low habitat values (orange areas).

Protected areas

Settlements can host important substitute habitats for various threatened species of natural and cultural landscapes (Müller, Abendroth, 2007). Following the principle of differentiated land use and nature conservation intensities (Haber, 1998) across the entire territory, protecting biodiversity in urban areas must include all categories of nature, from the remnants of natural landscapes (e.g., near-natural forests), cultural landscapes (orchards), urban-industrial areas including rural settlements and landscaped areas. The core areas of nature conservation, however, are represented by various legally protected areas, also in cities.

The conservation status implies – as a rule – the high value of an area in terms of biodiversity (and habitat services) but also a higher priority in planning and decision-making. We can find various categories of protected areas and objects in the cities of Dresden and Liberec (Table 3). There are also sites of the EU-wide Natura 2000 network, which is designed to help achieve the goal of the European Union of 'halting the loss of biodiversity and the degradation of ecosystem services in the EU by 2020, and restoring them in so far as feasible, while stepping up the EU contribution to averting global biodiversity loss' (COM, 2011). Natura 2000 comprises two basic categories: (especially and strictly) protected species and protected areas – FFH sites (Fauna-Flora-Habitat Sites = Special Areas of Conservation pursuant to the Habitats Directive 21.5.1992, 92/43/EWG), and Special Protection Areas (SPA, pursuant to the Wild Birds Directive 2. April 1979, 79/409/EWG).

T a b l e 3. Nature conservation areas and objects in Dresden and Liberec (Schutzgebiete Dresden, 2018; AOPK, 2011).

	Dresden	**Liberec**
Total size of the territory	328.28 km²	10.61 km²
Nature reserves	248 ha	71 ha
Natural monuments	134 ha	35 ha
Landscape protection areas	12,238 ha	1,460 ha
Protected habitats	1,470 ha	
Valuable trees	c. 1,950 individuals	
FFH	1,901 ha	1 ha
SPA	1,609 ha	

Unlike comprehensive habitat and land cover mapping, selective habitat mapping considers only for valuable (and protected) habitats (partly synonymous: biotopes) (Cudlín et al., 2005).

Examples of legally protected habitat types in Germany are natural running and standing waters, including their riparian vegetation, bogs, swamps and reeds, dunes, heaths, bogs, swamp and riverside forests, natural rocks, natural coast areas; in Saxony and some other German federal states additionally, for example, nutrient-poor fresh and mountain meadows, hollow trees, orchard meadows, clearance cairns, ravines and dry walls. As an additional category, the landscape plan of Dresden also mentions 'complex of habitats or land use types with great significance for biodiversity' (three-grade scale: medium, high, very high).

Economic valuations

Biodiversity is not only an ecological, but also a social and economic issue (Laurila-Pant et al., 2015). Economic valuations of biodiversity/ecosystems in cities are still not very common. They are of great significance, particularly in terms of cost calculations for compensation measures (for species and habitats) and for landscape management.

Habitat values are used in the framework of the Impact Mitigation Regulation (Albrecht et al., 2014) for comparison and compensation purposes. These values are regarded as physical indicators for the performance of an ecosystem to maintain biodiversity, or – depending on the specific application – as part of the physical capital value of an ecosystem. First, this value is non-monetary, but it has the characteristics of exchange values, which can easily be transformed to monetary values.

Monetary values for habitat types are also used in the Impact Mitigation Regulation. If values and functions of nature are impaired by an avoidable impact (e.g., by construction measures), the damage can be calculated as the equivalent of the costs of compensation (substitution and compensation costs) or replacement measures (replacement costs) necessary for the recovering of the functions of nature. The starting point for the derivation of monetary values are nature conservation requirements on the extent of compensation and replacement measures (compensation obligation, German Nature Conservation Act §8). This obligation can be seen as the consensus of society to maintain the functions of nature (or the ecosystem services). The compensation cost is adequate to society's willingness-to-pay.

This ties to the Habitat Equivalency Investment Model (Schweppe-Kraft, 2009), which calculates the restoration costs while taking the necessary time or duration of development into account. In a similar vein, Seják et al. (2003) created a Habitat Valuation Method (HVM) for habitat type quality assessment, which combines the expert and habitat-specific restoration cost assessment (Table 4). They awarded scores between 1 and 6 for each of the criteria phylogenetic maturity of vegetation, habitat naturalness, structure diversity, species diversity, rarity of the habitat type, rarity of species, habitat vulnerability, and threat to habitat type (Seják, Cudlín, 2010). The financial value of one point corresponds to the arithmetic mean of the costs of analysed restoration projects in the Czech Republic spent to improve the ecological state of 1 m^2 by one point. In 2003, the authors assigned monetary values to the score (1 point = 0.40 €); by 2015, it had increased due to inflation to 0.59 € (Pechanec et al., 2017); and in 2018, it was re-counted using new restoration projects and reached 1.14 €.

Using the Habitat Valuation Method, a total habitat value of 106,087,048 € was found in the Liberec cadastral territory. The total value for 1 m^2 was 22.3 € (220,300 € ha^{-1}), which means that the annual value was 1.12 € m^{-2} (10,000 € ha^{-1} year^{-1}). It must be stated, however,

T a b l e 4. Habitat types and their monetary values in the Czech Republic, calculated with the Habitat Valuation Method (Pekanec, unpublished data).

Habitat	Name	Area in CR (km^2)	Point value	Monetary value (EUR) of 1 m^2
T1.1	Mesic *Arrhenatherum* meadows	1 907.16	33	37.62
L5.1	Herb-rich beech forests	1 229.30	45	51.30
L3.1	Hercynian oak-hornbeam forests	1 010.61	47	53.58
L2.2	Ash-alder alluvial forests	796.06	42	47.88
L9.1	Montane *Calamagrostis* spruce forests	438.81	36	41.04
T1.5	Wet *Cirsium* meadows	416.78	49	55.86
T1.3	*Cynosurus* pastures	408.56	39	44.46
K3	Tall mesic and xeric scrub	351.90	33	37.62
L2.3	Hardwood forests of lowland rivers	241.38	66	75.24
L4	Ravine forests	209.34	42	47.88

that such monetary calculations with the Habitat Valuation Method have rarely been applied in practical work.

As nature conservation and habitat management are nearly always limited by scarce monetary resources, the description of biodiversity in monetary terms is useful, particularly in terms of the costs for maintaining, managing and restoring valuable habitats.

Grunewald et al. (2014) developed a methodology for a regional (state-wide) calculation of landscape management measures and costs. It was compiled in an exemplary manner for the Federal State of Saxony (c. 18,420 km²) in Germany. Table 4 shows an example for the city of Dresden. Costs of habitat management are required to plan landscape management and nature conservation measures. They would also be useful to calculate (and demand) a budget, which truly reflects the total annual expense of maintaining (and developing) valuable biotopes (Table 5).

T a b l e 5. Annual management costs for valuable biotopes in the city of Dresden, Germany (after Grunewald, Syrbe, 2013; Geißler, 2017, modified).

Acronym	**Biotope type**	**Total area (ha)**	**Ann. costs €/ ha**	**Ann. costs € / habitat type**
BS	Orchard meadows	278.4	600	167,040
GM	Extensively used nutrient-poor mesophilic grassland	264.0	413	109,032
RS	Acid dry grassland	40.0	520	20,800
	All protected biotope types in Dresden	1,470 ha		702,844 (total costs)

For ecosystem service assessments, the biophysical accounting is only one side of the coin. It represents only the biophysical basis or the supply side. It should be complemented by demand-oriented approaches that relate more to the perspective of users and beneficiaries. In the broadest sense, such preference analyses are also economic valuation methods.

Public appreciation of urban biodiversity

People place value on species, ecosystems and biodiversity in general as well as on ecosystem services without necessarily assigning monetary values. There are a lot of methods for expressing preferences in non-monetary yet quantifiable terms, including quantitative and qualitative techniques (i.e., surveys, interviews), participatory and deliberative tools (e.g., Hofmann et al., 2012; Botzat et al., 2016; Moyzeová, 2018). A recent representative study by Fischer, Kowarik (2018) regarding a question on the appreciation of urban species-diversity through people in the city indicates that a high degree of biodiversity in urban environments is accepted and supported the most. In this study, about 4000 people in five different European countries with diverse backgrounds were interviewed and asked about their preferences.

In recent years, a number of structured surveys have been carried out in Dresden, for example, in relation with urban climate adaptation and green spaces, the restoration of an urban watercourse *(Geberbach),* the design of an urban park *(Südpark)* or the upgrading of urban green spaces. The results of all these surveys reflected the public's high regard for biological diversity and the ecosystem services associated with it.

As part of the BIDELIN project, surveys on the perception of urban ecosystem services and their impact on well-being were carried out in the summer months of 2018. A total of 217 Dresden citizens were interviewed in various parks in the city. The assessment of decisions on important ecosystem functions and services, which can, for example, be provided by city parks, shows the high importance of biodiversity-related ecosystem services for the inhabitants (see Table 6). The most frequent choices (94 to 92%) relate to habitat services that parks can provide for plants, animals or bees. Only 'recreation' (99%) and 'improvement of air quality' (96%) were chosen more frequently.

Table 6. Assessment of choices concerning important ecosystem services and functions provided by Dresden city parks (n = 217).

Ecosystem services *and functions*	**"Very important"**
Recreation	99%
Improvement of the air quality	96%
Habitat for plants	*94%*
Habitat for animals	*92%*
Habitat for bees (pollination)	*92%*
Improvement of microclimate (shadowing, cooling, air humidity)	89%
…	…

The planning process of an urban park in Dresden, the so-called *Südpark*, can be considered as another example of the high valuation of biodiversity and thus the significant role of biodiversity in the city. Citizens were involved in this process and have been asked about their wishes for the park design and park facilities. Many aspects related to biodiversity emerged among the most common responses. The most frequently stated demand was the plantation of an orchard meadow with fruit trees, which is a legally protected biotope type in Saxony (see 3.1.4.). Further common demands concerned the creation of 'untouched' natural areas and the maintenance of meadows.

Discussion and conclusion

The aim of the study was to find which role methods to analyse and valuate biodiversity in its different dimensions play in urban nature conservation, city planning and decision-making.

Looking at the practice of planning and decision-making in cities on the example of Dresden and Liberec, we see a broad variety of tools applicable on different scales for various more or less complex tasks. As Chevassus-au-Louis et al. (2009) note, the quantification of biodiversity is a particularly ambitious goal, and the desire to define a simple and unique indicator analogous to the 'ton of coal equivalent' is an illusion, as is any definition of a monetary indicator to account for all of the aspects of biodiversity. The very specific nature of local situations also makes value transfers problematic.

Among the various methods, the classification of land cover data according to the habitat values by Bastian, Schreiber (1999) shows an easily applicable method for obtaining an overview of the spatial distribution of areas and their share with regard to their significance for biodiversity. The 'classical' biodiversity indices (or functions that take into account the relative frequencies of species present at the site), which describe the richness and distribution of species, do not play a major role.

Methods related to species and areas enjoying legal protection, however, have the highest significance for the practitioners in city administration. In daily planning and decision practice, such species and areas absolutely have to be taken into consideration. In principle, adequate methods have proven their worth, although enhancements by better data and knowledge progress are possible.

As Laurila-Pant et al. (2015) note, for all the classical indicators, threshold levels can be set, which dictate the minimum level of biodiversity to maintain. These thresholds serve as the minimum level of biodiversity that society seeks to preserve. Therefore, defining such a threshold represents the first social aspects of the analyses. Overall, these ecological biodiversity indicators are useful quantitative tools for assessing the state of biodiversity, as well as for communicating complex environmental issues in order to integrate them more thoroughly into policy decisions (TEEB 2010).

Despite the numerous advantages of the ecosystem services concept (e.g., De Groot et al., 2010; Ring et al., 2010; Spangenberg, Settele, 2010; Haaren, Albert, 2011; Matzdorf et al., 2019), the application of the ecosystem services concept is (still) not widespread in city planning and decision-making (including in Dresden and Liberec). Of course, meanwhile the concept is more or less broadly accepted, but primarily among scientists and some politicians. There is also increasing consensus that ecosystem services are a basic prerequisite for the quality of life in cities, and that for the future well-being of citizens, the quality and functions of urban ecosystems need to be improved.

To some degree, economic valuation of biodiversity and ecosystem services is used nevertheless, namely for the calculation of economic expenditures of ecosystem management. Thus, nature conservation efforts can be made financially sustainable over longer periods of time, also in cities (Mertz et al., 2007; Seják, Cudlín, 2010; Spangenberg, Settele, 2010).

The knowledge of economic (monetary) values in a strongly economically oriented world can also be helpful for awareness raising and for improving weighing decisions. It can produce a 'wow' effect when one hears that the annual economic value of FFH sites in the city of Dresden (1,901 ha) is 4.6–6.5 million €/a, and that of protected biotopes (1,470 ha) ranges between 3.6 and 5.1 million €/a. We downscaled the data calculated for the entire Natura 2000 network of the EU (annual value 2,447 €/ha (median) –3,441 (mean) (see IEEP, 2002; Kettunen et al., 2009; Gantioler et al., 2010; Bastian, 2013). It is obvious that the benefit of these protected areas exceeds the costs for habitat management (Table 5) by far.

Our preference studies showed the high appreciation of urban biodiversity by the citizens of Dresden. This is in line with indications from literature that reflect a high importance and perception of biodiversity in cities for the citizens (e.g., Botzat et al., 2016; Fischer, Kowarik, 2018).

We conclude that the existing methods of analysing and accounting for biodiversity and habitat services may provide a valuable knowledge base for decision-making, planning, nature conservation and management (not only) in cities. In most cases, only few selected methods are applied, depending on the object of investigation, data availability and costs. Laurila-Pant et al. (2015) argue for more integrative approaches that consider all three perspectives of biodiversity valuation. According to Ring, Schröter-Schlaack (2011), to achieve sustainable solutions in biodiversity conservation we need policy mixes consisting of a broader range of assessment and valuation methods. They should include biophysical ranking methods but also integrated approaches that consider the complexity of ecological, social and economic systems, among them measures of attitudes, preferences and intentions, civic valuation, ecosystem benefit indicators, and negotiations among stakeholders.

References

Albrecht, J., Schumacher, J. & Wende W. (2014). The German impact-mitigation regulation – A model for the EU's no-net-loss strategy and biodiversity offsets? *Environmental Policy and Law*, 44(3), 317–332.

AOPK (2011). *Data about area of protected territories*.http://www.nature.cz/publik_syst2/files/udaje_o_rozloze_biotopu_cr.pdf

Bastian, O. (1998). Landscape-ecological goals as guiding principles to maintain biodiversity at different planning scales. *Ekológia, Bratislava,* 17(1), 49–61.

Bastian, O. (2013). The role of biodiversity in supporting ecosystem services in Natura 2000 sites. *Ecological Indicators*, 24, 13–22. DOI: 10.1016/j.ecolind.2012.05.016.

Bastian, O. & Schreiber K.-F. (Eds.) (1999). *Analyse und* ökologische *Bewertung der Landschaft*. Heidelberg, Berlin: Spektrum Akademie Verlag.

Bastian, O. & Xiao N. (2018). The role of biodiversity in a city. In K. Grunewald, J. Li, G. Xie & L. Kümper-Schlake (Eds.), *Towards green cities. Urban biodiversity and ecosystem services in China and Germany* (pp. 49–55). Cham: Springer International Publishing. DOI: 10.1007/978-3-319-58223-8.

Blume, H.-P. & Sukopp H. (1976). Ökologische Bedeutung anthropogener Bodenveränderungen. *Schriftenreihe* für *Vegetationskunde*, 10, 75–89.

Botzat, A., Fischer, L.K. & Kowarik I. (2016). Unexploited opportunities in understanding liveable and biodiverse cities. A review on urban biodiversity perception and valuation. *Global Environmental Change*, 39, 220–233. DOI: 10.1016/j.gloenvcha.2016.04.008.

Chevassus-au-Louis, B., Salles, J.-M., Bielsa, S., Richard, D., Martin, G. & Pujol J.-L. (2009). *Approche économique de la biodiversité et des services liés aux écosystèmes*. Paris: Rapport du CAS.

City plan Liberec (2019). *City plan of the statutory city of Liberec*. https://www.liberec.cz/cz/radnice/strategie-projekty/uzemni-plan/uzemni-planovani/uzemni-plany-obci/liberec/platny-uzemni-plan/

COM (2011). *Final communication from the commission to the European Parliament, the Council, the Economic and Social Committee and the Committee of the Regions: Our life insurance, our natural capital: an EU biodiversity strategy to 2020*. Brussels: European Commission.

Cudlín, P., Prokopová, M., Francírková, T., Burešová, R., Smrž, T. & Boucníková E. (2005). System NATURA 2000 utilization for purposes of biotope valuation. *Ekológia (Bratislava)*, 24(Suppl. 1), 52–68.

de Groot, R.S., Alkemade, R., Braat, L., Hein, L. & Willemen L. (2010). Challenges in integrating the concept of ecosystem services and values in landscape planning, management and decision making. *Ecological Complexity,* 7(3), 260–272. DOI: 10.1016/j.ecocom.2009.10.006.

Diviaková, A., Kočická, E., Belčáková, I. & Beláňová E. (2018). Evaluation of the significance of linear non-forest woody vegetation in the development of an ecological network. *Applied Ecology and Environmental Research*, 16(2), 1221–1241. DOI: 10.15666/aeer/1602_12211241.

EEC (2007). *Guidance document on the strict protection of animal species of Community interest under the Habitats Directive 92/43/EEC 2007.* http://ec.europa.eu/environment/nature/conservation/species/guidance/pdf/guidance_en.pdf

Fischer, L.K. & Kowarik I. (2018). Urbane Artenvielfalt ist beliebt! Gesellschaftliche Unterstützung für eine vielfältige Natur im besiedelten Bereich. *GIT Labor-Fachzeitschrift*, 12, 24–27.

Gantioler, S., Rayment, M., Bassi, S., Kettunen, M., McConville, A., Landgrebe, R., Gerdes, H. & ten Brink P. (2010). *Costs and socio-economic benefits associated with the Natura 2000 network.* Final Report to the European Commission, DG Environment, Institute for European Environmental Policy/GHK/Ecologic, Brussels.

Geißler, M. (2017). *Bilanzierung der Landschaftspflege im urbanen Raum am Beispiel der Städte Chemnitz, Dresden und Leipzig*. Unpubl. Master thesis, Technical University, Dresden.

Grunewald, K. & Syrbe R.-U. (2013). *Bilanzierung von ausgewählten Leistungen und Anforderungen der Landschaftspflege in Sachsen aus landesweiter Sicht.* Dresden: Sächsisches Landesamt für Umwelt, Landwirtschaft und Geologie.

Grunewald, K., Syrbe, R.-U. & Bastian O. (2014). Landscape management accounting as a tool for indicating the need of action for ecosystem maintenance and restoration – exemplified for Saxony. *Ecological Indicators*, 37(A), 241–251. DOI: 10.1016/j.ecolind.2013.09.014 .

Grunewald, K. & Bastian O. (2017). Maintaining ecosystem services to support urban needs. *Sustainability*, 9(1647), 9. DOI:10.3390/su9091647

Haaren, C.V. & Albert C. (2011). The concepts of ecosystem services and environmental planning: An assessment of the scope for synergies. *International Journal of Biodiversity Science, Ecosystem Services & Management,* 7(3), 150–167. DOI: 10.1080/21513732.2011.616534.

Haber, W. (1998). Das Konzept der differenzierten Landnutzung – Grundlage für Naturschutz und nachhaltige Naturnutzung. Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit (Ed.), *Ziele des Naturschutzes und einer nachhaltigen Naturnutzung in Deutschland* (pp. 57–64). Bonn: BMU.

Haines-Young, R. & Potschin M.B. (2018). *Common International Classification of Ecosystem Services (CICES) V5.1 and Guidance on the Application of the Revised Structure.* https://cices.eu/content/uploads/sites/8/2018/01/Guidance-V51-01012018.pdf

Hofmann, M., Westermann, J.R., Kowarik, I. & Van der Meer E. (2012). Perceptions of parks and urban derelict land by landscape planners and residents. *Urban Forestry & Urban Greening,* 11(3), 303–312. DOI: 10.1016/j.ufug.2012.04.001.

IEEP (ten Brink, P., Monkhouse, C. & Richartz S.) (2002). *Promoting the socio-economic benefits of Natura 2000.* Background Report, Institute for European Environmental Policy (IEEP), European Conference, Brussels, November 28-29, 2002.

IÖR (2019). *The BIDELIN project.* http://bidelin.ioer.eu/en/Kettunen, M., Bassi, S., Gantioler, S. & ten Brink P. (2009). *Assessing socio-economic benefits of Natura 2000 – a toolkit for practitioners.* Brussels: Institute for European Environmental Policy (IEEP).

Kowarik, I. & Körner S. (2005). *Wild urban woodlands. New perspectives for urban forestry*. Heidelberg: Springer.

Landschaftsplan Dresden (2018). *The landscape plan of the capital city of Dresden.* http://www.dresden.de/de/stadtraum/umwelt/umwelt/landschaftsplan/geltender_landschaftsplan.php

Laurila-Pant, M., Lehikoinen, A., Uusitalo, L. & Venesjärvi R. (2015). How to valuate biodiversity in environmental management? *Ecological Indicators*, 55, 1–11. DOI: 10.1016/j.ecolind.20153.02.034.

MA (2005). *Millennium Ecosystem Assessment. Ecosystems and Human Well-Being.* Washington, D.C.: Island Press.

Mace, G.M., Norris, K. & Fitter A.H. (2012). Biodiversity and ecosystem services: a multilayered relationship. *Trends Ecol. Evol.*, 27(1), 19–26. DOI: 10.1016/j.tree.2011.08.006.

Matzdorf, B., Hansjürgens, B., Lienhoop, N., Schröter, B., Schröter-Schlaack, C., Stöcker, U., Suntken, S. & Szallies I. (2019). Erkenntnisse aus dem TEEB-Deutschland-Prozess für die Politikgestaltung in ländlichen Räumen nutzen. Handlungsempfehlungen am Beispiel dreier Politikfelder. *Natur und Landschaft*, 94(1), 21–27. DOI: 10.17433/1.2019.50153657.21-27.

Mertz, O., Ravnborg, H.M., Lövei, G.L., Nielsen, I. & Konijnendijk C.C. (2007). Ecosystem services and biodiversity in developing countries. *Biodivers. Conserv.*, 16(10), 2729–2737. DOI: 10.1007/s10531-007-9216-0

Moyzeová, M. (2018). Inclusion of the public in the natural capital, ecosystem services and green infrastructure assessments (Results of structured interviews with stakeholders of commune Liptovská Teplička). *Ekológia (Bratislava)*, 37(1), 42–56. DOI: 10.2478/eko-2018-0005.

Müller, N. & Abendroth S. (2007). Empfehlungen für die Nationale Strategie zur Biologischen Vielfalt in Deutschland. *Naturschutz und Landschaftsplanung*, 39, 114–118.

Pechanec, V., Machar, I., Štěrbová, L., Prokopova, M., Kilianova, H., Chobot, K. & Cudlín P. (2017). Monetary valuation of natural forest habitats in protected areas. *Forests*, 8, 427. DOI: 10.3390/f8110427.

Riedel, W., Lange, H., Jedicke, E. & Reinke M. (Eds.) (2016). *Landschaftsplanung*. Berlin, Heidelberg: Springer-Spektrum.

Ring, I., Hansjürgens, B., Elmqvist, T., Wittmer, H. & Sukhdev P. (2010). Challenges in framing the economics of ecosystems and biodiversity: the TEEB initiative. *Current Opinion of Environmental Sustainability*, 2(1–2), 15–26. DOI: 10.1016/j.cosust.2010.03.005.

Ring, I. & Schröter-Schlaack C. (Eds.) (2011). *Instrument mixes for biodiversity policies.* POLICYMIX Report, No. 2/2011. Leipzig: Helmholtz Centre for Environmental Research – UFZ. http://policymix.nina.no

Schlüter, H. (1982). Geobotanische Kennzeichnung und vegetationsökologische Bewertung von Naturraumeinheiten. *Archiv* für *Naturschutz und Landschaftforschung*, 22, 69–77.

Schutzgebiete Dresden (2018). *Schutzgebiete in Dresden.* http://www.dresden.de/de/stadtraum/umwelt/umwelt/naturschutz/Schutzgebiete_nach_Naturschutzrecht.php

Schweppe-Kraft, B. (2009). Natural capital in Germany – State and valuation; with special reference to biodiversity. In R. Döring (Ed.), *Sustainability, natural capital and nature conservation* (pp. 193–216). Beiträge zur Theorie und Praxis starker Nachhaltigkeit, Vol. 3. Marburg: Metropolis.

Seják, J., Dejmal, I., Petříček, V., Cudlín, P., Míchal, I., Černý, K., Kučera, T., Vyskot, I., Strejček, J., Cudlínová, E., Cabrnoch, J., Šindlar, M., Prokopová, M., Kovář, J., Kupka, M., Sčasný, M., Šafařík, M., Roušarová, Š., Stejskal, V. & Zapletal J. (2003). *Monetary valuation of the habitats of the Czech Republic.* Prague: Czech Ecological Institute, Ministry of Life Environment of the Czech Republic.

Seják, J. & Cudlín P. (2010). On measuring the natural and environmental resource value and damages. *Studia Ecologica*, 4(2), 53–68.

Spangenberg, J.H. & Settele J. (2010). Precisely incorrect? Monetising the value of ecosystem services. *Ecological Complexity*, 7(3), 327–337. DOI: 10.1016/j.ecocom.2010.04.007.

TEEB (2010). *The economics of ecosystems and biodiversity*. London: Earthscan.

TEEB (2011). *TEEB manual for cities: Ecosystem services in urban management*. London: Earthscan.

Tüxen, R. (1956). Die heutige potentielle natürliche Vegetation als Gegenstand der Vegetationskartierung. *Angewandte Pflanzensoziologie*, 13, 5–42.

UNEP (1992). *The United Nations Convention on Biological Diversity*.

User's Manual for CBI (2012): http://www.cbd.int/authorities/doc/User%27s%20Manual-for-the-City-Biodiversity-Index18April2012.pdf.

Werner, P. & Zahner R. (2009). Biological diversity and cities. A review and bibliography. *BfN-Skripten*, 245.

Werner, P. (2016). Biologische Vielfalt im urbanen Raum. Zusammenhang, Entwicklung und Einflussfaktoren auf Flora und Fauna. *Natur und Landschaft*, 91(7), 314–321. DOI: 10.17433/7.2016.50153400.

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RECREATION IN THE CITY-A PART OF CULTURAL ECOSYSTEM SERVICES

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Abstract

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Cultural ecosystem services (CES) are mainly intangible benefits, obtained by aesthetical and other experiences, recreation, learning and spiritual enrichment, or by the ability to distinguish values. In other words, what makes a service 'cultural' is its 'non-economical' character. CES are ecosystem services with direct impact on the quality of life in urban systems, and they are directly appreciated by inhabitants and visitors of these urban systems (Plieningeret al., 2013; Zulian et al., 2018). In order to satisfy the needs and expectations of the wider urban community, a 'broader portfolio of areas' is needed, which can meet the expectations of different users, from children to older adults. The new methodological approach tested on three model areas in the city of Nitra in the housing estate Chrenová was used to establish the level of benefits provided by existing vegetation areas in urban environment in terms of recreation as a CES. Following data were collected: the quality of vegetation, spatial design, management of vegetation elements, selected environmental aspects, available amenities and architectural elements needed for recreation. In our article, we present the assessment of vegetation in relation to the provision of recreation as a CES. Our results show that the assessed areas can be classified in the category of average to low provision of benefits related to recreation. Vegetation on area P1 is in good condition, mainly the quality of trees and shrubs was assessed as very good (4). Other areas (P2 and P3) assessed provide only low levels of benefits derived from recreation as one of the CES. This was caused by some vegetation deficiencies (mainly related to horticultural and compositional aspects). Methodological approach can be used for different vegetation areas in urban environment and after suitable modifications (e.g. adding other components for assessment) also for different cultural ecosystems services. The results can be used in landscape planning documents or in other types of documents dealing with the quality of vegetation in urban environment.

Key words: CES, recreation, public green spaces, vegetation.

Introduction

Cultural ecosystem services (CES) can be defined according to MA (2005) as the intangible benefits that people derive from ecosystems. CICES (2018) understands them as the characteristics of elements of nature that provide opportunities for people to derive cultural goods or benefits.

An integral part of human life in urbanised environments is vegetation. In the urban environment, we refer to it as urban ecosystems, which enrich the surroundings and life through their benefits. Vegetation as a biotic component serves to balance but also to complement abiotic components. Public greenery is one of the most important elements that provides urban ecosystem services, and it is regarded as a subject that is accessible to all residents. Public greenery represents a piece of nature for all residents (De La Barrera, 2016; Murgaš, 2018). Vegetation areas are an important component of urban green infrastructure, and they provide a wide range of ecosystem services, including cultural services such as recreation and recovery (Daniel et al., 2012; Izakovičová et al., 2017).

According to Špulerová (2006), vegetation has a positive impact in the urbanised environment especially on improving microclimate, ecological stability, elimination of threats to the urban environment and biodiversity in cities, such as climate change or unstable hydrological cycle, on some entities related to sustainable development, for example, environmental education, public health, recreational services, psychological and aesthetic functions.

Material and methods

Research sites

On the left bank of the Nitra River, the Chrenová I housing estate was built between 1964 and 1968, supervised by the architect Michal Maximilian Scheer. The quality of this residential area is also confirmed by the fact that it is inscribed in the UNESCO urban-architectural book. The residential area is dominated by continuous areas of residential greenery around the river and residential buildings. The meandering structure of the low-rise housing structure defines interesting block spaces (Jarabica, 2011).

The research is carried out in the localities of Chrenová I, II and III housing estates in Nitra (Fig. 1). Three areas are selected (P1, P2 and P3), which have the best potential for leisure, sport, recreation and so on.

Methodological approach

The aim of the research was to create a new methodological approach to analysis of CES in terms of recreation in urbanised environment. The quality (value) of elements and factors (quality of vegetation, spatial composition, environmental factors, management and equipment) on selected vegetation areas determines the level of benefit, based on which the provision of recreational services is determined (Turanovičová, Rózová, 2017). This approach builds on already existing CES assessment methods (Piscová et al., 2018). The method is applicable to all vegetation areas in urbanised environments to determine the current status of providing benefits related to CES – specifically recreation. It can be complemented from different perspectives and from different areas and needs (e.g. accessibility). In this article, we deal with the quality of vegetation (trees and shrubs, lawn and flower beds) for recreational use and for determining the level of benefit that vegetation provides to inhabitants.

The methodological approach is based on the work of Bastian et al. (2012) and Burkhard et al. (2009, 2012). The value of ecosystem services is expressed in a relative point scale of 0–5, where zero means 'zero ecosystem capacity to provide the selected ecosystem service', whereas value 5 represents 'very high capacity (quality) of ecosystem'.

The methodological approach includes analyses and assessments of individual vegetation elements on vegetation areas in housing estates. A value-creating process has been created, in which value means quality from a human perspective. A point scale of 0–5 was used to determine the quality of the monitored vegetation elements, where 0 means no quality of vegetation elements and 5 means the highest quality. The quality values have been assigned a benefit rate provided by the vegetation elements for recreation (Turanovičová, Rózová, 2018).

Analysis of vegetation elements quality

The vegetation quality analysis was carried out by assigning numerical values (0–5) to vegetation elements (trees and shrubs, flower beds and grass areas). Numerical values used for tree inventory according to Machovec (1982) were

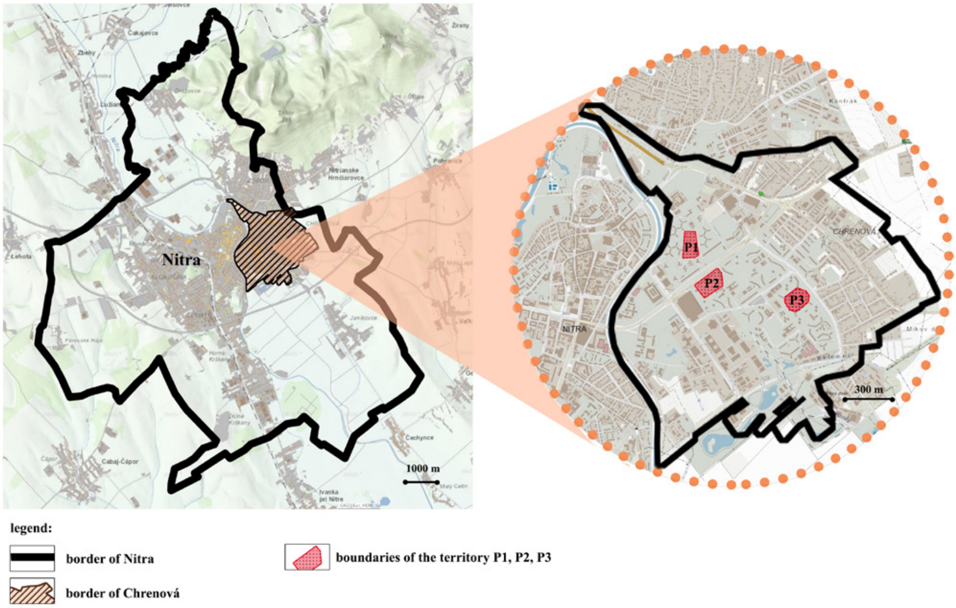

Fig. 1. Research localities.

used for this analysis. The numerical values reflect the quality of analysed vegetation elements. The highest quality trees are assigned a value of 5 and the least quality a value of 1. If there are no trees and shrubs on the area, a value of 0 is assigned. Parameters of *horticultural value* and *health status* were chosen, supplemented by the *composition value* according to the method of Kubišta (2008) adjusted for the needs of our evaluation, where 0 is the vegetation without compositional value, 1 represents the lowest value of the vegetation or elements in the composition, 5 represents the best applicability of given vegetation in the spatial composition. The analytical data about the vegetation elements obtained from the field survey are collected and then used in the evaluation. Individual vegetation elements together form one whole; therefore, they are evaluated by rounded average in individual parameters.

*Assessment of vegetation element*s

The aim of the evaluation is, based on the analytical data that indicated the quality of individual elements, to evaluate the extent of benefits obtained from the monitored area for the needs of recreation within the framework of CES. The level of benefit is manifested in the use of the area for recreational, leisure, sports and other activities. After evaluation, the individual elements are assigned a level of benefit within the framework of CES.

Assigning the level of benefits to quality values of vegetation elements

The analysis of vegetation quality in the monitored areas is carried out by assigning numerical values to vegetation elements (trees and shrubs, lawn and flower beds). The quality of vegetation corresponds to the level of benefit expressed in points. That is, the higher the quality of vegetation on the surface, the higher the utilisation (benefit) rate, in our case the benefit of recreational services. Therefore, a numerical value of 0 is assigned to a zero level of benefit. The highest quality of vegetation (4, 5) is assigned the highest benefit level (3). The quality assessed as 1 and 2 is assigned the value of 1, and the quality level of 3 is assigned a value of 2.

Categories of levels of benefits provided by vegetation to recreational ecosystem service in terms of quality of vegetation elements

The overall benefit levels of the monitored area in terms of vegetation are divided into four categories. It is the sum of the points assigned to benefits from vegetation elements. It expresses the degree to which vegetation provides benefits in terms of the recreational ecosystem service:
***4.** 9-8 points – a high level of benefits related to recreational cultural ES*
- vegetation of the highest quality, representation of all analysed vegetation elements in excellent health condition, vegetation is well placed from the compositional and horticultural point of view, high representation of native species,
***3.** 7-5 points – average level of benefits related to recreational cultural ES*
- vegetation of average quality, one vegetation storey is missing, health condition can be disturbed, but it has the potential to perform its functions after professional treatment, compositional and horticultural aspects have deficiencies that can be treated,
***2.** 4-2 points – low level of benefits related to recreational cultural ES*
- low-quality vegetation, one or two storeys missing, health condition unsuitable, vegetation does not meet the requirements to provide CES,
***1.** 1-0 points – very low level of benefits related to recreational cultural ES*
- unsatisfactory vegetation quality, only one storey can be represented, in poor health condition without compositional and horticultural value, areas very damaged, untreated.

Results

Analysis of quality of vegetation elements in area P1

The vegetation area P1 at Chrenová 1 is located between the streets Ľudovíta Okánika and Lomnická. It is located in the middle of the residential complex. It is attractive for families with children, thanks to the traffic playground (Fig. 2). Relaxation areas, sidewalks and a tavern provide leisure options for different ages. There is a large amount of predominantly deciduous trees. The grass areas are located along the edges of the vegetation area. They are little used. The area does not provide opportunities for a variety of different recreational, sports and leisure activities, which negatively affects visit rate.

The vegetation elements on the P1 area are of high (4) to average quality (3) (Table 1). The highest quality ratings (4) have trees and shrubs. Planted trees were assessed as average quality from the horticultural point of view. The plants are of average value; about half of the area includes native species with typical species and growth attributes. In particular, the health condition can be assessed as particularly good, with high quality (4). The occurrence of diseases and pests is rare; the vegetation is partly dried, concerning up to one-fifth of the vegetation. The trees are appropriately deployed in terms of composition.

In particular, grass areas have deficiencies, especially from the point of view of horticulture and health, having poor quality (2). The lawn is damaged; the growth characteristics are changed for more than 50% of the lawn. Considering health aspects, three-fifth of the area is dry, the damage due to diseases is 25–50%. Flower beds are missing, which negatively affects the assessment.

Analysis of quality of vegetation elements in area P2

The vegetation area P2 is located between the streets Lesná and Lipová. The area is situated near the busy main road Tr. A. Hlinku and shopping centre Centro Nitra. There is a playground and fitness

Fig. 2. Traffic playground.

Fig. 3. Fitness elements.

Fig. 4. Dried lawn.

equipment Street Workout Park Nitra (Fig. 3). The whole area is planted with many predominantly deciduous trees.

Analysis (Table 1) shows that the quality is average (3) to low (2). Trees and shrubs have the highest (4) value in the health category. There is rare occurrence (up to 5%) of diseases and pests in trees; vegetation is partly dried, concerning up to one-fifth of the vegetation volume, the stability is not disturbed. The horticultural value is average (3). Approximately half of the area includes native species with typical species and growth characteristics. From the compositional point of view, the vegetation covers about half of the area, and it has a long-term potential to fulfil various functions, and the potential for further use of the vegetation is 40–60%. Both horticultural value and health conditions of the lawn are of poor quality (2). The lawn is damaged. The growth characteristics are changed on more than 50% of the vegetation. More than three-fifth of the area are dried. After the treatment, it can be left on the surface. The occurrence of diseases is rare; the biggest deficiency is desiccation. The lawn covers 20–30% of the area; the usability of the lawn in the future is 40–60%. The absence of flower beds negatively affects the overall vegetation assessment of the monitored areas.

Analysis of quality of vegetation elements in area P3

The vegetation area P3 on Chrenová III is bordered by Karpatská and

Bajkalská streets. It connects to an important pedestrian communication linking the shopping centre to the former Lipa cinema and housing units. The vegetation is not properly placed in terms of composition, the central grass area is exposed to sunlight and is dry in summer months (Fig. 4).

On the basis of our analysis (Table 1), the vegetation on the area P3 has the biggest deficiencies of all the monitored areas. Trees and shrubs are of low quality (2) in terms of horticulture and composition. The quality of trees is below average, with significant predominance of non-native species (approximately two-third). Vegetation covers only 30% of the area; the usability of vegetation is 20–40%. The health of woody plants is average. Grass areas have a low rating in all parameters (2). The lawn is damaged, weedy on 50–80% and dried, covering 70% of the area

T a b l e 1. Analysis of quality of vegetation elements on areas P1, P2 and P3.

Vegetation elements	Analysis of quality of vegetation elements in different parameters									Average quality of vegetation elements		
	Horticultural value			Health condition			Compositional value					
	P1	P2	P3	P1	P2	P3	P1	P2	P3	P1	P2	P3
Trees and shrubs	3	3	2	4	4	3	4	3	2	4	3	2
Grass areas	2	2	2	2	2	2	4	3	2	3	2	2
Flower beds	0	0	0	0	0	0	0	0	0	0	0	0
Average quality of parameters	2	2	1	2	2	2	3	2	1	-	-	-

Notes: 5 – very high quality; 4 – high quality; 3 – average quality; 2 – low quality; 1 – very low quality; 0 – without vegetation elements.

Quality of vegetation elements in areas P1, P2 and P3

Vegetation in the monitored areas has average (3) to low (2) quality (Fig. 5). Only on area P1, the trees and shrubs are in the category of high quality (4). The quality of individual vegetation elements varies across the areas. The character of planting is similar, the planting comes from the period of construction of the housing estate Chrenová. Coniferous species of *Pseudotsuga menziesii* (Mirb.) that are represented in each of the monitored areas were planted in large quantities. Their planting is not appropriate in terms of climatic zone. The highest quality of trees and shrubs is on the area P1, with the value of 4, meaning 'high quality'. Area P2 has an average quality (3) of planted trees and shrubs. The greatest deficiencies are on the area P3 in terms of species composition and unsuitable compositional distribution.

The lawn has a quality in the range of 3-2, depending on the area. Area P1 has the best rating among the monitored areas. However, it is only average quality. The lawn in areas P2 and P3 is dry. In the area P3, the lawn has an unsuitable ratio to tree vegetation. It covers more than two-third of the area, resulting in sunburn and subsequent drying.

Flower beds are not found on any of the areas. Therefore, we can say that, on the monitored areas, the quality of trees and shrubs is better than the quality of grass areas, area P1

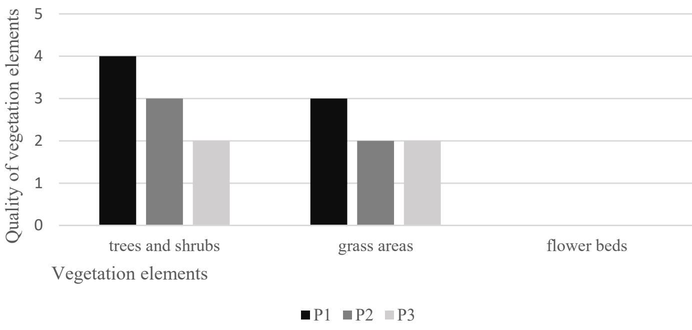

Fig. 5. Quality of vegetation elements on areas P1, P2 and P3.
Notes: 5 – very high quality; 4 – high quality; 3 – average quality; 2 – low quality; 1 – very low quality; 0 – without vegetation.

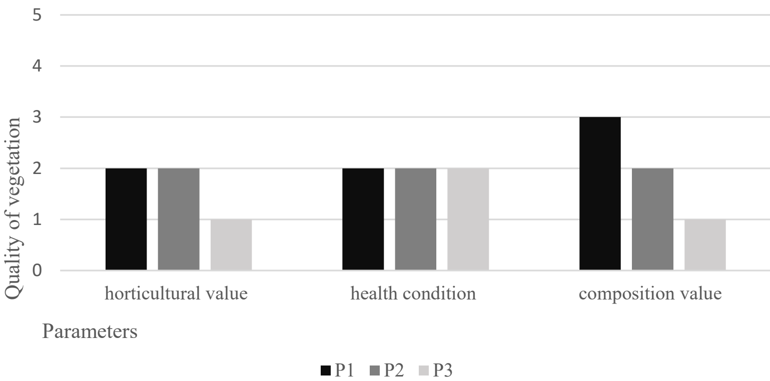

Fig. 6. Quality of vegetation by different parameters in areas P1, P2 and P3.
Notes: 5 – very high quality; 4 – high quality; 3 – average quality; 2 – low quality; 1 – very low quality; 0 – without vegetation.

is better than P2 and P3. The quality of trees, shrubs and lawns on the P3 surface is of poor quality (2) and is the worst of the monitored areas.

Quality of vegetation by different parameters in areas P1, P2 and P3

The monitored areas can be analysed not only from the point of view of the quality of vegetation elements but also from the point of view of the parameters – the horticultural value of the overall vegetation stand, its health condition and the composition value of the stand. Figure 6 shows the comparison of quality of the vegetation in terms of different parameters on three areas. Surfaces are of poor quality (2) in the monitored parameters. An exception is the area P1, which has an average vegetation quality from a compositional point of view (3). Overall, we can say that the quality of vegetation in individual parameters is low.

Assigning the level of benefit to the quality values of vegetation elements and the category of utility rates provided by vegetation on areas P1, P2 and P3

The utility rate provided by the vegetation area based on the quality of the vegetation elements is different on the areas P1, P2 and P3. From Fig. 7, we can see that the best result is on the area P1 (with 5 points), where the utility rate falls into the category 3 – average level of benefits provided by the vegetation. Vegetation is of average quality in terms of all evaluated parameters. The average horticultural value of trees manifests itself in growth characteristics such as habitus and species nativeness. The biggest deficiency is the occurrence of non-native species *P. menziesii* (Mirb.). From a health point of view, vegetation is in a good condition, which is reflected in appearance, fulfilment of functions (microclimate improvement, dust absorption and shadowing) and providing benefits to the population. In terms of the composition, the most suitable from three monitored areas is the area P1, with the greatest future usability.

The areas P2 (number of points: 3) and P3 (number of points: 2) are in category 2 a low level of benefits provided by vegetation. A common drawback in these areas is the inappropriate composition of woody plants. Area P2 has spots with overgrown vegetation and, conversely, P3 has few trees. The absence of flower beds negatively affects the assessment of all three studied areas.

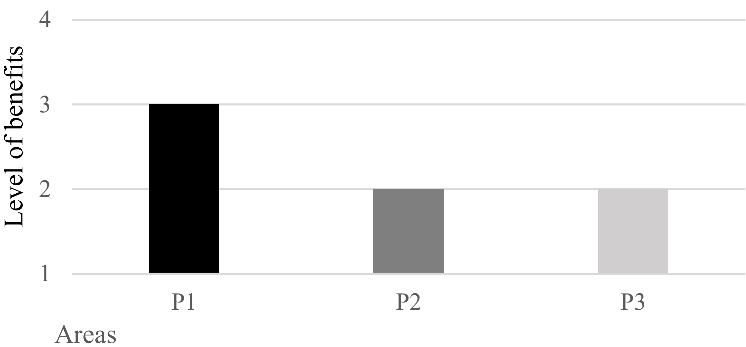

Fig. 7. Level of benefits provided by vegetation on areas P1, P2 and P3.
Notes: 4 – high level; 3 – average level; 2 – low level; 1 – insufficient.

Discussion

Research on CES in urban systems is yet under development and its applicability in planning processes is still a promise (Haase et al., 2014). This motivated us to look for ways of assessing vegetation areas and determining the level of benefits provided to residents in terms of recreational use. In order to understand the needs of the city for the provision of recreational services, it was necessary to do field research and observation of the use of vegetation areas. Environment and social conditions vary widely in urban spaces (Haase et al., 2014). We have verified this idea in the field, and we have found that every part of the city has its specific needs. But what connects all vegetation areas and living spaces is the need to provide residents with benefits in the form of relaxation, recreation, sports and other leisure activities.

The needs of residents in the housing estates perfectly define CES. This is confirmed by the claim that CES can be analysed using methods from different disciplines, including natural and environmental sciences, and also psychology, anthropology, landscape architecture and other social sciences. In other words, what makes the service 'cultural' is precisely its 'non-economic' character.

Similarly, research in Oslo (2015) assessed elements found in public spaces, but their assessment was based on the responses of users and their views on the importance of individual elements (Barton, 2015). It is in this assessment that we see a combination of different disciplines, such as psychology and the human perception of space. The methodology inspired us in the selection of evaluation elements, which was the basis for the creation of a new methodological approach in the field of recreational services in the housing estates in urbanised environments.

Various cities in Slovakia, such as Trnava and Bratislava, and also the winner of the architectural prize CE ZA AR 2018 for the exterior category, public space in Prešov, are examples of the fact that vegetation areas can be functional and modern and provide benefits for various recreational uses. The vegetation areas in the housing estate Chrenová do not meet the necessary quality. This is due to various factors. Vegetation areas are not being restored; only basic measures that do not meet the needs of rapidly changing trends and requirements are being taken. This methodological approach can be a tool for identifying missing elements and for evaluating the level of benefit of existing elements for recreational use in terms of CES. It is necessary to invest in vegetation areas, because these areas are the places where the inhabitants will find peace, relax, rest and necessary recreation during busy work weeks. Reconstruction, time and investment will return back in the form of the quality of life of inhabitants. The value of the quality of life is incalculable and reflected in many other areas.

Conclusion

Cultural ecosystem services are intangible benefits that a person perceives in everyday activity associated with outdoor movement. These services are also about understanding the way people live in urban spaces, and it is important to understand the relationship between man and the urban ecosystem.

On the basis of this knowledge, a new methodological approach to CES analysis was created in terms of recreation in urbanised environment. The quality (value) of elements and factors (quality of vegetation, composition of space, environmental factors, management and equipment), which is assessed on selected vegetation areas, determines the level of benefit, which further determines the degree of provision of recreational services. In this article, the quality of vegetation (trees and shrubs, lawn and flower beds), as well as its impact on recreational use, and the utility rates that vegetation provides to the population were elaborated.

The methodological approach was used on three selected vegetation areas marked P1, P2 and P3 in the internal blocks of individual parts of Chrenová housing estate in Nitra. After assessing the vegetation areas, we have acquired a notion about the diverse quality of vegetation in urbanized environments. Vegetation areas are part of the same housing estate, so the results vary by one degree in maximum. However, a detailed analysis of the vegetation elements (trees and shrubs, lawn and flower beds) pointed to their shortcomings and differences. The best results were observed on P1. Provision of benefits of this vegetation area is at average level (3), which is an acceptable outcome in terms of recreation. The vegetation in the area is in good condition, especially the trees and shrubs have gained high quality (4) in our assessment. Their high quality and fitness in terms of horticultural, compositional and health aspects brings benefits for recreation such as improving the microclimate in the area, aesthetic effects, providing shade in the summer, trapping dust particles and others.

Other assessed areas are at the level of low benefit in the context of cultural ecosystem services. This was caused by several shortcomings in vegetation (mainly the horticultural and compositional aspects). Trees and shrubs are unevenly distributed on P2 and P3, resulting in large shading or sunburn. Thereby the grass areas are affected (drying out and moss overgrowth), and their assessment is decreased. The species composition (horticultural/landscaping aspect) negatively influenced the evaluation result. The overall level of benefit for residents in terms of recreational use is low.

Vegetation quality is the basis for achieving a suitable environment for recreational and leisure activities. The methodological approach can be used for different vegetation areas in urbanised environments and can be used for various cultural ecosystem services after appropriate adjustments (e.g. after complementing other assessment elements).

The results can be used in landscape planning documents or other documents dealing with the quality of vegetation in urbanised areas.

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References

Bastian, O., Haase, D. & Grunevald K. (2012). Ecosystem properties, potentials and services – The EPPS conceptual framework and an urban application example. *Ecological Indicators,* 21, 7–16. DOI: 10.1016/j.ecolind.2011.03.014.

Barton, D.N., Stange, E., Blumentrath, S. & Vagnes Traaholt N. (2015). Economic valuation of ecosystem services for policy. A pilot study on green infrastructure in Oslo. *NINA Report* 1114.

Daniel, T.C., Muhar, A., Arnberger, A., Aznar, O., Boyd, J.W., Chan, K.M.A., Costanza, R., Elmquist, T., Flint, C.G., Gobster, P.H., Gret-Regamey, A., Lave, R., Muhar, S., Penker, M., Ribe, R.G., Schauppenlehner, T., Sikor, T., Soloviy, I., Spierenburg, M., Taczanowska, K., Tam, J. & Von der Dunk A. (2012). Contributions of cultural services to the ecosystem services agenda. *Proc. Natl. Acad. Sci. USA*, 109(23), 8812–8819. DOI: 10.1073/pnas.1114773109.

De La Barrera, F., Reyes-Paecke, S. & Banzhaf E. (2016). Indicators for green spaces in contrasting urban settings. *Ecological Indicators,* 62, 212–219. DOI: 10.1016/j.ecolind.2015.10.027.

Burkhard, B., Kroll, F., Müller, F. & Windshorst W. (2009). Landscapes‘ Capacities to Provide Ecosystem Services – A Concept for Land Cover Based Assessments. *Landscape Online*, 15, 1–22. DOI: 10.3097/LO.200915.

Burkhard, B., Kroll, F., Nedkov, S. & Müller F. (2012). Mapping ecosystem service supply, demand and budgets. *Ecological Indicators,* 21, 17–29. DOI: 10.1016/j.ecolind.2011.06.019.

Haase, D., Larondelle, N., Andersson, E., Artmann, M., Borgstrom, S., Breuste, J., Gomez-Baggethun, E., Gren, Å., Hamstead, Z., Hansen, R., Kabisch, N., Kremer, P., Langemeyer, J., Rall, E., Mcphearson, T., Pauleit, S., Qureshi, S., Schwarz, N., Voigt, A. & Elmqvist T. (2014). A quantitative review of urban ecosystem service assessments: Concepts, models, and implementation. *Ambio,* 43(4), 413−433. DOI: 10.1007/s13280-014-0504-0.

Haines-Young, R. & Potachin M.B. (2018). *Common International Classification of Ecosystem Services (CICES) V5.1 and guidance on the application of the revised structure*. https://cices.eu/content/uploads/sites/8/2018/01/Guidance-V51-01012018.pdf

Izakovičová, Z., Mederly, P. & Petrovič F. (2017). Long-term land use changes driven by urbanisation and their environmental effects (Example of Trnava city, Slovakia). *Sustainability,* 9(9), 1553. DOI: 10.3390/su9091553.

Jarabica, V. (2011). Chrenová I v Nitre. *Urbanita*, 23(1), p. 30−33.

Kubišta, R. (2008). Inventarizačné metódy v záhradnej architektúre. *Acta Horticulturae et Regiotecturae*, 11(Suppl.), 12−15.

Machovec, J. (1982). *Sadovnická dendrologie*. Praha: SPN.

Millennium Ecosystem Assessment (2005). *Ecosystems and Human Well-being*: *Synthesis*. Washington: Island Press. https://www.millenniumassessment.org/documents/document.356.aspx.pdf

Murgaš, F. & Klobučník M. (2018). Quality of life in the city, quality of urban life or well-being in the city: conceptualization and case study. *Ekológia (Bratislava)*, 37(2), 183−200. DOI: 10.2478/eko-2018-0016.

Piscová, V., Hrnčiarová, T., Hreško, J., Dobrovodská, M., Izakovičová, Z., Izsoff, M., Kalivoda H., Kanka, R., Kenderessy, P., Považan, R., Roháč, J., Slabejová, M., Slivinský, J., Špulerová, J., Švajda, J., Topercer, J., Vlachovičová, M., Borovský, I. & Hurta V. (2018). *Využívanie vysokohorskej krajiny a jeho dôsledky na zmenu prostredia (na príklade Tatier a Nízkych Tatier)*. Bratislava: Veda, vydavateľstvo SAV.

Plieninger, T., Dijks, S., Oteros-Rozas, E. & Bieling C. (2013). Assessing, mapping, and quantifying cultural ecosystem services at community level. *Land Use Policy,* 33, 118−129. DOI: 10.1016/j.landusepol.2012.12.013.

Špulerová, J. (2006). Funkcie nelesnej drevinovej vegetácie v krajine. *Životné Prostredie*, 40(1), 37−40.

Turanovičová, M. & Rózová Z. (2017). Metodické východiská hodnotenia kultúrnych ekosystémových služieb v urbanizovanom prostredí. Životné *Prostredie*, 51(4), 232−239.

Turanovičová, M. & Rózová Z. (2018). Vplyv kvality manažmentu na úžitok kultúrnych ekosystémových služieb. *Geografické Informácie*, 22(2), 315−326. DOI: 10.17846/GI.2018.22.2.315-326.

Zulian, G., Stange, E., Woods, H., Carvalho, L., Dick, J., Andrews, Ch., Baró, F., Vizcaino, P., Barton, D.N., Nowel, M., Rusch, G.M., Autunes, P., Fernandes, J., Ferraz, D., Ferreira dos Santos, R., Aszalós, R., Arany, I., Czúcz, B., Priess, J.A., Hoyer, Ch., Bürger-Patricio, G., Lapola, D., Mederly, P., Halabuk, A., Bezák, P., Kopperoinen, L. & Viinikka A. (2018). Practical application of spatial ecosystem service models to aid decision support. *Ecosystem Services*, 29(Part C), 465−480. DOI: 10.1016/j.ecoser.2017.11.005.