Ekológia (Bratislava) Vol. 38, No. 4, p. 301–317, 2019
DOI:10.2478/eko-2019-0023

LANDSLIDING AS A LIMIT TO POSSIBLE TERRITORIAL DEVELOPMENT IN THE KYSUCE REGION

MÁRIA BARANČOKOVÁ, PETER BARANČOK

Institute of Landscape Ecology, Slovak Academy of Sciences, Štefánikova 3, P.O. Box 254, 814 99 Bratislava, Slovak Republic; e-mail: maria.barancokova@savba.sk, peter.barancok@savba.sk

Abstract

Barančoková M., Barančok P.: Landsliding as a limit to possible territorial development in the Kysuce region. Ekológia (Bratislava), Vol. 38, No. 4, p. 301–317, 2019.

The growing development of settlements in mountainous areas and their sustainable development constantly requires new approaches to assess the land in terms of occurrence of landslides. The flysch zone, where the monitored area is located, is one of the most landslide prone areas in Slovakia. Landslides respond sensitively to the quality of the individual factors that form the landscape and to the change of natural conditions. Their occurrence is a geo-barrier that reduces or totally prevents the use of natural environment and negatively affects the life of population and territorial development. The reason for the increased hazard of landslides is not only demographic pressure on territories, but also its poor management. Consistent spatial planning addresses not only the spatial layout but also the functional use of the territory. Landslides represent one of the limits of land use. This study is based on the assessment of landsliding as a limit to possible territorial development. The input parameters for the assessment were elements of the current landscape structure (built up structure, forest stands, transitional woodland-shrubs, traditional agricultural land, permanent grasslands and arable land) and occurring landslides (active, potential and stabilized). On most of the determined elements of the landscape, landslides occur on about a quarter of their area. They have a smaller share only in areas of mixed forests, built up areas and have the smallest share on arable land. Potential landslides have the largest proportion on all landscape elements. They occupy the largest areas on coniferous forests (1578.93 ha) and on permanent grasslands (741.33 ha). By evaluating the overall endangerment of the area by landslides according to the degree of threat, we found that the greatest threat of landslides is in the Skalité and Svrčinovec cadastral areas, the smallest threat is in the Čadca cadastral area. In addition to the danger of landsliding in the individual elements of the landscape, we have also set limits for its development. Spatial planning limits have been divided into two categories according to the sectors they affect the most: limiting the development of an area assigned for residential building, or restricting the development of an area designed for agricultural and forestry purposes.

Key words: landsliding, territorial development, current landscape structure, flysch area, territorial stability.

Introduction

More than 15,000 of the potential landslides (which is 63% of all landslides in Slovakia), which cover an area of about 1620 km^2, have been registered in Slovakia by regional research. These landslides are mainly concentrated in flysch highlands, in the intra-mountain basins and at the edges of the young volcanic mountains (Malgot, Baliak, 2002).

The Kysuce region is characterized by the occurrence of various geodynamic phenomena, which can be understood as a geo-barrier reducing or completely hampering the use of the natural environment and negatively affecting the life of society and territorial development. Each slope has some degree of stability. Its reduction is due to the impact of a natural factor, for example, precipitation and temperature anomalies, weathering of rocks due to exogenous agents, gradual slope loading, uneven slope destruction, seismic shocks, slope inclination, flow erosion, and so on. In addition to natural factors, human activity also plays an important role.

The further development of settlements in mountainous areas with hilly character and their sustainable development requires new approaches to the studies of landslide threats (Bathrellos et al., 2009; Mancini et al., 2010; Guillard, Zêzere, 2012). In the last decades, occurrence of landslides has been constantly increasing in the world. The reason for this increased risk is not only demographic pressure on territories, but also its poor management. Some countries have already made progress in developing the management practices of population growth and minimizing the risks associated with them. Geographic Information Systems (GIS) and Earth Remote Sensing have become integral tools for assessing natural hazards. In addition, GIS is a useful tool for spatial analysis of landslides and for mapping sensitivity of the territory to landslides (Bathrellos et al., 2009; Lan et al., 2004). Improving urban planning and development by zoning of risks and hazards using new technologies and mathematical modeling, while working on the field, is also an important part and is dealt by Cascini et al. (2005). An important parameter for predicting and managing natural disasters is the assessment of landslide hazards. It is also a necessary step for natural and urban planning in government policies around the world (Lekkas, 2000, Carrara et al., 2003; Nowak, Tokarczyk, 2013). The legal and economic consequences resulting from changes of the local spatial plan related to the qualification of landslide areas are analyzed in Bydłosz, Hanus (2013). Landslide areas have limited use for new investments, including construction of buildings, as they are threatened by geological and gravitational slope forces. Spatial planning is a key element in the sustainable development of each country.

An important task in the territorial development is to assess the security of the territory for its development, where it is necessary to consider the amount of input data and factors that can significantly affect these data, especially time and place (Řehák et al., 2013). The evaluation is based on promoting a proactive approach to environmental safety.

Elaboration of maps of areas prone to landslides may be an auxiliary material in the urban planning process in the regions (Mihai et al., 2014; Barančoková, Kenderessy, 2014). This process is important because maps of areas prone to landslides provide a simple tool to inform and control planning and decision-making processes in relation to the new development of proposals within an existing built-up area.

Material and methods

This study is based on the assessment of landsliding as a limit to possible territorial development. The input parameters for the evaluation were elements of the current landscape structure and the occurring landslides. Elements of current landscape structure were processed by combined method, visual interpretation of aerial orthophomaps, field verification and subsequent processing in GIS environment. The following elements were processed (Fig. 1):

a. Built up area/structure, resp. settlement – is a complex system in which the settlements of various functions and sizes act distributed in its territory. At present, the notion of the settlement structure is often identified with the structure of municipalities, which form the basic territorial element of public administration and, at the same time, the self-governing community of residents permanently residing in their territorial district. It occupies 4.3% of the area.

b. Forest stands – include coniferous, deciduous and mixed forests. Forest stands occupy 43.2% of the territory, the largest representation have coniferous forests, which cover up to 39.7% of the territory.

c. Transitional woodland-shrub – represented by young forest trees planted after excavations or calamities, natural developmental forest formations, shrubby formations on abandoned meadows and so on. They occupy 13.3% of the territory in the monitored area.

d. Permanent grasslands (PG) – include meadows and pastures which are a very important phenomenon in the agricultural landscape. They can be characterized as semi-natural, secondary grass-herb formations created in the primary forest zone by human activity (cutting, burning, intentional grazing and mowing) and at the same time are the living space of a number of microorganisms and animals belonging to this complex natural community. They occupy 21.2% of the area.

e. Traditional agricultural landscape (TAL) – reflects the long-term interrelation between man and country and is represented by strip fields alternating with meadows and pastures, separated by buffer strips or scrub vegetation. It occupies 15.1% of the area.

f. Arable land – in the form of large intensively managed areas (large-scale arable land) is situated in the lower parts of the slopes or on the floodplains of the main streams of the area. It occupies only 1.7% of the area.

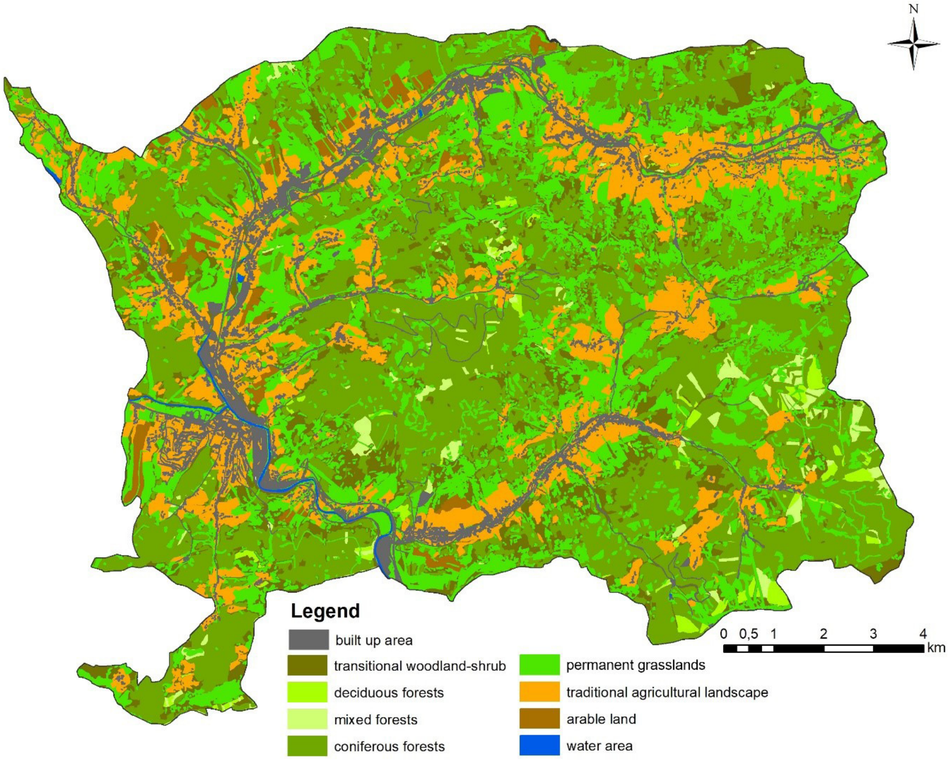


Fig. 1. Current landscape structure.

The vector layer of occurring landslides was processed on the basis of the State Geological Institute of Dionýz Štúr database (Šimeková et al., 2006; updated in 2013) and field vitrificated (Fig. 2). According to the activity, the landslides were divided into three categories: active, potential and stabilized. The active (live) landslide is one that is currently moving. If movement is currently calm, but the causes of its formation can be restored under appropriate conditions, then it is a potential (temporarily calm) landslide. In a stabilized (permanently calm – inactive) landslide, the causes of movement have ceased to exist or have been removed by human intervention. Their significance is currently lower from a societal point of view, but their re-activation in connection with various activities cannot be eliminated, for example, construction of new buildings. Monitoring at these sites is predominantly maintenance-related, but if necessary, its range and frequency may increase and sites may be upgraded to a higher significance category. Vector layer synthesis was processed by the union tool in ArcGIS 10.3.

The next step in the evaluation was to determine the overall threat to land by landslides. In this assessment, the basis was the representation of landslides in individual cadastral areas and the representation of landslides in individual elements of the landscape structure (expressed as a percentage). On the basis of this, we have created three levels of threat: 1 – low level of landslide threats, 2 – medium level of landslide threats and 3 – high level of landslide threats. To assess the risk of landslides in individual cadastral areas, a following scale was used: threat level 1 – interval <0%, 2%); threat level 2 – interval <2%, 6%); threat level 3 – above 6% resp. interval <6%, 14%>. To assess the risk of landslides in individual elements of the landscape structure, a following scale was used: threat level 1 – interval <0%, 12%); threat level 2 – interval <12%, 24%); threat level 3 – above 24% resp. interval <24%, 36%>.

The last step in the evaluation was to determine the limits of spatial planning resulting from the natural conditions of the monitored area, the character of the elements of the current landscape structure and the occurrence of landslides.

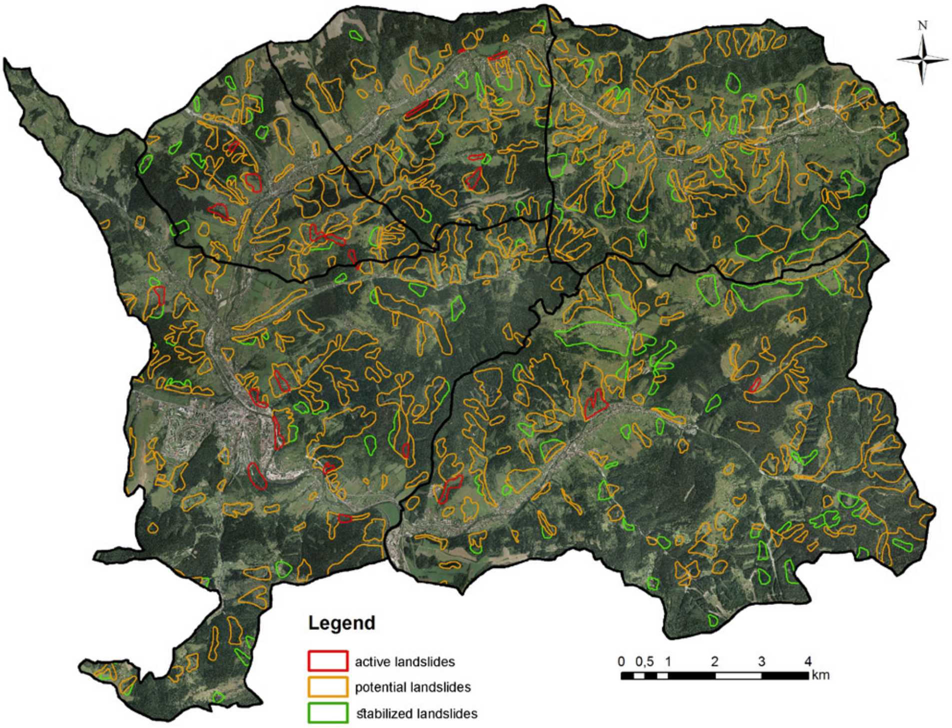


Fig. 2. Extension of individual landslides in the monitored area (processed according to Šimeková et al., 2006 and updated in 2013).

Study area

The monitored area of the Kysuce region is located in the northern part of Slovakia (Fig. 3). It consists of five cadastral areas (CAs) (Čadca, Oščadnica, Svrčinovec, Čierne and Skalité) and is characterized by the diversity of abiotic and biotic components of the landscape. Based on the classification of Slovakia into geomorphological units (Mazúr, Lukniš, 1986), the area belongs to three territories: Central Beskids with units: Kysucká vrchovina Uplands and Kysucké Beskydy Mts, Western Beskids with units: Moravian-Silesian Beskids, Jablunkovské medzihorie Mts and Turzovská vrchovina Uplands and Slovak-Moravian Carpathians with unit Javorníky Mts.

The whole monitored territory has an area of 185.14 km^2. The most extent geomorphologic unit in the territory is Kysucké Beskydy Mts, which occupy up to 100 km^2 and the Jablunkovské medzihorie Mts with an area of 52 km^2. Hilly to mountainous character of the landscape dominates with the altitude of 500 m to 1236 m a.s.l. Slopes with inclination from 12° to 17° (36.8%) and from 7° to 12° (26.1%) cover the largest area. From the geological aspect, the area consists of flysch, of which, the largest representation has the Vsetín member (Bystrica-type claystones, sandstones with glauconite, arcoses and conglomerates) with an area of 75.5 km^2, Oščadnica member (green-gray shales, arcose-, quartzy- and graywacky sandstones) with an area of 14.5 km^2 and Kýčera member (graywacky sandstones, locally shales) with an area of 12.4 km^2. Dystric cambisol covers the largest area – 44.0% of the area and modal cambisol covers 31.1% of the area. The most widespread are forests and woodland-shrubs covering up to 56.5% of the area, of which coniferous forests cover 39.7% of the area. Agricultural land covers 38.0% of the area. In terms of precipitation, the area belongs to a humid climate zone. Average annual rainfall sums from Čadca, between 1994 to 2016 reach 700–1110 mm. The highest average monthly precipitations occur from May to July. Between 2000 to 2017, the monthly precipitation in July was 130 mm.

There are up to 475 registered landslides in this area, which make this area very vulnerable in terms of environmental hazards occurrence. Landslides occupy almost ¼ of the area (44.14 km^2). Most of them are potential landslides, 331 (35.52 km^2), then stabilized landslides, 124 (7.54 km^2) and 20 active landslides (1.08 km^2). Potential landslides cover 80% of the area of all landslides. The largest potential landslides (with an area of 1.5 km^2 and 0.9 km^2) are located in the Oščadnica village.

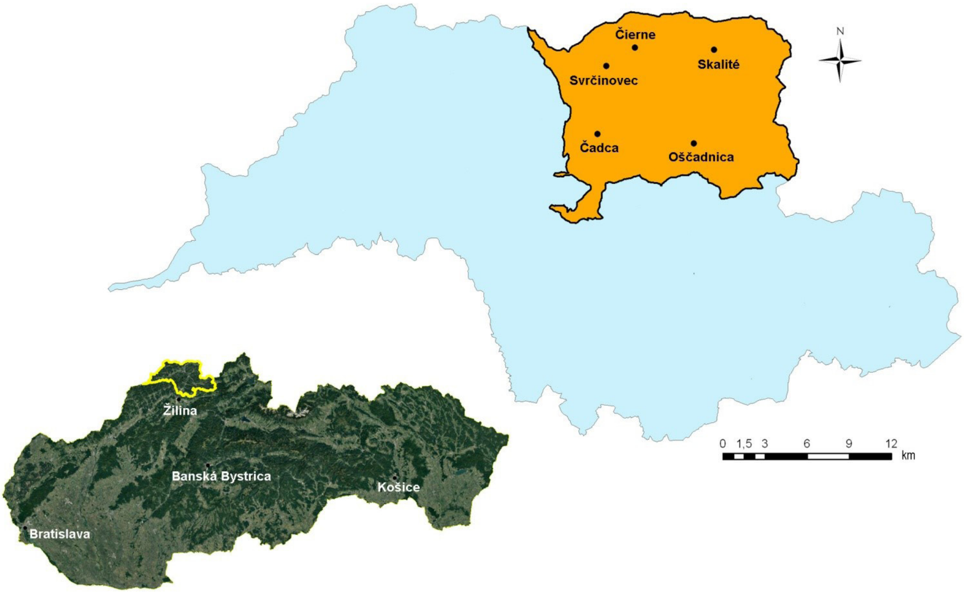


Fig. 3. Localization of study sites.

Results and discussion

Spatial planning addresses the spatial layout and functional use of land. Landslides represent one of the limits of land use. They determine the coordination of activities affecting the environment, ecological stability, territorial development and landscape formation in accordance with the principles of sustainable development. Spatial planning creates the preconditions for the permanent consistency of all activities in the area, with particular regard to the environment cultivation. The goal of spatial planning is to create preconditions for sustainable development through a consistent and comprehensive solution of the spatial layout of the territory and the functional use of the territory.

a) Built up area (structure)

The continuous built up area is concentrated in the monitored territory mainly along the Kysuca, Čierňanka, Skaličianka and Oščadnica watercourses. Its total area is 798.52 ha. 12.02% (95.98 ha) of buildings are affected by landslides, of which 9.99% are potential landslides, 0.63% active landslides and 1.40% stabilized landslides (Table 1). Family houses (6%), sheds (2%), garages (0.5%) and cabins (0.6%) are most endangered by landslides. The most active landslides within the built up area are in the Čadca cadastral area (CA), where they occupy 3.19 ha. Built up areas in Svrčinovec CA are most threatened by potential landslides, on the area of 17.82 ha (Fig. 4). Of the total landslide area (4414.26 ha), 2.17% of landslides are in the built up areas.

b) Forest stands

Coniferous forests have the largest representation of all the forest stands (8000.42 ha), covering 7343.07 ha. There are potential landslides on 21.45% of the forest area (1687.51 ha). The stabilized landslides cover 300.33 ha and active landslides cover 20.07 ha. In coniferous forests, potential landslides (1578.93 ha) cover the largest area, accounting for 78.64% of all landslides in forest stands. Stabilized landslides in coniferous forests occupy 283.88 ha and active landslides cover 20.07 ha. Deciduous and mixed forests have less representation in the area. Mixed forests cover an area of 430.92 ha and deciduous forests occur on an area of 226.43 ha. Potential or stabilized landslides have been recorded on these sites, as shown in Table 1.

Forest stands have the largest representation in the Oščadnica CA (3027.54 ha) and Čadca CA (2406.32 ha), and they cover the smallest area in the CA of Svrčinovec (only 521.94 ha). In terms of the extension of landslides in the monitored area according to the cadasters and individual types of forest stands, it can be stated that the largest occurrence have potential landslides in coniferous forests (Fig. 5). Potential landslides in coniferous forests in the Oščadnica CA have the largest representation, where they are on 522.92 ha (which is up to 31% of all potential landslides in the forests of the area) and Čadca CA, where they occupy 445.29 ha (26% of all potential landslides in the forests of the area). They are less represented in the Skalité CA (292.20 ha, which is 17%), Čierne CA (209.17 ha, 12%) and at least

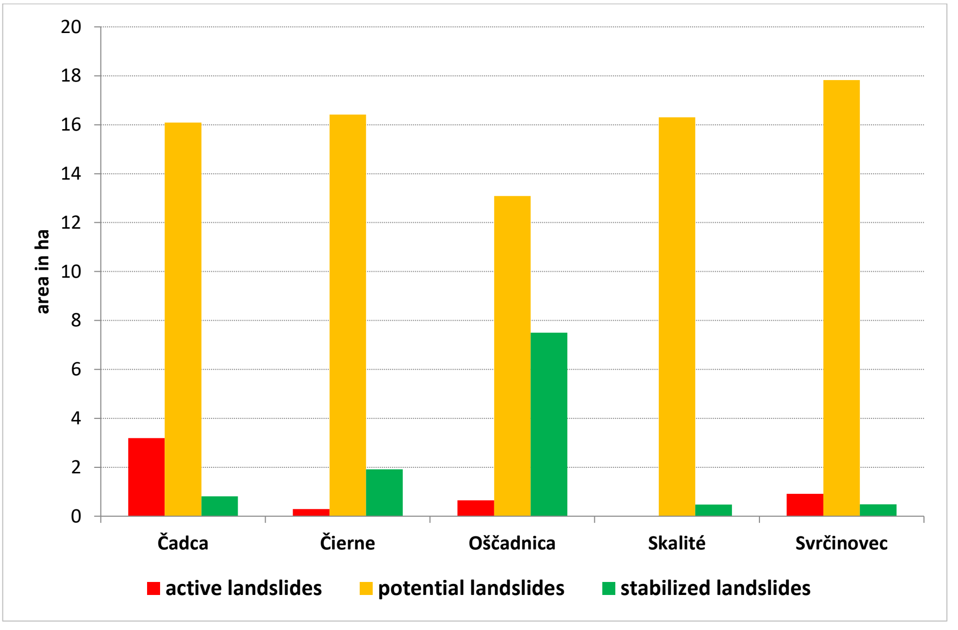


Fig. 4. Representation of landslides in the built up areas of individual cadastral areas.

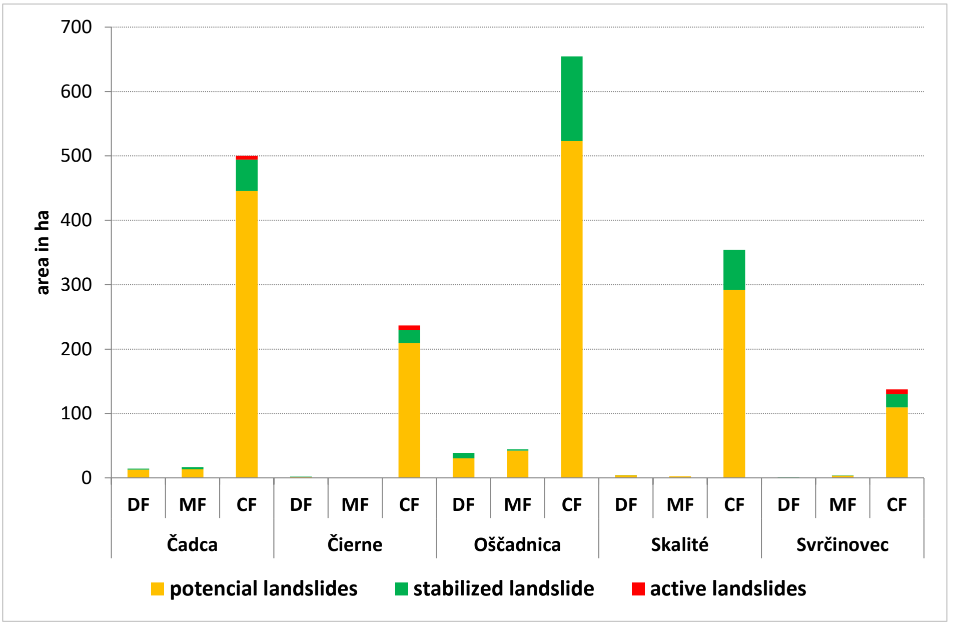


Fig. 5. Representation of landslides in individual types of forest stands in cadastral areas (DF – deciduous forests; MF – mixed forests; CF – coniferous forests).

Svrčinovec CA, where they occupy 109.35 ha (which is 6% of all potential landslides in the forests of the area).

In deciduous and mixed forests, potential landslides have a much smaller occurrence, for example, in Oščadnica CA, they occur in the deciduous forests on an area of 30.21 ha and in mixed forests on an area of 42.39 ha, which together represents 4.3% of all potential landslides in the forests of the area. In Čadca CA, potential landslides are represented in deciduous forests on an area of 12.74 ha and in mixed forests on an area of 13.17 ha, which makes up 1.5%. The other CAs – Čierne, Skalité and Svrčinovec have only a very small occurrence of potential landslides in deciduous and mixed forests (Fig. 5).

A similar situation is also in the case of distribution of stabilized landslides in individual forest types. The largest representation is in the Oščadnica CA., where they occupy 131.75 ha (which represents almost 44% of all stabilized landslides in the forests of the area). They are still quite often found in the forests of Skalité and Čadca CAs, less represented in the CAs Svrčinovec and Čierne.

Active landslides have the least representation in the forests of the monitored area. In the territory of the cadasters of Oščadnica and Skalité, they were not recorded at all, and in the territory of the cadasters of Čadca, Čierne and Svrčinovec, they were only in the areas of coniferous forests (Fig. 5). There they cover the areas of 5.62 ha (Čadca), 7.10 ha (Čierne) and 7.36 ha (Svrčinovec), which are small areas compared to other landslides.

c) Transitional woodland-shrub

Transitional woodland-shrubs cover 2463.21 ha in the monitored area, of which 24.66% are located on landslides. The largest area is occupied by landslides on the areas of woodland-shrubs in the Oščadnica CA, where all three types of landslides occupy up to 171.10 ha, which is 28% of landslides on woodland-shrub areas. It is similar in the Čadca CA, where landslides cover 161.65 ha (26%) and in Skalité CA, where landslides cover 146.91 ha (24%). In other areas of Čierna and Svrčinovec, there are fewer landslides (Fig. 6).

Potential landslides are most widespread in the Čadca and Oščadnica CAs (126.98 ha and 126.42 ha). The same applies to active landslides, where the largest area occupies CAs of Čadca (12.85 ha) and Oščadnica (7.20 ha). Active landslides were not recorded in Skalité CA (Fig. 6).

d) Traditional agricultural landscape

In the monitored area, the elements of traditional agricultural landscape (TAL) occupy 2802.72 ha and landslides cover 718.21 ha in total (which is 25.63% of their area – a quarter of their area). Overall, the largest area of all landslides is in Oščadnica CA (225.17 ha, which represents 36% of landslides in a given land unit), it is slightly less in Skalité CA (188.26 ha, 26%) and Čadca CA (160.32 ha, 22%).

Active landslides occupy a total of 25.65 ha, of which most are in Oščadnica CA (8.05 ha), Čadca CA (7.95 ha) and in Svrčinovec CA (7.35 ha). Potential landslides have the largest share in the CAs Skalité (166.68 ha) and Čadca (140.89 ha). Stabilized landslides occupy the largest area in Oščadnica CA (88.30 ha) (Fig. 7).

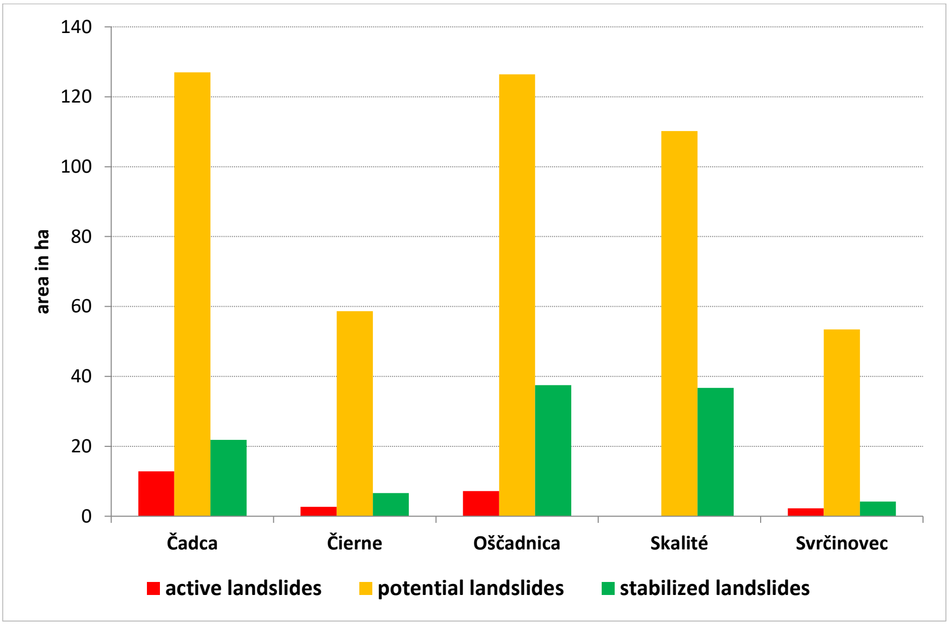


Fig. 6. Representation of landslides in the areas of woodland-shrub in individual cadastral areas.

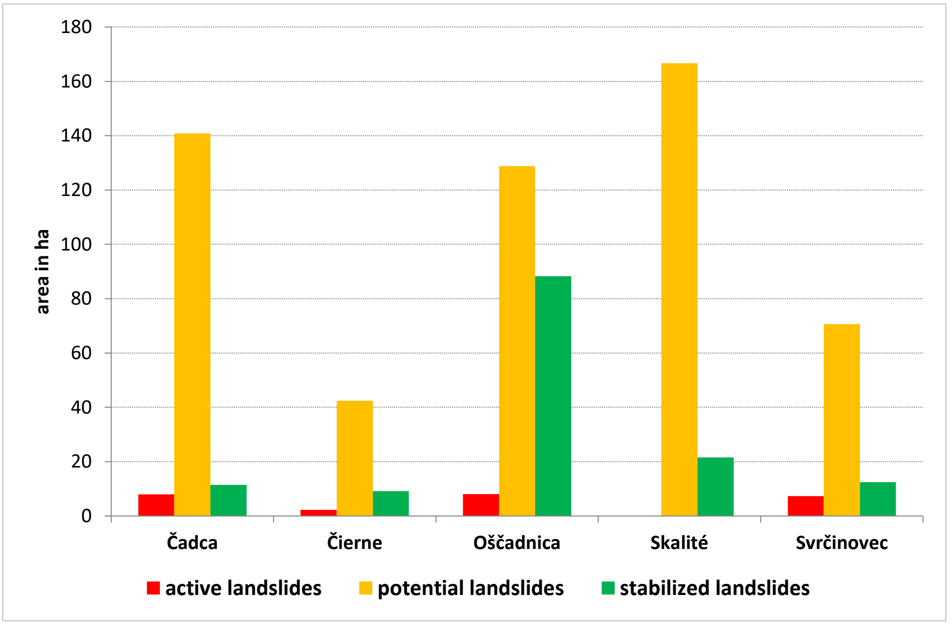


Fig. 7. Representation of landslides in the areas of traditional agricultural landscape elements in individual cadastral areas.

e) Permanent grasslands

Permanent grasslands (PG) cover an area of 3914.86 ha, of which 965.76 ha are disturbed by landslides (which is 24.68% of the area of PG in the territory). Overall, the highest rate of landslides is in Skalité CA, where landslides occur on 289.22 ha, in CA Oščadnica landslides cover 275.75 ha and in Čadca CA landslides are on 188.62 ha.

Potential landslides represent the largest area. Potential landslides affect 741.33 ha (which is almost 77% of the total landslide area of PG), most of which are in CAs of Skalité (216.28 ha) and Oščadnica (192.64 ha). Stabilized landslides cover 192.44 ha (20% of the total landslide area on PG), with the largest area in CAs of Oščadnica (73.24 ha) and Skalité (72.94 ha). By active landslides, 31.99 ha are disturbed (3% of the total landslide area on PG). The largest active landslides are in Svrčinovec, Čadca and Oščadnica CAs, having the areas of 10.52, 10.42 and 9.88 ha, respectively.

f) Arable land

Arable land occupies the smallest area of all monitored landscape elements in the territory, only 311.36 ha. The landslides occur on an area of 18.93 ha of arable land, which is only 6.08% of the area of arable land and there are mainly potential landslides. They occupy the largest area in the CAs Čierne (8.51 ha) and Svrčinovec (4.25 ha).

Overall evaluation of landslides representation and evaluation of threat in the monitored area

Landslides occupy about a quarter of their area in most of the selected landscape elements. They have a smaller share only in the areas of mixed forests, built up areas and the smallest share in the areas of arable land (Table 1, Fig. 10). However, the largest landslides occur in the areas of coniferous forests, smaller areas occupying the sites of PG, TAL and transitional woodland-shrubs. The lowest proportion and share have landslides on arable land.

Potential landslides cover the largest areas in all the selected landscape elements, where the largest areas occupy the coniferous forests (1578.93 ha) and permanent grasslands (741.33 ha). By evaluating the overall endangerment by landslides according to the degree of threat, we found that the greatest threat of landslides is in the Skalité and Svrčinovec CAs, the lowest threat is in the Čadca CA (Table 2).

The biggest threat of landslides is in forest stands and permanent grasslands, while the transitional woodland-shrubs and TAL are slightly less endangered. The least endangered is the arable land in all monitored cadasters (Table 2).

Forests in the monitored area occupy the largest area and the natural forest stands were most often preserved in localities that were not suitable for intensive use (e.g., for development or agricultural use) in terms of natural conditions and processes. Therefore forests cover the largest area of landslides area. Given the persistent threat of landslides, there is no expectation that the nature of their use will change.

In the past farming in Kysuce was dominated by sheep and cattle breeding and the associated use of meadows and pastures. Arable land has always been less represented and

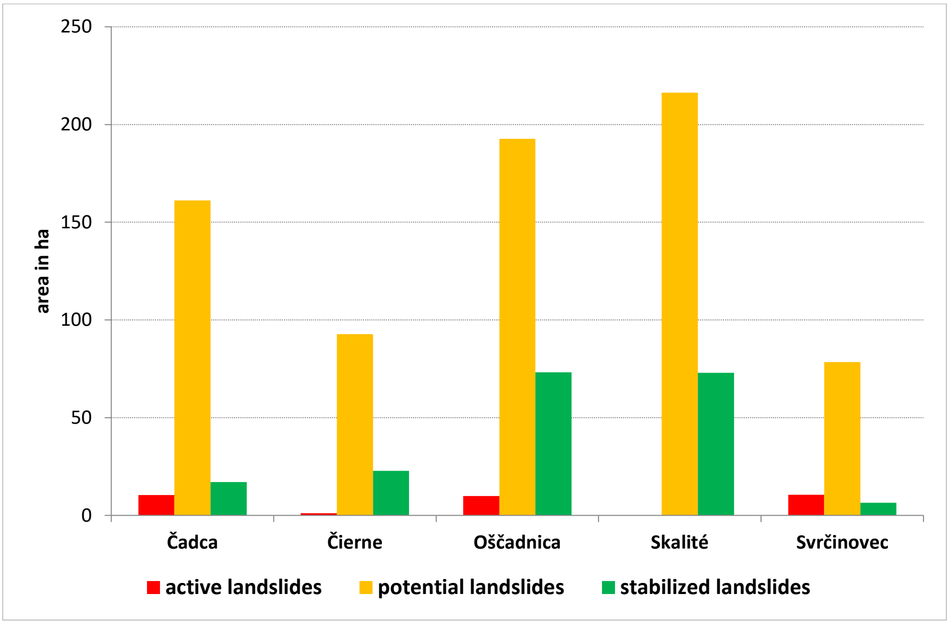


Fig. 8. Representation of landslides on permanent grasslands.

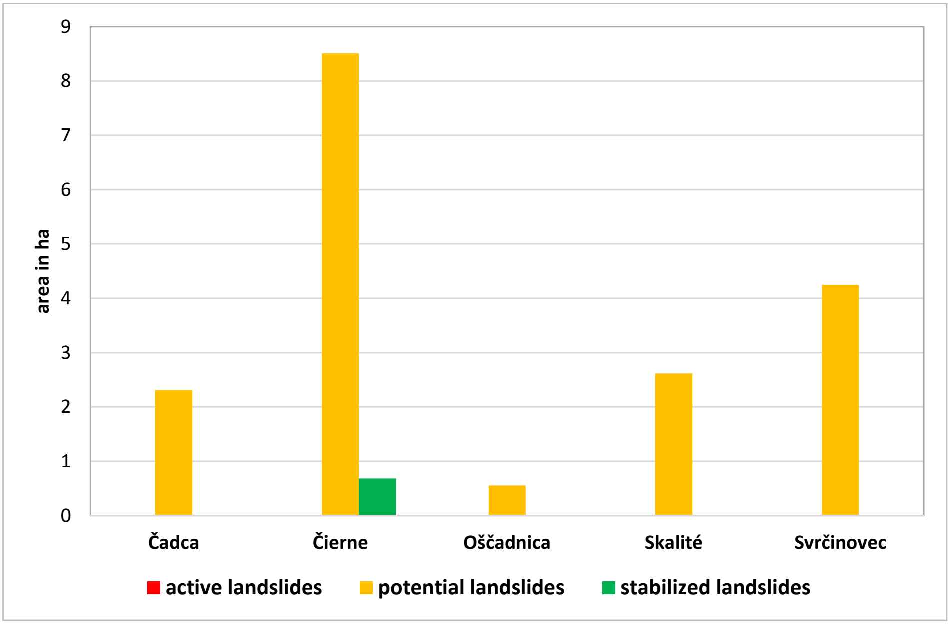


Fig. 9. Representation of landslides on arable land.

Table 1. Representation of individual landslides in the area.

Elements of landscape structure	**Element area (ha)**	**Active landslides (ha)**	**Potential landslides (ha)**	**Stabilized landslides (ha)**	**Sum of landslides (%) ***
Built up area	798.52	5.05	79.73	11.20	**12.02**
Deciduous forests	226.43	0	46.89	10.63	**25.40**
Mixed forests	430.92	0	61.69	5.82	**15.67**
Coniferous forests	7343.07	20.07	1578.93	283.88	**25.64**
Transitional woodland-shrubs	2463.21	24.99	475.65	106.83	**24.66**
Permanent grasslands	3914.86	31.99	741.33	192.44	**24.68**
Traditional agricultural landscapes	2802.72	25.65	549.51	143.05	**25.63**
Arable land	311.36	0	18.24	0.69	**6.08**
Others	222.91	not evaluated			
SUM	**18514.00**	107.75	3551.97	754.54	**24.14**

Note: * – reflects the share of landslides in % of the total area of a given landscape structure element.

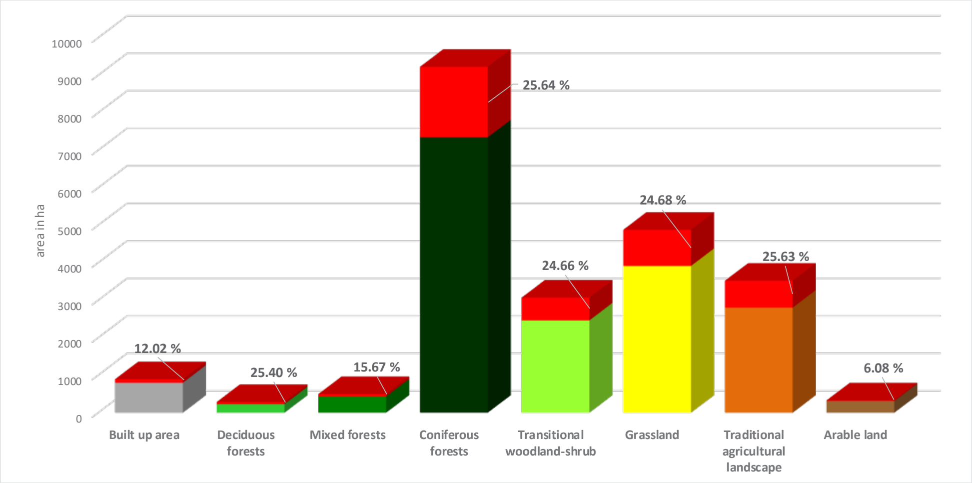


Fig. 10. Overall representation of landslides in the area.

crop production was focused on potato growing. Due to the hilly character of the area, small terraced fields prevailed. Cereal cultivation has always had a secondary position due to the climatic conditions of the territory. That is why PG dominates in the monitored area on agricultural land and many TAL areas have been preserved up to now. At the time of intensive use of PG and TAL elements, man also took care of their protection against landslides (adjustment and stabilization of slopes, diversion of running water from sites, etc.). After the abandonment of farming in these areas, the influence of natural processes prevailed and the factors that activate the landslide processes often occur in these areas.

T a b l e 2. Endangerment of the area by landslides.

Cadastral area (extent)	**Monitored element**	**Built up area**	**Forest stands**	**Transit. woodland-shrub**	**PG**	**TAL**	**Arable land**
Čadca (5679 ha)	area (ha)*	19.98	531.23	161.65	188.62	160.33	2.31
	share ** (%)	0.35	9.35	2.85	3.32	2.82	0.04
	share *** (%)	6.10	22.08	21.15	19.04	17.86	1.91
	threat in CA	**1**	**3**	**2**	**2**	**2**	**1**
	threat in CLS	**1**	**2**	**2**	**2**	**2**	**1**
Čierne (2082 ha)	area (ha)*	18.51	238.46	67.98	116.69	53.98	9.19
	share ** (%)	0.89	11.45	3.27	5.60	2.59	0.44
	share *** (%)	14.63	24.04	27.77	23.25	23.54	11.37
	threat in CA	**1**	**3**	**2**	**2**	**2**	**1**
	threat in CLS	**2**	**3**	**3**	**2**	**2**	**1**
Oščadnica (5863 ha)	area (ha)*	21.12	737.96	171.10	275.75	225.17	0.56
	share ** (%)	0.36	12.59	2.92	4.70	3.84	0.01
	share *** (%)	13.64	24.37	22.52	24.58	31.88	1.51
	threat in CA	**1**	**3**	**2**	**2**	**2**	**1**
	threat in CLS	**2**	**3**	**2**	**3**	**3**	**1**
Skalité (3316 ha)	area (ha)*	16.67	360.47	146.91	289.22	188.26	2.62
	share ** (%)	0.50	10.87	4.43	8.72	5.68	0.08
	share *** (%)	17.34	34.24	27.94	31.42	28.68	22.00
	threat in CA	**1**	**3**	**2**	**3**	**2**	**1**
	threat in CLS	**2**	**3**	**3**	**3**	**3**	**2**
Svrčinovec (1574 ha)	area (ha)*	19.11	142.65	59.85	95.49	90.47	4.25
	share ** (%)	1.21	9.06	3.80	6.07	5.75	0.27
	share *** (%)	20.45	27.33	35.50	25.13	28.91	6.99
	threat in CA	**1**	**3**	**2**	**3**	**2**	**1**
	threat in CLS	**2**	**3**	**3**	**3**	**3**	**1**

Notes: * – the total area of landslides in hectares located in the area of the given element of the landscape structure within the cadastral area; ** – the proportion of landslides in % within an element of the landscape structure to the total area of the cadastral area; *** – the proportion of landslides area in % to the total area of the element of the landscape structure within the cadastral area.

The continual reduction in the proportion of actively used areas of PG or arable land results in their overgrowth by natural seeding bushes, and in some areas, there are even continuous forest stands. This increases the significant proportion of this non-forest tree and shrub vegetation in the area among the elements of the current landscape structure. As in the case of PG and TAL, the impact of natural processes is increasing and also the susceptibility of the area to landslides is increasing.

At present, only a small area of arable land has been preserved and is still being used. It is situated in the valley part of the area or on gentle slopes, where the landslides occur only to a lesser extent and the favorable conditions for the formation of landslides develop to a much lesser degree. Therefore, the current threat to arable land is considerably less compared to other elements of the landscape.

When selecting areas for residential development, complicated areas are usually being avoided, so that the buildings and the lives of people are not endangered. Under certain conditions, however, there are signs of slope movement or they occur accidentally under significant environmental influences such as storm rainfalls, prolonged rains causing soil and subsoil supersaturation with water and so on. The development of settlements and the need to build new areas often come across the sites that are potentially susceptible to landslides.

Limits of territorial development

The overall development of the territory must be subject to certain rules, which should be included in the territorial plan. Based on the evaluation of landslides on the individual elements of the landscape structure and in individual cadastral areas, we determined the limits of the development in the monitored area. Spatial planning limits have been divided into two categories according to the sectors that are most influenced by these limits – limiting the development of land intended for residential development or limiting the development of land intended for agricultural and forestry use.

1. The limits of the territorial development designated for residential development are based on the given area conditions. natural conditions and factors causing landsliding. They are based on the following restrictions and warnings:
    - new construction cannot be located in areas where there are active and potential landslides, including the additional construction and renovation of existing buildings;
    - new construction in areas of stabilized landslides is possible only after the elaboration of geological and engineering documentation confirming that the proposed investment will not affect the balance of the territory and will not activate landslides;
    - existing buildings in the areas of active, potential and stabilized landslides can be maintained and used with recommendations based on the elaborated geological and engineering documentation identifying the need to implement safety measures (e.g., dewatering the ground and surface water from a slope by building drainages. relocating soil from an overburdened landslide base or lightening the landslide in the upper part, construction of retaining walls and other geotechnical structures);
    - unprofessional intervention into the disturbed slope can cause the slope movements to be activated (it can be, e.g., when the active parts by the weight of shallow imbedded objects and by building the embankment during landscaping and road construction);
    - poorly stable slopes are damaged by disruption of their passive part during landscaping. by notches for buildings or roads and during construction of utility networks;
    - negative changes in hydrogeological conditions on the slope also contribute to the disruption of slope stability, for example, long-term construction causes acceleration of rock weathering and erosion on slopes.

2. The limited development of an agricultural and forestry area should be based on the following restrictions and warnings:
   - arable and forest land management in areas with active and potential landslides without heavy machinery;
   - the area with active and potential landslides should not be used as arable land, but only as meadows and pastures;
   - utilization of contoured agrotechnics – usually associated with deep plowing and includes all common agrotechnical interventions, which are carried out in the direction of contour lines, or utilization of no-till farming (the most progressive protection measure against landslides);
   - rational crop rotation – crop rotation with protective effect (e.g., use of multiannual fodder crops and grasses with very good protective effect from the beginning of the formation of closed stands up to its removal);
   - planting of special-purpose agricultural and protective greenery, reduction of arable land blocks;
   - stabilization of large grassy areas in territories with active and potential landslides in the form of woody vegetation dominated by scrub vegetation;
   - do not use the clear-felling method in forest management and do not build roads in areas with active and potential landslides;
   - the quality of bedrock has a great influence on the occurrence of landslides, the slopes on the flysch are most endangered (if the upper layers of rocks slide alongside the soil, the risk of landslide increases), climate conditions are also important, for example, short and violent thunderstorms cause shallow landslides, but long-lasting rains of lower intensity cause deeper and more extensive landslides to which the forest is less resistant;
   - the risk of shallow landslides in the forest can be mitigated by the roots of trees that reach deeper into the soil. trapping more firmly and permanently than roots of other plants, the roots of the trees penetrate through the soil layers to the bedrock and thus increase the slip resistance; the ability of the forest to dry the soil is also very important;
   - as a protection against landslides, deep-rooting trees (e.g., pine, larch, oak, valuable deciduous trees) are suitable, although in real conditions (e.g., waterlogged soils), these trees can also create a shallow root system – suitable woody plants for individual habitats should be preferred;
   - shallow rooted trees such as spruce (*Picea abies*) are more susceptible to landslides and it is not advisable to create continuous monocultural stands in landslide prone areas;
   - crop density is also important – gaps in which there are no interweaving roots are more prone to landslides;
   - slopes endangered by landslides should be used as permanent forests and natural forest regeneration should be preferred – natural regeneration in stands with a natural species composition often speeds up the creation of new stands;
   - following the principles of forest protection and restoration with a natural species

composition will mitigate the potential threat to areas that are currently endangered by large-scale logging and tree dying in monocultural spruce stands;

- forest roads that run across steep slopes can cause disruption of the stability; drainage of water off the road towards the hillside is dangerous, it is preferable to use indentation that drains water onto the embankment slope and into the forest;
- do not use heavy machinery when loading slopes, they can disrupt the soil and vibrations can cause the soggy soil to slip;
- landslides can also be caused by the use of explosives in road construction – explosion shocks can detach landslides.

Given the diversity of habitat conditions and land use, it is necessary to define conditions for the protection of land from landslides in individual sites, taking into account all their specific characteristics and conditions.

Conclusion

With the gradual urbanization and the increasing need for higher living comfort, developers are forced to deal with the evaluation of complex engineering-geological conditions in more detail when assessing not only ground, underground, line, but also other types of buildings. Proper placement of buildings with a thorough knowledge of the current state of the geological environment, but especially the anticipation of geological processes in the future, with an emphasis on geobarriers, can be a means of saving high financial costs for possible remediation in the future; and last but not least, it will increase the safety of the population.

Changing land use in the local zoning plan due to changing conditions is a complex process not only for the municipality but also for the landowner. Spatial planning within the municipality must identify areas of natural geological hazards and take into account the safety for its inhabitants and their property. It is necessary for municipalities to undertake appropriate activities in relation to the localization of hazards, in particular, the risks of formation of landslides and the prevention of investments in unsuitable areas.

Compensatory costs associated with land use change and decline in value appear to be negligible compared to the potential losses due to the occurrence of landslides in built-up and inhabited areas.

Acknowledgements

The contribution was prepared within the grant project of the Ministry of Education of the Slovak Republic and the Slovak Academy of Sciences No. 2/0132/18 "Historical and present changes in the landscape diversity and biodiversity caused by natural and anthropogenic factors".

References

Barančoková, M. & Kenderessy P. (2014). Assessment of landslide risk using GIS and statistical methods in Kysuce region. *Ekológia (Bratislava)*, 33(1), 26–35. DOI: 10.2478/eko-2014-0004

Bathrellos, G.D., Kalivas, D.P. & Skilodimou H.D. (2009). GIS-based landslide susceptibility mapping models applied to natural and urban planning in Trikala, Central Greece. *Estud. Geol.*, 65(1), 49–65. DOI: 10.3989/egeol.08642.036.

Bydłosz, J. & Hanus P. (2013). The impact of landslide areas on municipal spatial planning. *Real Estate Management and Valuation*, 21(4), 5–10. DOI: 10.2478/remav-2013-0031

Carrara, A., Giovanni, C. & Frattini P. (2003). Geomorphological and historical data in assessing landslide hazard. *Earth Surface Processes and Landforms*, 28, 1125–1142. DOI: 10.1002/esp.545.

Cascini, L., Bonnard, Ch., Corominas, J., Jibson, R. & Montero-Olarte J. (2005). Landslide hazard and risk zoning for urban planning and development. In O. Hungr, R. Fell, R. Couture & E. Eberhardt (Eds.), *Landslide risk management* (pp. 199–235). London: Taylor & Francis Group.

Guillard, C. & Zêzere J.L. (2012). Landslide susceptibility assessment and validation in the framework of municipal planning in Portugal: the case of Loures municipality *Environ. Manag.*, 50, 721–735. DOI: 10.1007/s00267-012-9921-7.

Lan, H.X., Zhou, C.H., Wang, L.J., Zhang, H.Y. & Li R.H. (2004). Landslide hazard spatial analysis and prediction using GIS in the Xiaojiang watershed, Yunnan, China. *Engineering Geology*, 76, 109–128. DOI: 10.1016/j.enggeo.2004.06.009.

Lekkas, E. (2000). *Natural and technological disasters destructions*. Athens: Access Pre-press.

Malgot, J. & Baliak F. (2002). The influence of human activity on the development of landslides in Slovakia. *Geografický Časopis*, 54(1), 21–38.

Mancini, F., Ceppi, C. & Ritrovato G. (2010). GIS and statistical analysis for landslide susceptibility mapping in the Daunia area, Italy. *Natural Hazards and Earth Systems Science*, 10, 1851–1864. DOI: 10.5194/nhess-10-1851-2010.

Mazúr, E. & Lukniš M. (1986). *Geomorfologické členenie SSR a ČSSR.* Časť SSR. 1:50 000. Bratislava: Slovenská kartografia.

Mihai, B., Savulescu, I., Sandric, I. & Chitu Z. (2014). Integration of landslide susceptibility assessment in urban development: a case study in Predeal town, Romanian Carpathians. *Area*, 46(4), 377–388. DOI: 10.1111/area.12123.

Nowak, A. & Tokarczyk N. (2013). Evaluation of soil resilience to anthropopressure in Łosie village (Lower Beskids Mts) – preliminary results. *Ekológia (Bratislava)*, 32(1), 138–147. DOI: 10.2478/eko-2013-0012.

Řehák, D., Sikorová, K. & Senovský P. (2013). Safety assessment for spatial development. *Ekológia (Bratislava)*, 32(2), 220–241. DOI: 10.2478/eko-2013-0019.

Šimeková, J., Martinčeková, T., Abrahám, P., Gejdoš, T., Grenčíková, A., Grman, D., Hrašna, M., Jadroň, D., Záthurecký, A., Kotrčová, E., Liščák, P., Malgot, J., Masný, M., Mokrá, M., Petro, Ľ., Polaščinová, E., Solčiansky, R., Kopecký, M., Žabková, E., Wanieková, D., Baliak, F., Caudt, Ľ., Rusnák, M. & Sluka V. (2006). *Atlas máp stability svahov SR v mierke 1: 50 000*. Záverečná správa. Bratislava: Geofond, MŽP SR.

Ekológia (Bratislava) Vol. 38, No. 4, p. 318–335, 2019
DOI:10.2478/eko-2019-0024

CHANGES IN SNOWBED VEGETATION IN THE WESTERN CARPATHIANS UNDER CHANGING CLIMATIC CONDITIONS AND LAND USE IN THE LAST DECADES

ANDREJ PALAJ, JOZEF KOLLÁR

Department of Ecology and Environmental Sciences, Faculty of Natural Sciences CPU in Nitra, Trieda A. Hlinku 1, 949 74 Nitra, Slovak Republic; e-mail: andrej.palaj@savba.sk
Institute of Landscape Ecology, Slovak Academy of Sciences, Štefánikova 3, P.O. Box 254, 814 99 Bratislava, Slovak Republic; e-mail: j.kollar@savba.sk

Abstract

Palaj A., Kollár J.: Changes in snowbed vegetation in the Western Carpathians under changing climatic conditions and land use in the last decades. Ekológia (Bratislava), Vol. 38, No. 4, p. 318–335, 2019.

Snowbed vegetation is one of the most sensitive alpine vegetation type to the climate change, because shortened period of snow cover has essential impact on the snowbed environment. We focus on its changes in the Western Tatras, which is a part of the Western Carpathians (Slovakia). The assessment of changes in snowbed vegetation is based on the method of pair comparison. In 2016–2018, we resampled 21 historical phytocoenological relevés of *Festucion picturatae* and *Salicion herbaceae* alliances from 1974 and 1976. Historical data include 45 species, while recent data include 50 species. We observed a decrease in the frequency of species characteristic for snowbeds and, on the other hand, an increase in that for strong competitors, especially grasses and small shrubs from adjacent habitats. According to Ellenberg's ecological indices, there is some increase in temperature and decrease in light ecological factors in snowbed habitats. In *S. herbaceae* data, a statistically significant increase in the average species number was observed with new species that penetrated from the adjacent habitats. Changes in species composition between historical and recent data are confirmed by Non-metric multidimensional scaling (nMDS) ordination diagram. Linear mixed-effect models showed big variability in factors that have impact on phytodiversity; nevertheless, temperature is the most significant factor.

Key words: snowbed vegetation, changes, climate, land-use, Western Carpathians.

Introduction

Alpine vegetation is adapted to the specific natural conditions such as diversified relief and bedrock, resulting in varied soil properties, dynamic meso- and microclimate, and so on. Balance between alpine vegetation and its environment is fragile (Kliment, Valachovič, 2007), and change in environment factors leads to the changes in composition and structure of vegetation species.

Changes in temperature are crucial for alpine vegetation. It is accompanied by migration of vertical plant species (Klanderud, Birks, 2003; Pauli et al., 2007; Wilson, Nilsson, 2009; Grytnes et al., 2014; Kyyak et al., 2014) hand in hand with the increase in plant species diversity (Holzinger et al., 2008; Odland et al., 2010; Erschbamer et al., 2011; Kanka et al., 2011; Stöckli et al., 2011; Pauli et al., 2012; Matteodo et al., 2016).

In the European mountains, climate change includes an increase in average annual temperature and a decrease in the precipitation in the form of snow, which result in prolongation of vegetation period (Dye, 2002; Matteodo et al., 2016; Czortek et al., 2018). It promotes migration of eurytopic species from the lower altitudes (Vittoz et al., 2008; Britton et al., 2009; Engler et al., 2011; Palaj, Kollár, 2018). This leads to vegetation homogenization, because expansion of new species upward is accompanied by disappearing of stenotopic species (Jurasinski, Kreyling, 2007; Britton et al., 2009; Erschbamer et al., 2011; Ross et al., 2012; Carbognani et al., 2014; Kobiv, 2018).

Land use change, such as grazing cessation, is another essential driver of plant redistribution. Grazing cessation and climate warming have similar impact on alpine flora, and they operate in synergy (Kobiv, 2017). Grazing abandonment triggers successional changes in few years (Hulme et al., 1999; Velev, Apostolova, 2008), with a decrease in light- and cold-demanding species (Klein et al., 2004; Wesełowska, 2009; Vassilev et al., 2011; Matteodo et al., 2013), which are replaced by strong competitors such as grasses and small shrubs (Baur et al., 2007; Kucharzyk, Augustyn, 2010; Vassilev et al., 2011; Kobiv, 2014; Kyyak et al., 2014). Contrary to the climate change, land-use change is often accompanied by diversity decrease (Fabiszewski, Wojtuń, 2001; Baur et al., 2007; Korzeniak, 2016).

Snowbed vegetation is one of the plant communities most sensitive to changes because it includes many ecological specialists (Kliment, Valachovič, 2007). Shortened period of snow cover is followed here by disappearance of small chinophilous and cold-demanding species, which are replaced by strong competitors such as grasses and small shrubs from adjacent communities and lower altitudes (Kobiv, Nesteruk, 2001; Virtanen et al., 2003; Britton et al., 2009; Micu, 2009; Daniëls et al., 2011; Kudo et al., 2011; Kapfer et al., 2012; Elumeeva et al., 2013; Carbognani et al., 2014; Sandvik, Odland, 2014; Kobiv, 2016, 2017; Czortek et al., 2018). As a result, contemporary snowbed communities resemble alpine grassland much more than 50 years ago (Matteodo et al., 2016) and are among the most vulnerable alpine vegetation types.

Therefore, this vegetation type offers a good opportunity to study changes taking place in alpine habitats driven by climate change and grazing cessation. We focused on this topic in the alpine environment in the part of Western Tatras. In 2016–2018, we resampled 21 historical relevés of *Festucion picturatae* and *Salicion herbaceae* alliances sampled in 1974–1976. In this article, we examined both qualitative and quantitative changes in species composition, changes in environment indicated by Ellenberg's ecological indices (EIV) (Ellenberg et al., 1992), and impact of those environment changes on the diversity.

Material and methods

Study area

The study covers an area of 2,345 ha in the Jamnícka and Račkova valley parts of the Western Tatras Mountains (Fig. 1). This area is formed from granite, migmatite, metamorphic rocks and has numerous mylonite zones in the summit areas (Nemčok, 1994). The highest point is Bystrá peak at 2,248 m a.s.l., which is also the highest peak in the Western Tatras. The area lies in the cold climate region (Plesník, 1974), and we use climate data from 1966 to 2018

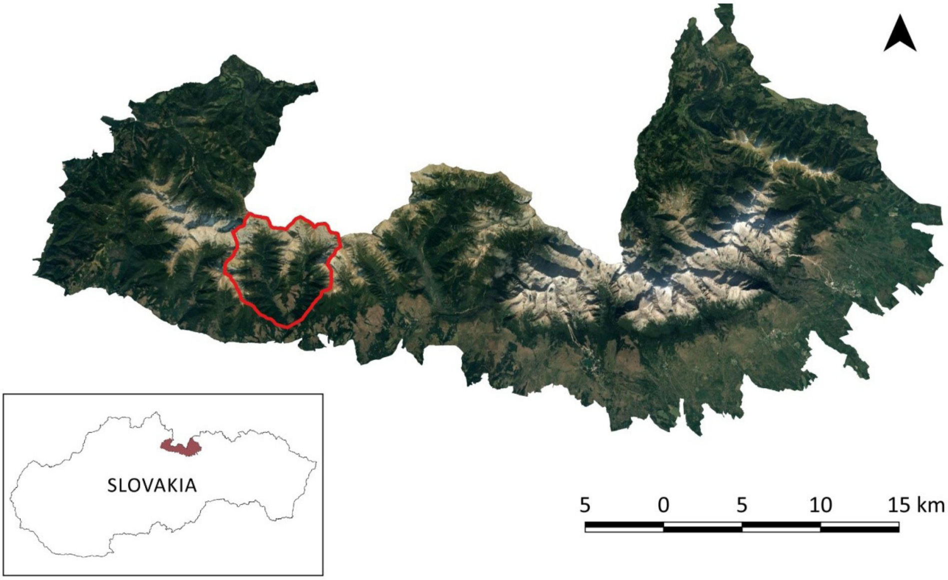


Fig. 1. Location of the study area in the Tatra National Park and in Slovakia.

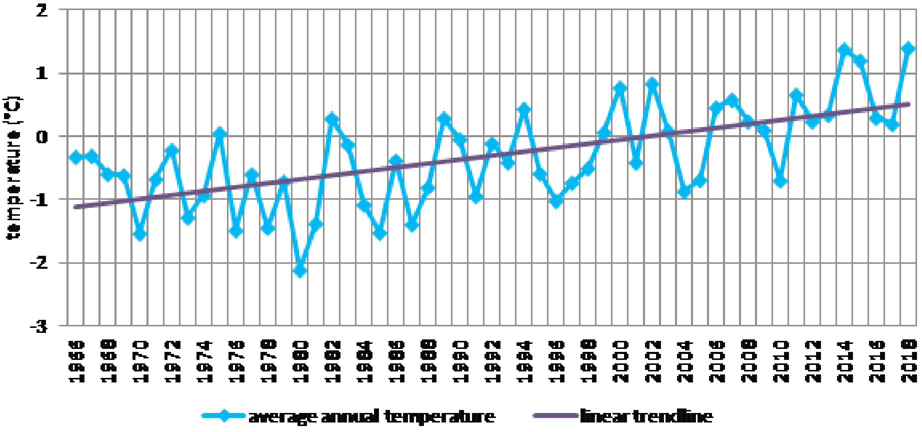


Fig. 2. Average annual temperature in the Kasprovy Wierch meteorological station over 53 years (Kasper peak meteorological station).

from Kasper peak meteorological station (Fig. 2), located 11 km east from Bystrá peak at an altitude of 1,987 m, to specify the development of the mean annual temperature in the study area. Changes in the values of meteorological elements from this station was analyzed by Czortek et al. (2018), who performed linear regression models using ANOVA to find whether there is a relationship between the year of observation and chosen climate variables (Table 1). According to their results, the mean annual temperature on Kasper peak increased significantly by 1.30 °C during the monitoring period. This increase in temperature was driven by an increase in mean minimum rather than mean maximum annual temperature. The number of days with rain has increased significantly by about 43 days here, and the growing season has extended significantly by about 20 days (Czortek et al., 2018). Study area was intensively grazed by cattle and sheep since the 13th century, which is associated with the period of Wallachian colonization. The damping of pastoral activities in the early 1980s was connected with adding a section of the Western Tatras to the Tatra National Park, while grazing was definitely excluded from the study area in 1986.

Communities of alliance *S. herbaceae* are the first to occupy snow soils. They prefer flat or slightly inclined habitats, which are covered by snow for about 8–10 months during year. Habitats of *Festucion picturatae* communities are situated on stabilized scree slopes with an inclination of 20°–50°, not with long-term snow cover (6.5–7.5 months) (Dúbravcová, 2007). These two floristically poor snowbed communities are most threatened by longer annual snow-free period, resulting from an increase in the mean annual temperature and a decrease in the number of days with snowfall (Carbognani et al., 2014; Pickering et al., 2014; Sandvik, Odland, 2014; Matteodo et al., 2016).

Table 1. Climate changes in the Kasprowy Wierch metorological station (significance level: 0.05*) (Czortek et al., 2018).

Variables	**Change**	**Time period**
Mean annual temperature	+1.30 °C*	1951–2015
Max. mean annual temperature	+1.93 °C*	1951–2015
Min. mean annual temperature	+2.04 °C*	1951–2015
Number of days with rain	+43*	1959–2015
Number of days with snow	-13	1959–2015
Growing season length	+20*	1951–2015

Data collection

The assessment of changes in plant communities was based on a pair comparison of historical and current relevés. In 2016–2018, we resampled historical relevés from 1973 to 1976 (Turečková, 1974; Dúbravcová, 1976), stored in the Central database of phytocoenological relevés (CDF) (Hegedüšová, 2007; Šibík, 2012). Localization of the study sites was based on the header data of historical relevés. If the site conditions did not comply with the localization notes in the headers, we chose the study area at a maximum distance of 50 m from the probable center of the historical plot. In the case of longer distance, relevés were not resampled.

The methods of Zürich-Montpellier School of Phytosociology (Braun-Blanquet, 1964) were used to study the plant communities. The 7-degree Braun-Blanquet scale was used to estimate the species coverage. Vascular plant nomenclature was modified according to Marhold and Hindák (1998); cryptograms are not included. A major part of the phytocoenological relevés was published in our previous study (Palaj, Kollár, 2017). Our syntaxonomical classification of the studied vegetation follows Jarolímek et al. (2008). For the storage of vegetation data, we used Turboveg (Hennekens, Schaminée, 2001), and for its editing and post-processing, we used the Juice program (Tichý, 2002).

Data analyses

We use relative frequency change to assess the changes in species composition of snowbed communities. Special emphasis is placed on the frequency and abundance changes of species indicating the grazing to evaluate the effect of cessation of pasture in the 1980s. These indicators are defined by the statistical significance of the relationship between species occurrence/abundance and grazing presence of sites. For determining the level of influence, we use 125 relevés sampled by Horák (1970) who specified the absence/presence of grazing. These data are analyzed using the Indicspecies R package (De Cáceres, Legendre, 2009), and changes in species abundance are tested with t-test.

Shannon–Wiener (SW) (Hill, 1973) and evenness indices (Pielou, 1966) are calculated for all historical and current relevés, and the values are depicted in the box plots using R package reshape2 (Wickham, 2007). Historical and current site conditions are estimated by bioindication using EIV with the Juice software (Tichý, 2002). We focus on the relationship between species and ecological factors, such as light, temperature, continentality, moisture, soil reaction, and nutrients with species abundances included. The box plots (Wickham, 2007) show the results, and the paired t-test was performed using RStudio software (R Core Team, 2017).

Relevé (dis)similarity is estimated by nMDS indirect ordination methods with the function metaMDS of RStudio Vegan package (Oksanen et al., 2017). Analysis is performed on log-transformed data using Bray–Curtis dissimilarity matrix. We allow the algorithm to ordinate in three dimensions (k=3) to reduce the stress value equal to or below 0.1, which indicate good results (Hebák et al., 2007). Finally, we use envfit function (Oksanen et al., 2017) to fit environmental factors onto an ordination. The projections of points onto vectors have maximum correlation with corresponding environmental variables, and the factors show the averages of factor levels.

To assess the impact of environmental factors on diversity of communities, we use linear mixed-effect models (Gellman, Hill, 2007; Winter, 2013). Diversity is predicted by fixed effect factors—grazing status of study sites and EIV. As a random effect, we use relevé affiliation to the study sites. The variable "grazing status" includes only the absence and presence of grazing and is determined by the availability of the study site for the livestock. According to Cook (1966), Wilcox and Wood (1988), Crofts and Jefferson (1999), Coldea (2003), Ditsch et al. (2006), Baur et al. (2007), and Catorci et al. (2012), livestock is able to graze only areas with inclination not exceeding 35°. To identify such areas, we created a vector layer from the digital elevation model (10 m) using QGIS module Terrain Analysis (QGIS Development Team, 2019) and overlapped it with the study site coordinates.

Results

Changes in phytodiversity

As a first step, we identified species that we consider to indicate grazing to separate changes triggered by climate change and land use (grazing cessation). These are as follow: *Anthoxanthum alpinum*, *Deschampsia cespitosa*, *Nardus stricta*, *Omalotheca supina*, and *Phleum rhaeticum.* Except for *Nardus stricta*, all of them show decrease in their frequency. In addition, *Anthoxanthum alpinum*, *Deschampsia cespitosa*, and *Phleum rhaeticum* decreased even in their abundance. However, paired t-test at a 0.05 level of significance did not show any statistically significant differences (Table 2).

Table 2. Changes in abundance and frequency of indicator species and results of the paired t-test.

Indicator species	**F_h**	**F_c**	**A_h**	**A_c**	**t-value**	**p-value**
Antoxanthum alpinum	61.90	42.86	6.08	2.22	1.644	0.116
Deschampsia cespitosa	19.05	9.52	2.75	2.00	1.099	0.284
Nardus stricta	23.81	38.10	2.20	3.38	-1.099	0.285
Omalotheca supina	23.81	9.52	14.40	20.50	0.412	0.685
Phleum rhaeticum	23.81	19.05	2.40	2.25	0.413	0.684

Notes: F_h – frequency in group of historical relevés (%); F_c – frequency in group of current relevés (%); A_h – average non-zero abundance in group of historical relevés (%); A_c – average non-zero abundance in group of current relevés (%).

In summary, we recorded 50 species of vascular plants, while both Turečková (1974) and Dúbravcová (1976) recorded only 45 species. Also phytodiversity parameters such as SW index and equitability show some differences between historical and actual data set (Fig. 3). However, these changes are not statistically significant, except for the average number of species per relevé in *Salicion herbaceae* dataset with its increase from 9.14 to 11.29 (Table 3).

Relevés of *Festucion picturatae* alliance show distinct decrease in the frequency of the most of alliance diagnostic species, such as *Doronicum stiriacum* (-80%), *Soldanella carpatica* (–80%),

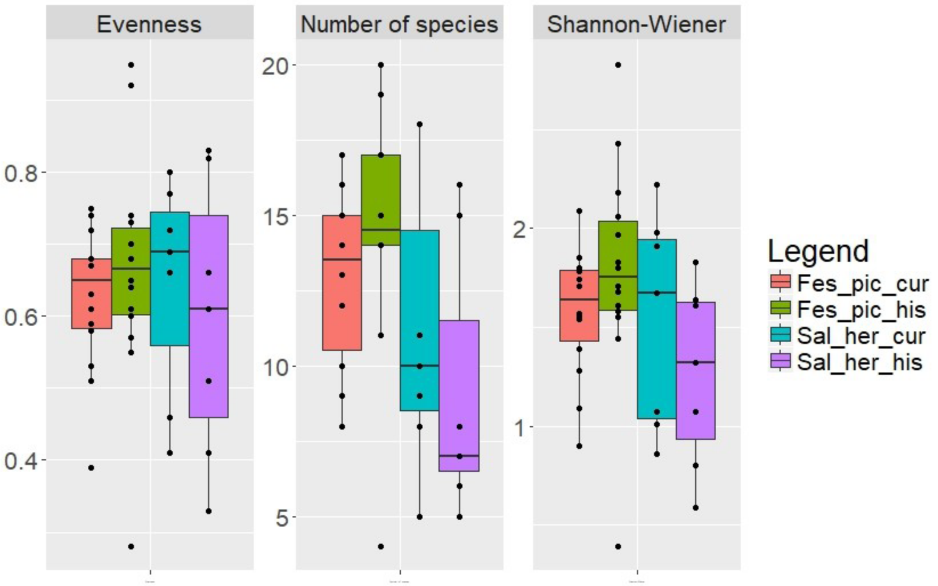


Fig. 3. Comparison of historical and contemporary diversity parameters.
Notes: Fes_pic_cur – current relevés of *Festucion picturatae*; Fes_pic_his – historical relevés of *Festucion picturatae*; Sal_her_cur – current relevés of *Salicion herbaceae*; Sal_her_his – historical relevés of *Salicion herbaceae*.

Table 3. Comparison of average values for historical and contemporary site conditions estimated by Ellenberg's eco-indices and results of the paired t-test (significance level: 0.05*).

Atribute	*Festucion picturatae*				*Salicion herbaceae*			
	Mean h	**Mean c**	**t-value**	**p-value**	**Mean h**	**Mean c**	**t-value**	**p-value**
Shannon-Wiener	1.790	1.577	1.499	0.158	1.266	1.531	-1.009	0.352
Evenness	0.666	0.625	0.910	0.380	0.596	0.644	-0.461	0.661
Number of species	14.643	12.786	1.738	0.106	9.143	11.286	-2.680*	0.037
Light	7.167	6.914	2.155*	0.050	7.316	7.243	0.538	0.610
Temperature	2.282	2.539	-1.871	0.084	1.827	2.130	-1.649	0.150
Continentality	2.919	3.021	-0.912	0.379	2.941	3.133	-0.802	0.453
Moisture	5.667	5.544	0.759	0.462	6.693	6.007	1.982	0.095
Soil reaction	3.224	3.336	-0.488	0.634	3.444	3.513	-0.338	0.747
Nutrients	2.816	2.832	-0.086	0.933	4.217	3.701	1.557	0.171

Ranunculus pseudomontanus (−55.6%), and *Potentilla aurea* (−45.5%) (Fig. 4). These species are replaced by species of adjacent communities, especially by those typical for *Caricetea curvulae*, *Loiseleurio-Vaccinieta*, and *Nardetea strictae* classes (*N. stricta* [+66.7%], *Vaccinium myrtillus* [+28.6%], *Agrostis pyrenaica* [+20%], and *Hieracium alpinum* [+11.1%]). Of species indicating grazing, we observed a decrease in the frequency of *Omalotheca supina* (−100%), *Phleum rhaeticum* (−75%), *Deschampsia cespitosa* (−66.7%), and *Anthoxanthum alpinum* (−30%); only

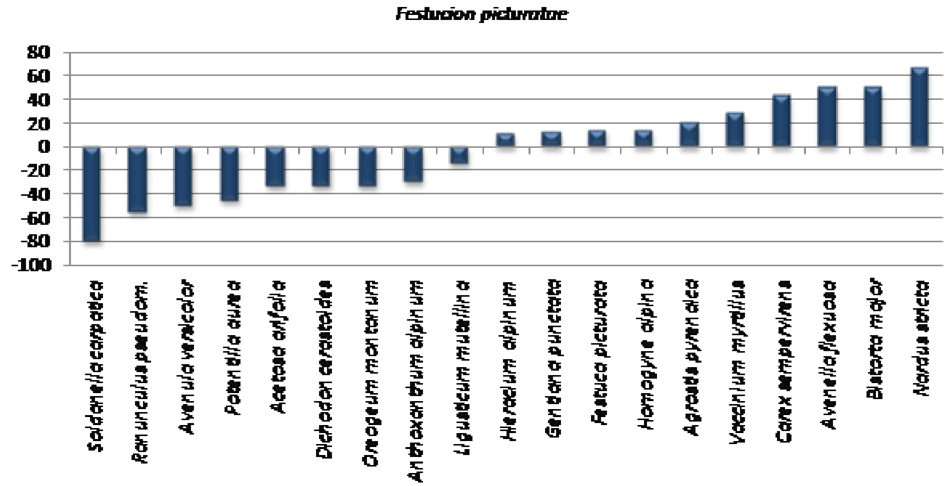


Fig. 4. Relative change of species frequencies of *Festucion pictuaratae* alliance.

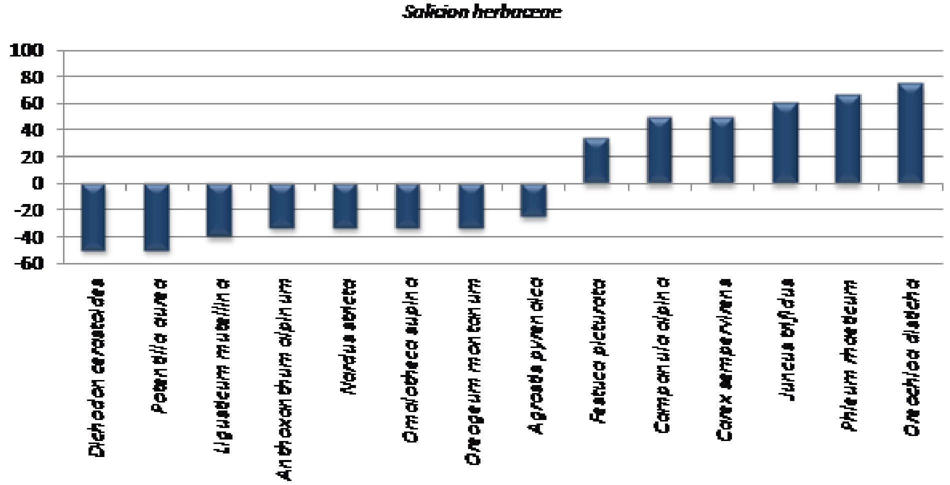


Fig. 5. Relative change of species frequencies of *Salicion herbaceae* alliance.

Nardus stricta (+66.7%) showed an increase. Compared to the historical data, we recorded the occurrence of some additional species. These includes some Carpathian endemic species such as *Campanula serrata* and *Cardaminopsis neglecta*; endangered species such as *Primula minima* (EN), *Juncus filiformis* (LR), *Silene acaulis* (LR), and *Viola lutea* ssp. *sudetica* (LR); and some others (*Achillea millefolium*, *Bartsia alpina*, *Carex nigra*, *Poa alpina*, *Pulsatilla scherfelii* and *Vaccinium gaultherioides*). On the contrary, our research did not confirm some endangered species such as *Leucanthemopsis alpina* (VU) and *Rhodiola rosea* (VUr), Tatra Mts. endemic *Poa granitica*, Carpathian subendemic *Campanula tatrae* and some others, such as *Adenostyles alliariae*, *Calamagrostis villosa*, *Cerastium fontanum*, *Omalotheca supina* and *Sedum alpestre*.

Salicion herbaceae alliance shows significant decrease in the frequency of its dominant and characteristic species (Fig. 5), especially *Leucanthemopsis alpina* (−100%), *Sedum alpestre* (−100%), *Dichodon cerastoides* (−50%), *Ligusticum mutellina* (−40%), *Omalotheca supina* (−33.3%), and *Oreogeum montanum* (−33.3%). On the other hand, we observed an increase in some species characteristic for *Juncion trifidi* alliance (*Avenella flexuosa* [+100%], *Festuca supina* [+100%], *Oreochloa disticha* [+75%], *Juncus trifidus* [+60%], *Campanula alpina* [+50%]) or *Luzuletum obscurae* association (*Homogyne alpina* [+100%] and *Festuca picturata* [+33.3%]). Of the grazing indicators, *Anthoxanthum alpinum* (−33.3%), *Nardus stricta* (−33.3%), and *Omalotheca supina* (−33.3%) decreased in frequency, contrary to *Deschampsia cespitosa* (+0%) with no change and *Phleum rhaeticum* (+66.7%) with an increase. Our results revealed an increase in the occurrence of some reptile small shrubs, especially those of *Vaccinium* genus with new occurrence in some plots. Contrary to the historical data, we did not record Tatra Mts. endemic *Poa granitica* and endangered *Leucanthemopsis alpina* (VU) and found some new species, including *Gentiana punctata* (LR) and *Primula minima* (EN).

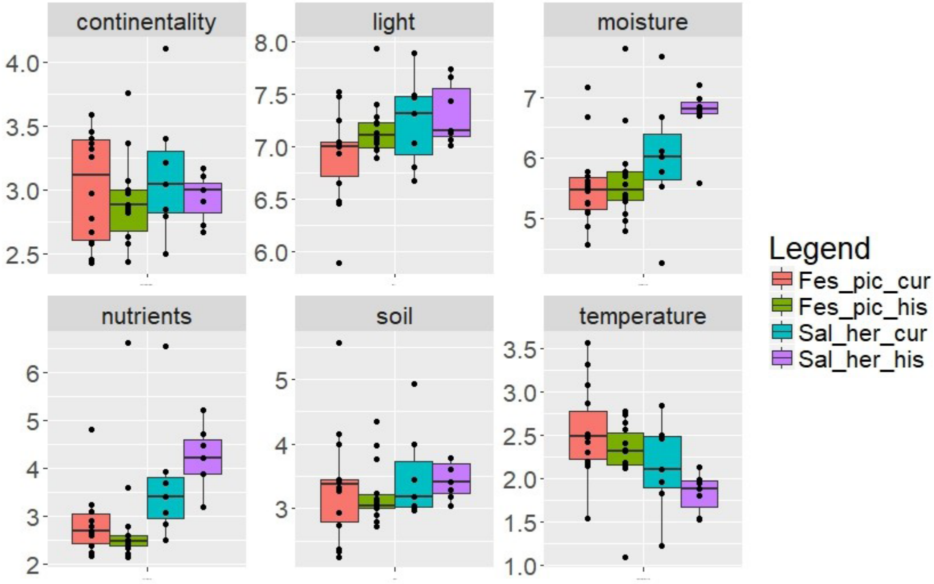


Fig. 6. Comparison of historical and contemporary site conditions estimated by Ellenberg‘s eco-indices.
Notes: Fes_pic_cur – current relevés of *Festucion picturatae*; Fes_pic_his – historical relevés of *Festucion picturatae*; Sal_her_cur - current relevés of *Salicion herbaceae*; Sal_her_his – historical relevés of *Salicion herbaceae*.

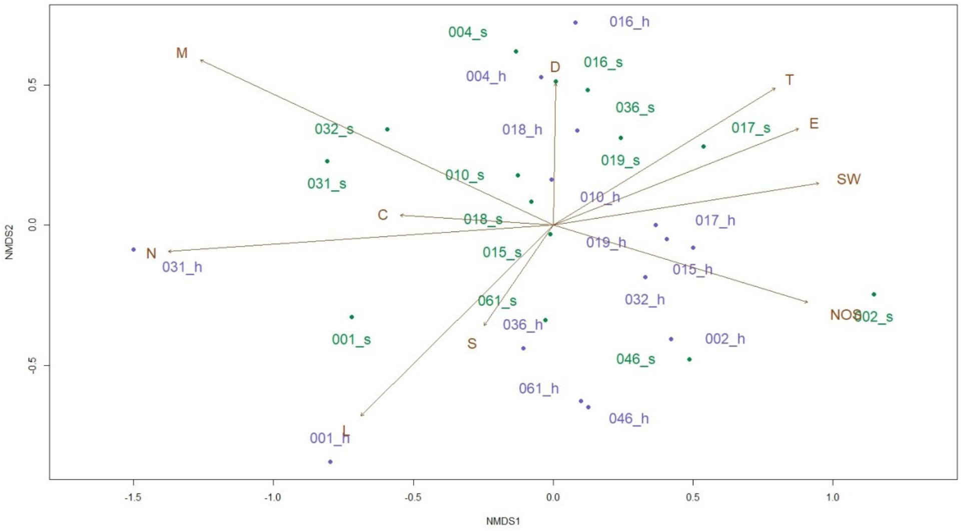


Fig. 7. nMDS ordination diagram (k=3) showing the distribution of current and historical relevés of *Festucion picturatae* alliance with fitted environmental factors. Blue colour indicates historical relevés, green indicates current relevés. Notes: sufix _h – historical relevés; sufix _s – current relevés; C – continentality; D – date; L – light; M – moisture; N – nutrients; S – soil reaction; T – temperature; E – evenness index; NOS – number of species; SW – Shannon-Wiener index.

Phytoindicated changes in environment conditions

Phytoindicated changes in the environment of *Festucion picturatae* alliance (Fig. 6) include statistically significant increase in the temperature and decrease in light. The average values for light decreased from 7.17 to 6.91 (Table 3). Such shift is caused by the decrease in some light-demanding species (*Avenula versicolor*, *Dichodon cerastoides*, *Oreochloa disticha*, *Phleum rhaeticum*, *Potentilla aurea*, *Sedum alpestre*) and the increase in abundances of some relatively shadow-tolerant species (*Avenella flexuosa*, *Homogyne alpina*, *Solidago virgaurea* subsp. *minuta*, and *Vaccinium myrtillus*). Data on *Salicion herbaceae* alliance revealed some increase in temperature, soil moisture, and soil reaction and decrease in light; however, these changes were not statistically significant by paired t-test.

Ordination plot of historical and recent relevés

NMDS plot (stress 0.10) shows clustering of two groups of *Festucion picturatae* relevés, especially along the second axis (Fig. 7). Most of the historical data are on the right side of the biplot, while the current ones are located mostly on the left side. However, except for sites 2, 31, 32, and 36, relevés of the other sites are mixed on the plot, which suggests small difference in species composition between the two sampling times. On the study site 2, we recorded a

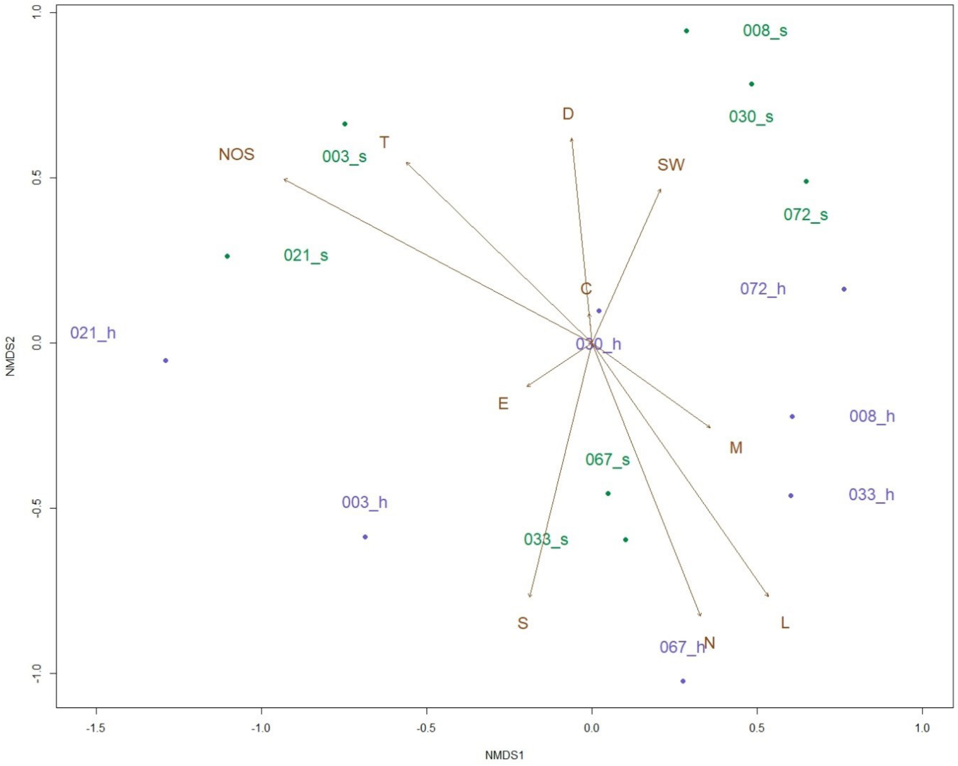


Fig. 8. nMDS ordination diagram (k=3) showing the distribution of current and historical relevés of *Salicion herbaceae* alliance with fitted environmental factors. Blue colour indicates historical relevés, green indicates current relevés.
Notes: sufix _h – historical relevés; sufix _s – current relevés; C – continentality; D – date; L – light; M – moisture; N – nutrients; S – soil reaction; T – temperature; E – evenness index; NOS – number of species; SW – Shannon-Wiener index.

relatively high increase in soil nutrient factor (from 2.38 to 2.65); site 31 shows a decrease in light (from 7.94 to 7.52). The differences observed in the species composition of historical and current relevés in sites 36 and 32 are caused by the decrease in the light variable and the decrease in the number of species.

Figure 8 illustrates that nMDS ordination distinctly separated the historical and current relevés of *Salicion herbaceae* alliance. At the top of the biplot (stress 0.09), current relevés are predominant, while historical ones are mainly situated at the bottom. Almost all relevés are located on the periphery of the ordination diagram, only a historical relevé from the site 30 is situated in the center of the configuration. Sites 3, 8, and 30 show the biggest differences in species composition. Long distances between historical and current relevés in sites 3 and

Table 4. Results of linear mixed-effect models. SW index is used as a dependent variable. Diversity is predicted by fixed effect factors - grazing status of study sites and EIV. As a random effect is used relevé affiliation to the study site (significance level: 0.05*).

Alliance	Factors	Estimate	Standard Error	t-value	p-value
Festucion picturatae	Light	-0.545	0.320	-1.702	0.105
	Temperature	-0.833	0.265	-3.140*	0.006
	Moisture	0.128	0.182	0.702	0.491
	Soil reaction	0.149	0.119	1.253	0.229
	Continentality	0.462	0.196	2.356*	0.031
	Nutrients	-0.637	0.176	-3.624*	0.002
	Pasture	0.182	0.180	1.011	0.326
Salicion herbaceae	Light	0.195	2.055	0.095	0.928
	Temperature	-0.853	1.312	-0.651	0.542
	Moisture	0.060	1.569	0.038	0.971
	Soil reaction	0.597	1.647	0.362	0.730
	Continentality	-0.719	0.660	-1.088	0.318
	Nutrients	-0.787	1.336	-0.589	0.577
	Pasture	-0.270	0.273	-0.990	0.394

8 follow soil nutrient gradient. We recorded an expansion of nutrient-poor soil species (*Juncus trifidus*, *Oreochloa disticha*, and *Vaccinium gaultherioides*) and a decline in nutrient-rich soil species (*Dichodon cerastoides*, *Ligusticum mutellina*, *Phleum rhaeticum*, *Poa alpina*, *P. granitica*, and *Veronica alpina*) here. Changes in species composition indicate decrease in light and increase in temperature. Distance between historical and current relevé in site 30 is due to the increase in SW index (from 1.57 to 2.20) and evenness index (from 0.63 to 0.78).

Impact of phytoindicated environment conditions and grazing on the phytodiversity

According to the results of mixed effect models (Table 4), SW of *Festucion picturatae* alliance is significantly affected by 3 factors: temperature, continentality, and nutrients. In addition to these, light factor also contributes to the changes in values of SW index. Each increase in the value of light, temperature, and nutrients by one unit decreases the value of SW index. Light affected SW (p=0.105), lowering it by about 0.545 (±0.320 standard errors); temperature (p=0.006) decreases it by about 0.833 (±0.265); and nutrients affected SW (p=0.002), lowering it by about 0.637 (±0.176). This effect is most visible on study site 32, where we observed an increase in the values of all three variables with a simultaneous decrease in the values of SW index. Probably, the biggest impact from all factors has temperature, whose change in values has the opposite character (increase/decrease) as a change in values of SW index on 11 of 14 sites. The change in temperature is mainly represented by its increase, which is the result of the expansion of thermophilous species on study sites, for example, *Achillea millefolium*, *Bistorta major*, *Juncus filiformis*, and *Viola lutea* ssp. *sudetica*.

In the case of the *Salicion herbaceae* alliance, none of the factors has a statistically significant effect on the SW diversity index. However, the impact of the temperature can be considered quite marked. Each increase in the values of this factor by one unit results in a decrease in the SW index values by about 0.85 (±1.31). Despite the relatively high standard error, this phenomenon is visible on study sites 3, 8, 21, 30, and 67. Here we observed an expansion of thermophilous species (*Bistorta major*, *Homogyne alpina*, *Huperzia selago*, *Phleum rhaeticum*, *Senecio subalpinus*) and a decline in cold-demanding species (*Aagrostis pyrenaica*, *Dichodon cerastoides*, *Ligusticum mutellina*, *Omalotheca supina*, *Oreogeum montanum*, and *Sedum alpestre*), with a simultaneous decrease in SW index values.

Discussion

Festucion picturatae communities are considered to be permanent under stable ecological conditions. On the contrary, this vegetation become sensitive with changes, especially those linked to the climate (Carbognani et al., 2014). It complies with our research, which revealed penetration of species from adjacent habitats as did Carbognani et al. (2014) in the Rhaetian Alps. Of these, mainly species of *Calamagrostion villosae* alliance occur as Dúbravcová (2007) expected for the Tatra Mts. as well as *Nardus stricta*.

Its obvious increase in the frequency does not comply with natural evolution accompanied by its decrease in the area with grazing reduction or cessation (Austrheim et al., 2007; Speed et al., 2014; Korzeniak, 2016). Owing to the significant decrease in other species indicating grazing (*Anthoxanthum alpinum*, *Deschampsia cespitosa*, *Omalotheca supina*, and *Phleum rhaeticum*) along with an increase in diagnostic species of *Nardion strictae* alliance (*Carex sempervirens*, *Agrostis rupestris*, *Gentiana punctata*, and *Hieracium alpinum*), it can be assumed that the increase in *Nardus stricta* is a result of climate change.

Good adaptability of *N. stricta* to the increased temperature is also evidenced by the simulation of climate change in Harz Mts. (Bruelhide, 2003). Increased frequency of *N. stricta* was also reported by Kanka et al. (2011), Kollár et al. (2016), and Matteodo et al. (2016), who as well as Kudernatsch et al. (2005), Pauli et al. (2007), Holzinger et al. (2008), Britton et al. (2009), Vittoz et al. (2009), Odland et al. (2010), Erschbamer et al. (2011), Stöckli et al. (2011), Pickering et al. (2014), and Sandvik, Odland (2014) confirmed the increase in species of alpine communities with the increase in temperature. This is, in compliance with Norton et al. (2009), in conflict with our findings.

Despite these changes, both historical and current physiognomy is determined by same dominant species, such as *Festuca picturata* and *Luzula alpinopilosa* ssp. *obscura* with no obvious changes in the frequency and abundance. This is also confirmed by nMDS ordination, which showed similarity in relevés from both periods. Besides the above-mentioned reasons, differences were observed for less-abundant species such as decrease in light-demanding species (*Avenula versicolor*, *Dichodon cerastoides*, *Potentilla aurea*, and *Soldanella carpatica*) and increase in relatively thermophilous species (*Bistorta major*, *Homogyne alpina*, and *Juncus filiformis*), and reptile small shrubs (*Vaccinium gaultherioides* and *V. myrtillus*). Generally, increase in small shrub in snowbeds is assigned to shortened period with snow cover and prolongation of vegetation period (Klanderud, Birks 2003; Cannone et al., 2007; Elmendorf et al., 2012; Grytes et al., 2014; Vanneste et al., 2017).

Communities of *Salicion herbaceae* alliance are, as those of *Festucion picturatae*, also sensitive to climate conditions (Carbognani et al., 2014). They are made of chinophilous species with optima in sites covered by snow for about 8–10 months (Krajina, 1933). Gradual truncation of snow cover thickness and lasting in the European alpine landscape, including Tatra Mts. (Czortek et al., 2018), has a crucial impact on the disappearance of snowbeds and their species (Pauli et al., 2007; Kobiv, 2017, 2018). Decline or even loss of such species, for example, *Dichodon cerastoides* and *Veronica alpina*, was also reported in the Romanian Carpathians (Micu, 2009).

In addition, also relatively hygrophilous species follow this trend, because precipitation in the summer decreased in the past decade, despite its annual amount increased (Čepčeková, 2013). This phenomenon was also observed in the Carpathians in Ukraine (Balabukh, Lukianets, 2015; Kobiv, 2018) and, along with temperature increase and associated higher evaporation, it causes decrease in some relatively hygrophilous species (*Juncus filiformis*, *Omalotheca supina*, *Poa granitica*, and *Sedum alpestre*). Most of the study sites are dominated by *Salix herbacea.* We observed significant decrease in its abundance. This reptile small shrub is adapted to the short vegetation period and drops leaves in the break of August and September.

However, because of the climate change, temperature stays relatively high in this period. This prolonged vegetation period promotes some strong competitors (Kobiv, 2018). Of these, we observed increase in frequency of *Festuca picturata* and abundance of *Luzula alpinopilosa*. Partly, it confirms the assumption that communities dominated by *S. herbacea* are changing into the *Festuca picturatae* communities (Dúbravcová, 2007). In addition, we recorded significant increase in *Vaccinium* species, also assigned to the truncation of period with snow cover and prolongation of vegetation period (Klanderud, Birks, 2003; Cannone et al., 2007; Elmendorf et al., 2012; Grytes et al., 2014; Vanneste et al., 2017).

The most distinct difference between changes in the observed alliances is in the change in some diversity parameters. An increase in number of species recorded in *Salix herbaceae* is assigned to the penetration of eurytopic species and the decrease in ecological specialists (Jurasinski, Kreyling, 2007; Britton et al., 2009; Ross et al., 2012; Carbognani et al., 2014). Thus, this vegetation resembles alpine grasslands much more than 50 years ago (Matteodo et al., 2016). On the other hand, *Festuca picturatae* alliance shows a decrease in the average number of species per relevé and SW index as well as increase in temperature according to the EIV.

As suggested by Matteodo et al. (2016), these changes may be a result of (1) real loss of species on the sites in time; (2) more precise historical sampling than that of recent; and (3) inaccurate localization of historical sites. The last reason can be omitted, as did Matteodo et al. (2016), because same trend was observed in almost all the study sites. The second one is associated with rare species, not for those we found to significantly decrease in the numerous sites (*Anthoxanthum alpinum*, *Avenula versicolor*, *Campanula alpina*, *Cerastium fontanum*, *Phleum rhaeticum*, *Ranunculus pseudomontanus*, *Soldanella carpatica*, and *Sedum alpestre*).

Therefore, the decrease in diversity parameters on the *Festucion picturatae* sites is probably caused by the denser coverage of plants compared to *Salicion herbaceae* communities. Windmaißer and Reisch (2013) reported that such vegetation structure leads to the competition of light and soil resources; thus, the penetration of new species is restrained. Also, according to Olsen and Kladerud (2014), communities poor in species (*S. herbaceae*) are more predisposed to penetration of new species than the species-rich ones (*Festucion picturatae*). The decrease in the diversity of *F. picturatae* may be affected also by grazing cessation (Baur et al., 2007; Kucharzyk, Augustyn, 2010; Vassilev et al., 2011) despite its impact on SW index was not found to be statistically significant.

Vegetation of both alliances has some trends in common. We observed a decrease in small species, which were replaced by tall species resulting in lower light income as did Tilman (1988) and Sammul et al. (2000). This change in vegetation vertical structure is linked to the expansion of more thermophilous species reported from several European mountains (Klanderud, Birks, 2003; Pauli et al., 2007; Erschbamer et al., 2011; Kyyak et al., 2014). It is assigned to climate warming having impact especially on small cryophilous and chinophilous species (Pauli et al., 2007; Micu, 2009; Kobiv 2017).

Their decline is caused not only by the increase in temperature leading to overheating of plant tissues (Dahl, 1998; Kobiv, 2018) but also by strong competitors such as grasses and small reptile shrubs from lower altitudes or adjacent habitats (Abeli et al., 2012). Common phenomenon for both snowbed vegetation types is also the increase in relatively xerophilous species, which results from shorter period with snow cover. It is in compliance with the findings from Daniëls et al. (2011), Carbognani et al. (2014), Balabukh, Lukianets (2015), Czortek et al. (2018), and Kobiv (2018).

Changes in alpine vegetation are induced also by grazing cessation. Similar to climate warming, it promotes the growth of small reptile shrubs and competitive grasses, which results in denser vegetation coverage and suppressing of small species (Pakeman, 2004; Kobiv, 2014; Kyyak et al., 2014). Expansion of *Avenella flexuosa* results from the synergistic effect of both grazing reduction/cessation (Pakeman, 2004) and temperature increase (Press et al., 1998; Patsias, Bruelheide, 2013). Its significant expansion, sometimes lasting only limited period (Hulme et al., 1999; Velev, Apostolova, 2008), was observed also in the former pastures of northern Wales (Hill et al., 1992), Polish part of Krkonoše Mts. (Fabiszewski, Wojtuń, 2001), Norway region of Setesdal Vesthei (Austrheim et al., 2007; Speed et al., 2014), and Polish part of Tatra Mts. (Korzeniak, 2016).

Acknowledgements

This study was supported by the Slovak Agency for Science (VEGA) grant 2/0132/18.

References

Abeli, T., Rossi, G., Gentili, R., Mondoni, A. & Cristofanelli P. (2012). Response of alpine plant flower production to temperature and snow cover fluctuation at the species range boundary. *Plant Ecol.*, 213, 1–13. DOI: 10.1007/s11258-011-0001-5.

Austrheim, G., Mysterud, A., Hassel, K., Evju, M. & Okland R.H. (2007). Interactions between sheep, rodents, graminoids, and bryophytes in an oceanic alpine ecosystem of low productivity. *Ecoscience*, 14, 178–187. DOI:10.2980/1195-6860(2007)14[178:IBSRGA]2.0.CO ;2.

Balabukh, V.O. & Lukianets O.I. (2015). Zmina klimatu ta yoho naslidky u Rakhivskomu rayoni Zakarpatskoyi oblasti. *Hydrol. Hydrochem. Hydroecol.*, 37, 132–148.

Baur, B., Cremene, C., Groza, G., Schileyko, A.A., Baur, A. & Erhardt A. (2007). Intensified grazing affects endemic plant and gastropod diversity in alpine grasslands of the Southern Carpathian mountains (Romania). *Biologia (Bratislava)*, 62, 438–445. DOI: 10.2478/s11756-007-0086-4.

Bohuš, I. (1966). *History of Liptov Tatras (in Slovak)*. Tatranská Lomnica: ŠL Tanap-u.

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

Britton, A.J., Beale, C.M., Towers, W. & Hewison R.L. (2009). Biodiversity gains and losses: evidence for homogenisation of Scottish alpine vegetation. *Biol. Conserv.,* 142, 1728–1739. DOI: 10.1016/j.biocon.2009.03.010.

Bruelheide, H. (2003). Translocation of a montane meadow to simulate the potential impact of climate change. *Appl. Veg. Sci.*, 6, 23–24. DOI: 10.1111/j.1654-109X.2003.tb00561.x.

Cannone, N., Sgorbati, S. & Guglielmin M. (2007). Unexpected impacts of climate change on alpine vegetation. *Frontiers in Ecology and the Environment*, 5, 360–364. DOI : 10.1890/1540-9295(2007)5[360:UIOCCO]2.0.CO ;2.

Carbognani, M., Tomaselli, M. & Patraglia A. (2014). Current vegetation changes in an alpine late snowbed community in the south-eastern Alps (N-Italy). *Alpine Botany*, 124, 105–113. DOI: 10.1007/s00035-014-0135-x.

Catorci, A., Gatti, R. & Cesaretti S. (2012). Effect of sheep and horse grazing on species and functional composition of sub-Mediterranean grasslands. *Appl. Veg. Sci.*, 15, 459–469. DOI: 10.1111/j.1654-109X.2012.01197.x.

Coldea, G. (2003). The alpine flora and vegetation of the southeastern Carpathians. In L. Nagy, G. Grabherr, Ch. Korner & D.B.A. Thompson (Eds.), *Alpine biodiversity in Europe* (pp. 65–73). Berlin: Springer-Verlag.

Cook, W. (1966). Factors affecting utilization of mountain slopes by cattle. *J. Range Manag.*, 19, 200–204. DOI: 10.2307/3895647.

Crofts, A. & Jefferson R.G. (Eds.) (1999). *The lowland grassland management.* Peterborough: English Nature/The Wildlife Trusts.

Czortek, P., Kapfer, J., Delimat, A., Eycott, A.M., Grytnes, J.-A., Orczewska, A., Ratyńska, H., Zieba, A. & Jaroszewicz B. (2018). Plant species composition shifts in the Tatra Mts as a response to environmental change: a resurvey study after 90 years. *Folia Geobot.*, 53, 333–348. DOI: 10.1007/s12224-018-9312-9.

Čepčeková, E. (2013). Atmospheric precipitation and storms (in Slovak). In S. Bičárová (Ed.), *Observatory of SAS at Skalnaté pleso. 70 years of meteorological measurements* (pp. 37– 42). Stará Lesná: Geofyzikálny ústav SAV.

Dahl, E. (1998). *The phytogeography of Northern Europe: British Isles, Fennoscandia, and Adjacent Areas*. Cambridge: Cambridge University Press.

Daniëls, F.J.A., Molenaar, J.G., Chytrý, M. & Tichý L. (2011). Vegetation change in Southeast Greenland? Tasiilaq revisited after 40 years. *Appl. Veg. Sci.,* 14, 230–241. DOI: 10.1111/j.1654-109X.2010.01107.x.

DeCáceres, M. & Legendre P. (2009). Associations between species and groups of sites: indices and statistical inference. *Ecology*, 90, 3566–3574. DOI: 10.1890/08-1823.1.

Ditsch, D.C., Schwab, G., Green, J.D., Johns, J.T., Coleman, R., Hutchens, T. & Piercy L. (2006). *Managing steep terrain for livestock forage production*. Lexington: College of Agriculture, University of Kentucky.

Dúbravcová, Z. (1976). *Subalpine and alpine vegetation in Kamenistá and Gáborova dolina valleys (Western Tatras) (in Slovak)*. Msc, PhD thesis, Faculty of Natural Sciences, Comenius University, Bratislava.

Dúbravcová, Z. (2007). *Salicetea herbaceae* (in Slovak). In J. Kliment & M. Valachovič (Eds.), *Plant communities of Slovakia. 4. Alpine vegetation* (pp. 251–282). Bratislava: Veda, vydavateľstvo SAV .

Dye, D.G. (2002). Variability and trends in the annual snow-cover cycle in Northern Hemisphere land areas, 1972-2000. *Hydrol. Process.*, 16, 3065– 3077. DOI: 10.1002/hyp.1089.

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

Elmendorf, S.C., Henry, G.H.R., Hollister, R.D., Bjork, R.G., Bjorkman, A.D., Callaghan, T.V., Collier, L.S., Cooper, E.J., Cornelissen, J.H.C., Day, T.A., Fosaa, A.M., Gould, W.A., Gretarsdottir, J., Harte, J., Luise, H., Hik, D.S., Hofgaard, A., Jarrad, F., Jonsdottir, I.S., Keuper, F., Klanderud, K., Klein, J.A., Koh, S., Kudo, G., Lang, S.I., Loewen, V., May, J.L., Mercado, J., Michelsen, A., Molau, U., Myers-Smith, I.H., Oberbauer, S.F., Pieper, S., Post, E., Rixen, Ch., Robinson, C.H., Schmidt, N.M., Shaver, G.R., Stenstrom, A., Tolvanen, A., Totland, O., Troxler, T., Wahren, C.-H., Webber, P.J., Welker, J.M. & Wookey P.A. (2012). Global assessment of experimental climate warming on tundra vegetation: heterogeneity over space and time. *Ecol. Lett.*, 15, 164–175. DOI : 10.1111/j.1461-0248.2011.01716.x.

Elumeeva, T.G., Onipchenko, V.G., Egorov, A.V., Khubiev, A.B., Tekeev, D.K., Soudzilovskaia, N.A. & Cornelissen J.H.C. (2013). Long-term vegetation dynamic in the Northwestern Caucasus: which communities are more affected by upward shifts of plant species? *Alpine Botany,* 123, 77– 85. DOI: 10.1007/s00035-013-0122-7.

Engler, R., Randin, C.F., Thuiller, W., Dullinger, S., Zimmermann, N.E., Araujo, M.B., Pearman, P.B., Le Lay, G., Piedallu, C., Albert, C.H., Choler, P., Coldea, G., De Lamo, X., Dirnböck, T., Gégout, J.-C., Gómez-García, D., Grytnes, J.-A., Heegaard, E., Høistad, F., Nogués-Bravo, D., Normand, S., Puşcaş, M., Sebastiá, M.-T., Stanisci, A., Theurillat, J.-P., Trivedi, M.R., Vittoz, P. & Guisan A. (2011). 21st century climate change threatens mountain flora unequally across Europe. *Glob. Change Biol.*, 17, 2330– 2341. DOI: 10.1111/j.1365-2486.2010.02393.x.

Erschbamer, B., Unterluggauer, P., Winkler, E.& Mallaun M. (2011). Changes in plant species diversity revealed by long-term monitoring on mountain summits in the Dolomites (northern Italy). *Preslia*, 83, 387–401.

Fabiszewski, J. & Wojtuń B. (2001). Contemporary floristic changes in the Karkonosze Mts. *Acta Soc. Bot. Pol.*, 70, 237–245. DOI: 10.5586/asbp.2001.031.

Gellman, A. & Hill J. (2007). *Data analysis using regression and multilevel/hierarchical models*. Cambridge: Cambridge University Press.

Grytnes, J.-A., Kapfer, J., Jurasinski, G., Birks, H.H., Henriksen, H., Klanderud, K., Odland, A., Ohlson, M., Wipf, S. & Birks H.J.B. (2014). Identifying the driving factors behind observed elevational range shifts on European mountains. *Glob. Ecol. Biogeogr.*, 23, 876–884. DOI : 10.1111/geb.12170.

Hebák, P., Hustopecký, J., Pecáková, I., Plašil, M., Průša, M., Řezanková, H., Vlach, P. & Svobodová A. (2007). *Multivare statistical methods (3) (in Czech)*. Praha: Informatorium.

Hegedüšová, K. (2007). Central database of phytosociological samples (CDF) in Slovakia (in Slovak). *Bulletin Slovenskej Botanickej Spoločnosti*, 29, 124–129.

Hennekens, S. & Schaminée J.H.J. (2001). TURBOVEG, a comprehensive data base management system for vegetation data. *J.Veg.Sci.*, 12. 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.

Hill, M.O., Evans, D.F. & Bell S.A. (1992). Long-term effects of excluding sheep from hill pastures in north Wales. *J. Ecol.*, 80, 1–13. DOI: 10.2307/2261058.

Holzinger, B., Hülber, K., Camenisch, M., Grabherr G. (2008). Changes in plant species richness over the last century in the eastern Swiss Alps: elevational gradient, bedrock effects and migration rates. *Plant Ecol.,* 195, 179–196. DOI: 10.1007/s11258-007-9314-9.

Horák, J. (1970). *Geobiocoenosis of timber line. Part I. – Western Tatras, Jamnická and Račková dolina valleys (in Czech).* Brno: University of Agriculture.

Hulme, P.D., Pakeman, R.J., Torvell, L., Fischer, J.M. & Gordon I.J. (1999). The effects of controlled sheep grazing on the dynamics of upland *Agrostis-Festuca* grassland. *J. Appl. Ecol.*, 36, 886–900. DOI: 10.1046/j.1365-2664.1999.00452.x.

Jarolímek, I., Šibík, J., Hegedüšová, K., Janišová, M., Kliment, J., Kučera, P., Majeková, J., Michalková, D., Sadloňová, J., Šibíková, J., Škodová, I., Uhlířová, J., Ujházy, K., Ujházyová, M., Valachovič, M. & Zaliberová M. (2008). A list of vegetation units of Slovakia. In I. Jarolímek & J. Šibík (Eds.), *Diagnostic, constant and dominant species of higher vegetation units of Slovakia* (pp. 295–329). Bratislava: Veda, vydavateľstvo SAV .

Jurasinski, G. & Kreyling J. (2007). Upward shift of alpine plants increases floristic similarity of mountain summits. *J. Veg. Sci.*, 18, 711– 718. DOI: 10.1111/j.1654-1103.2007.tb02585.x.

Kanka, R., Barančok, P. & Krajčí J. (2011). Research of the vascular plants diversity in alpine belt of the Tatry Mts. as a platform for monitoring of the climate changes (in Slovak). *Životné Prostredie*, 45, 89–92.

Kapfer, J., Virtanen, R. & Grytnes J.-A. (2012). Changes in arctic vegetation on Jan Mayen Island over 19 and 80 years. *J. Veg. Sci.*, 23, 771– 781. DOI: 10.1111/j.1654-1103.2012.01395.x.

Klanderud, K. & Birks H. (2003). Recent increases in species richness and shifts in altitudinal distributions of Norwegian mountain plants. *Holocene*, 13, 1–6. DOI : 10.1191/0959683603hl589ft.

Klein, J.A., Harte, J. & Zhao X.Q. (2004). Experimental warming causes large and rapid species loss, dampened by simulated grazing, on the Tibetan Plateau. *Ecol. Lett.*, 7, 1170–1179. DOI: 10.1111/j.1461-0248.2004.00677.x.

Kliment, J. & Valachovič M. (Eds.) (2007). *Plant communities of Slovakia. 4. Alpine vegetation (in Slovak).* Bratislava: Veda, vydavateľstvo SAV .

Kobiv, Y. & Nesteruk Y. (2001). *Pedicularis oederi* (Scrophulariaceae) in the Chornohora Mts (Ukrainian Carpathians): distribution, biology, ecology and threat. *Polish Bot. J.*, 46, 241–250.

Kobiv, Y. (2014). Response of rare plant species to current changes of vegetation in the Ukrainian Carpathians. *Forum Carpaticum 2014: local responses to global challenges* (pp. 56–58). Lviv.

Kobiv, Y. (2016). *Saxifraga aizoides* (Saxifragaceae) in Ukraine. *Polish Bot. J.*, 61, 65–71. DOI: 10.1515/pbj-2016-0004.

Kobiv, Y. (2017). Response of rare alpine plant species to climate change in the Ukrainian Carpathians. *Folia Geobot.*, 52 , 217–226. DOI: 10.1007/s12224-016-9270-z.

Kobiv, Y. (2018). Trends in population size of rare plant species in the alpine habitats of the Ukrainian Carpathians under climate change. *Diversity*, 10(3), 62. DOI: 10.3390/d10030062.

Kollár, J., Kanka, R. & Barančok P. (2016). Sites of the GLORIA Initiative in Slovakia Included into LTER Network (in Slovak). *Životné Prostredie*, 50, 51–53.

Korzeniak, J. (2016). Mountain *Nardus stricta* grasslands as a relic of past farming – the effects of grazing abandonment in relation to elevation and spatial scale. *Folia Geobot.*, 51, 93–113. DOI : 10.1007/s12224-016-9246-z.

Krajina, V. (1933). Die Pflanzengesellschaften des Mlynica-Tales in den Vysoké Tatry (Hohe Tatra). *Beih. Bot. Centralbl.*, 51, 774–957.

Kucharzyk, S. & Augustyn M. (2010). Stability of mountain glades in the Bieszczady National Park (in Polish). *Roczniki Bieszczadzkie*, 18, 45–58.

Kudernatsch, T., Beck, S., Krenzer, M., Fischer, A., Bernhardt, M., Franz, H., Vogel, M. & Abs C. (2005). Recent changes in species composition and species richness of alpine grasslands in Berchtesgaden biosphere reserve and national park. *2nd and 3rd GLOCHAMORE workshops* (pp. 103–115). L'Aquila.

Kudo, G., Amagai, Y., Hoshino, B. & Kaneko M. (2011). Invasion of dwarf bamboo into alpine snow-meadows in northern Japan: pattern of expansion and impact on species diversity. *Ecology and Evolution,* 1, 85– 96. DOI: 10.1002/ece3.9.

Kyyak, V., Bilonoha, V., Dmytrach, R., Gynda, L., Nesteruk, Y. & Shtupun V. (2014). Changes in plant population pattern under the natural and man-induced ecosystem transformations of high mountain zone of Ukrainian Carpathians. *Forum Carpaticum 2014: local responses to global challenges* (pp. 58–60). Lviv.

Marhold, K. & Hindák F. (Eds.) (1998). *Checklist of non-vascular and vascular plants of Slovakia*. Bratislava: Veda, vydavateľstvo SAV.

Matteodo, M., Wipf, S., Stöckli, W., Rixen, C. & Vittoz P. (2013). Elevation gradient of successful plant traits for colonizing alpine summits under climate change. *Environmental Research Letters*, 8, 024043. DOI: 10.1088/1748-9326/8/2/024043.

Matteodo, M., Ammann, K., Verrecchia, E.P. & Vittoz P. (2016). Snowbeds are more affected than other subalpine-alpine plant communities by climate change in the Swiss Alps. *Ecology and Evolution*, 6, 6969 – 6982. DOI: 10.1002/ece3.2354.

Micu, D. (2009). Snow pack in the Romanian Carpathians under changing climatic conditions. *Meteorology and Atmospheric Physics*, 105, 1–16. DOI: 10.1007/s00703-009-0035-6.

Nemčok, J. (1994). Geological bedrock (in Slovak). In I. Vološčuk (Ed.), *Tatras National Park. Biosphere Reserve* (pp. 14–24). Martin: GRADUS.

Norton, L.R., Murphy, J., Reynolds, B., Marks, S. & Mackey E.C. (2009*). Countryside survey: Scotland results from 2007*. Battleby: NERC Centre for Ecology and Hydrology & The Scottish Government, Scottish Natural Heritage.

Odland, A., Høitomt, T. & Olsen S.L. (2010). Increasing vascular plant richness on 13 high mountain summits in southern Norway since the early 1970s. *Arcti. Antarct. Alp. Res.*, 42, 458–470. DOI: 10.1657/1938-4246-42.4.458.

Oksanen, J., Guillaume Blanchet, F., Friendly, M., Kindt, R., Legendre, P., McGlinn, D., Minchin, P.R., O´Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H., Szoecs, E. & Wagner H. (2017). *Vegan: Community ecology package*. R package version 2.4-4.

Olsen, S.L. & Klanderud K. (2014). Biotic interactions limit species richness in an alpine plant community, especially under experimental warming. *Oikos*, 123, 71–78. DOI: 10.1111/j.1600-0706.2013.00336.x.

Pakeman, R.J. (2004). Consistency of plant species and trait responses to grazing along a productivity gradient: A multi-site analysis. *J. Ecol.*, 92, 893–905. DOI: 10.1111/j.0022-0477.2004.00928.x.

Palaj, A. & Kollár J. (2017). Contribution to the knowledge of the alpine vegetation of the Western Tatras (in Slovak). *Phytopedon*, 16(1), 9–13.

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.

Patsias, K. & Bruelheide H. (2013). Climate change – Bad news for montane forest herb layer species? *Acta Oecol.*, 50, 10–19. DOI: 10.1016/j.actao.2013.02.004.

Pauli, H., Gottfried, M., Reiter, K., Klettner, C. & Grabherr G. (2007). Signals of range expansions and contractions of vascular plants in the high Alps: observations (1994–2004) at the GLORIA master site Schrankogel, Tyrol, Austria. *Global Change Biology*, 13, 147–156. DOI: 10.1111/j.1365-2486.2006.01282.x.

Pauli, H., Gottfried, M., Dullinger, S., Abdaladze, O., Akhalkatsi, M., Alonso, J.L.B., Coldea, G., Dick, J., Erschbamer, B., Calzado, F.R., Ghosn, D., Holten, J.I., Kanka, R., Kazakis, G., Kollar, J., Larsson, P., Moiseev, P., Moiseev, D., Molau, U., Mesa, J.M., Nagy, L., Pelino, G., Puşcaş, M., Rossi, G., Stanisci, A., Syverhuset, A.O., Theurillat, J.P., Tomaselli, M., Unterluggauer, P., Villar, L., Vittoz, P. & Grabherr G. (2012). Recent plant diversity changes on Europe's Mountain Summits. *Science*, 336(6079), 353–355. DOI : 10.1126/science.1219033.

Pickering, C., Green, K., Barros, A.A. & Venn S. (2014). A resurvey of late-lying snow patches reveals changes in both species and functional composition across snowmelt zones. *Alpine Botany,* 124, 93–103. DOI: 10.1007/s00035-014-0140-0.

Pielou, E.C. (1966). The measurement of diversity in different types of biological collections. *J. Theor. Biol.*, 13, 131–144. DOI: 10.1016/0022-5193(66)90013-0.

Plesník, P. (1974). Physical geography (in Slovak). In M. Konček (Ed.), *Tatras climate* (pp. 17–26). Bratislava: Veda, vydavateľstvo SAV .

Press, M.C., Potter, J.A., Burke, M.J.W., Callaghan, T.V. & Lee J.A. (1998). Responses of a subarctic dwarf shrub heath community to simulated environmental change. *J. Ecol.*, 86, 315–327. DOI: 10.1046/j.1365-2745.1998.00261.x.

QGIS Development Team (2019). *QGIS Geographic Information System. Open Source Geospatial Foundation Project*. http://qgis.osgeo.org

R Core Team (2017). *R: A language and environment for statistical computing (online)*. Vienna: R Foundation for Statistical Computing. https://www.gbif.org/tool/81287/r-a-language-and-environment-for-statistical-computing

Ross, L.C., Woodin, S.J., Hester, A.J., Thompson, D.B.A. & Birks H.J.B. (2012). Biotic homogenization of upland vegetation: patterns and drivers at multiple spatial scales over five decades. *J. Veg. Sci.,* 23, 755– 770. DOI: 10.1111/j.1654-1103.2012.01390.x.

Sammul, M., Kull, K., Oksanen, L. & Veromann P. (2000). Competition intensity and its importance: results of field experiments with *Anthoxanthum odoratum*. *Oecologia*, 125, 18-25. DOI: 10.1007/PL00008887.

Sandvik, S.M. & Odland A. (2014). Changes in alpine snowbed-wetland vegetation over three decades in northern Norway. *Nord. J. Bot.,* 32, 377– 384. DOI: 10.1111/j.1756-1051.2013.00249.x.

Speed, J.D.M., Martinsen, V., Mysterud, A., Mulder, J., Holand, O. & Austrheim G. (2014). Long-term increase in aboveground carbon stocks following exclusion of grazers and forest establishment in an alpine ecosystem. *Ecosystems*, 17, 1138–1150. DOI : 10.1007/s10021-014-9784-2.

Stöckli, V., Wipf, S., Nilsson, C. & Rixen C. (2011). Using historical plant surveys to track biodiversity on mountain summits. *Plant Ecol. Divers,* 4, 415– 425. DOI: 10.1080/17550874.2011.651504.

Šibík, J. (2012). Slovak vegetation database. In J. Dengler, J. Oldeland, F. Jansen, M. Chytry, J. Ewald, M. Finckh, F. Glockler, G. Lopez-Gonzalez, R.K. Peet & J.H.J. Schaminee (Eds.), Vegetation databases for the 21st century. *Biodiversity & Ecology*, 4(1), 429–429. DOI : 10.7809/b-e.00216.

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

Tilman, D. (1988). *Plant strategies and the dynamics and structure of plant communities*. Princeton: Princeton University Press.

Turečková, J. (1974). *Subalpine and alpine vegetation of Jamnícka dolina valley (Western Tatras) (in Slovak)*. Diploma work (msc.), Faculty of Natural Sciences, Comenius University, Bratislava.

Vanneste, T., Michelsen, O., Graae, B.J., Kyrkjeeide, M.O., Holien, H., Hassel, K., Lindmo, S., Kapas, R.E. & De Frenne P. (2017). Impact of climate change on alpine vegetation of mountain summits in Norway. *Ecol. Res.*, 32, 579–593. DOI : 10.1007/s11284-017-1472-1.

Vassilev, K., Pedashenko, H., Nikolov, S.C., Apostolova, I. & Dengler J. (2011). Effect of land abandonment on the vegetation of upland semi-natural grasslands in the Western Balkan Mts., Bulgaria. *Plant Biosyst.*, 145, 654–665. DOI: 10.1080/11263504.2011.601337.

Velev, N. & Apostolova I. (2008). Successional changes of *Nardus stricta* communities in the Central Balkan Range (Bulgaria). *Phytol. Balc.,* 14(1),75–84.

Virtanen, R., Eskelinen, A. & Gaare E. (2003). Long-term changes in alpine plant communities in Norway and Finland. In L. Nagy, G. Grabherr, Ch. Korner & D.B.A. Thompson (Eds.), *Alpine biodiversity in Europe* (pp. 411–422). Berlin: Springer-Verlag.

Vittoz, P., Bodin, J., Ungricht, S., Burga, C. & Walther G. R. (2008). One century of vegetation change on Isla Persa, a nunatak in the Bernina massif in the Swiss Alps. *J. Veg. Sci.*, 19, 671– 680. DOI: 10.3170/2008-8-18434.

Vittoz, P., Randin, C., Dutoit, A., Bonnet, F. & Hegg O. (2009). Low impact of climate change on subalpine grasslands in the Swiss Northern Alps. *Glob. Change Biol.,* 15, 209– 220. DOI: 10.1111/j.1365-2486.2008.01707.x.

Weselovska, M. (2009). Changes in meadow vegetation of the Western Tatra Mts and their foreland in the last half century (in Polish). In M. Guzik (Ed.), *Long-term changes in nature and management of Tatra National Park* (pp. 91–104). Zakopane: Wydawnictwa Tatrzańskiego Parku Narodowego.

Wickham, H. (2007). Reshaping data with the reshape package. *Journal of Statistical Software*, 21(12), 1–20. http://www.jstatsoft.org/

Wilcox, B.P. & Wood K.M. (1988). Hydrologic impacts of sheep grazing on steep slopes in semiarid rangelands. *J. Range Manag.,* 41(4), 303–306. DOI: 10.2307/3899383.

Wilson, S. & Nilsson C. (2009). Arctic alpine vegetation change over 20 years. *Glob. Change Biol.*, 15, 1676–1684. DOI: 10.1111/j.1365-2486.2009.01896.x.

Windmaißer, T. & Reisch C. (2013). Long-term study of an alpine grassland: local constancy in times of global change. *Alpine Botany,* 123, 1– 6. DOI: 10.1007/s00035-013-0112-9.

Winter, B. (2013). *Linear models and linear mixed effects models in R with linguistic applications.* arXiv:1308.5499. http://arxiv.org/pdf/1308.5499.pdf

Ekológia (Bratislava)
Vol. 38, No. 4, p. 336–352, 2019
DOI:10.2478/eko-2019-0025

FLORISTIC DISTRIBUTION ACCORDING TO THE EDAPHIC PARAMETERS OF A STEPPE ZONE, CASE OF STUDY: THE NATURE RESERVE "EL-MERGUEB" M'SILA, ALGERIA

AMINA ADJABI[1*], HACHEMI SIDI[1], RABAH BOUNAR[2], HAMID REZA NASERI[3]

[1]Laboratory of Research in Ecology and Environment (LREE), University Abdurrahman Mira of Bejaia, Bejaia, Algeria; e-mail: adjabi_amina@yahoo.com, shaokas@yahoo.fr
[2]Department of Natural and Life Sciences, University Mohamed Boudiaf of M'sila, M'sila, Alegria; e-mail: rabah.bounar@univ-msila.dz
[3]International Desert Research Center (IDRC) University of Tehran, Tehran, Islamic Republic of Iran; e-mail: hrnaseri@ut.ac.ir

*Author for correspondence

Abstract

Adjabi A., Sidi H., Bounar R., Naseri H.R.: Floristic distribution according to the edaphic parameters of a steppe zone, case of study: the Nature Reserve "El-Mergueb" M'sila, Algeria. Ekológia (Bratislava), Vol. 38, No. 4, p. 336–352, 2019.

The natural site of El-Mergueb, is one of the specific steppe ecosystem, that occupies an area of 16,481 ha; it is among the particular ecosystems in Algeria and is unique in northern Africa. It is located in the geographical limit of the Tell Atlas and the Saharan Atlas. It is at an altitude of 550 to 800 m, with a typically arid climate. The faunistic and floristic richness, quite specifically, contains species protected and classified in the IUCN: *Chlamydotis* and *Gazella*, as well as an autochthonous plant diversity: *Stipa, Pistacia* and *Ziziphus*. This study is based on the analysis of the vegetation by characterizing the associated soil type through the monitoring of the Eco-pedological relation during two seasons in three years: 2015–2016–2017. Nine soil profiles with a collection of twenty-one samples for each station of experimentation were collected. The results thus obtained showed that the content of the parameters analyzed is between low and average of the three stations. The average of the floristic analysis of the study area made it possible to define 34 botanical families spread over 69 botanical genus and 275 species. The analysis and the description of a natural space is important to establish a typology which is that is the basis for the development of management plans to conserve this category of protected areas. The FAC (Factorial Analysis of Correspondence) is relative to the individualization and the typology for three sequences, which corresponds to the groups (A), (B) and (C) also for the places in arid climate variants with very cold winters.

Key words: protected areas, steppe soils, anthropic pressure, arid climate, steppe vegetation.

Introduction

Climate change is anticipated to have far reaching effects on the sustainable development of developing countries, including their ability to attain the United Nations Millennium

Development Goals by 2015 (United Nations Framework Convention on Climate Change, 2007). In recent years, climate change caused dramatic shifts in species' distributions and extinctions, particularly across fragmented or vulnerable ecosystems (Hilbert et al., 2007). The need to protect diverse biological resources from the ongoing development pressures is one of today's most pressing environmental challenges. In response, "ecosystem services" has emerged as a conservation framework that links human economies and natural systems through the benefits that people receive from nature (Stephen, 2015). Natural or biological diversity on earth is an important prerequisite for humans to exist, as it provides valuable ecosystem services. Changes of biodiversity affect ecosystem functions and services, and consequently, human well-being in urban areas. This is an example of a cycle of socio-ecological interactions within global environmental change, significantly intensified over the last few decades (Breuste et al., 2013). However, there are many threats to biodiversity, including loss of habitat, overexploitation, pollution, alien species and climate change. In order to reduce the rate of biodiversity loss significantly the Convention on Biological Diversity was adopted at the Rio Conference 1992, forming the core of the international Regime on global biodiversity (Wetang·ula, 2009).

Biodiversity loss and the erosion of the capacity of ecosystems to deliver services often respond in similar ways to shared drivers; however, the relationship between them is not simple, and may be different for the various dimensions of biodiversity. For example, the links between local species extinctions and reduced capacity to deliver ecosystem services remain, in many cases, elusive (Leadley et al., 2010). The Convection on Biological Diversity (CBD) defines biodiversity as "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems." It is the variety of life on earth at all levels, from genes to worldwide populations of the same species; from communities of species sharing the same small area of habitat to worldwide ecosystems.

Conservation and loss: Biological diversity can be measured in terms of different components (genetic, population/species, and community/ecosystem), and composition refers to the identity and variety of elements in each of the biodiversity components. Structure refers to the physical organization or pattern of the elements. Function refers to ecological and evolutionary processes acting among the elements (Wetang·ula, 2009). The extent of protected areas (PA) in Africa has been increased over the last few years and currently represents about 5% of land area. However, the data concerning the extent of protected areas in the region differed from one source to another. According to World Resource Institute, there are 746 protected areas in Africa, covering 1.54 million km^2, which is 5.2 per cent of the total land area. Although there has been a rapid increase in the of PAs, which has been observed during the last 10–20 years. Through the region, there tends to be inadequate legislation and ineffective application of the legal measures that do exist, weak institutional support, management that is frequently deficient or even non-existent and inadequate funding are also ubiquitous (Czudek, 2001). Consequently, there is a strong tendency towards "paper parks", whose existence is largely theoretical and not reflected by substantive and durable conservation reserves on the ground (Czudek, 2001). Steppe ecosystems in Algeria cover more than

20 million ha. The combination of many regressive processes, such as woodcutting, wildfires, overstocking and clearing, have contributed to the present steppe conditions. According to the biogeography, these steppe rangelands belong to the Mediterranean basin, one of the 25 biodiversity hotspots of the globe. Mediterranean ecosystems are considered among the most vulnerable to global change. These ecosystems have evolved under the long history of grazing, and over time, the various farming methods in these areas have shaped the landscapes and resources that are found there. The evaluation of the effect of grazing on plant diversity and structure has been studied extensively in rangeland ecosystems. In more humid environments (productive), grazing is predicted to increase species diversity. Where as, in arid environments grazing can have a negative effect on diversity. In semi-arid Mediterranean grasslands, grazing reduces plant diversity. North African arid grasslands exhibits the same trend where grazing has decreased diversity (Merdas et al., 2017). Mediterranean bioclimates are characterized by winter rains and summer drought (LeHouérou, 2005a,b). In parts of North Africa, a drought occurs when no rain has fallen for at least two years. Perhaps, then, a drought can be defined as a period during which rainfall is insufficient to meet the needs of plants. Droughts meeting this definition have led to the ongoing debates on desertification (Gamoun, 2013).

In this context, this study describes the importance to analyze and investigate the relationship between soil characteristics with plant species inside one of the three municipalities constituting the nature reserve of El- Mergueb, characterized by a typical arid climate, to determine the most important factors affecting the separation of vegetation types. Second aim was to identify the soil characteristics that are an indicator of specific species who characterize this particular ecosystem. Moreover, to have information about the relationship between soil and vegetation of the area of study, it will be possible to apply these results to explain the importance of conserving the biodiversity inside the nature reserves and the rangelands in arid regions; also, for other similar regions exposed to severe arid climate and extensive grazing, to recommend suitable guidance for management and protection of particular and vulnerable arid ecosystem.

Material and methods

Study area

The natural site of El-Mergueb is located in the steppe routes of the highlands of Algeria, it extends between the coordinates of Lambert relative to the topographic map of, Aïn El-Hadjel to (1/50.000, following: X (608.5 and 626.7) km and Y (243.6–263.8) km. It straddles three steppe communes on the northern side of the Saharan Atlas belonging to the M'sila province. A protected, area according to the law 83–03 of 5th February 1983 (joradp.dz), relative to the protection of the environment, which elaborates the first legal instrument for the creation of national parks and nature reserves (Kaabeche, 2003) characterized by a potential. The three experimental stations lie between latitudes (35°36'12,6"N & 35°35'05,7"N) and longitudes (03°56'23,8"E & 03°58'08,7"E) with an altitude of 575 and 634 m located in the municipality of Ain EL-Hadjel (Fig. 1).

The climate is characterized by hot dry summers and very cold winters; so according to Emberger (1954) and M'sila Weather Station (2017), it has a Saharan superior Mediterranean bioclimate. As per the climatic data of M'sila, a mean minimum temperature of -0.5 °C in February and a mean maximum temperature of 46.2 °C characterize the study area in July, with an annual mean temperature of 19.61 °C, showing very hot summers and very cold winters. The rainfall in the region is erratic and irregular. The average annual rainfall is between (121 and 181 mm), which places the study area in an arid bioclimatic variant to very cold winter. Geomorphologically, the study area is a succession of undulations that dominate the relief in the northern part of the reserve forming elongated depressions that converge and concentrate on the surface flows in the form of fine particles, eroded and transported; this zone presents a dominant lithology or limestone where the topographic depressions consist of polygenic quaternary fillings (presence of sandy-sandy facies, marls, limestone, clays and sandy-clayey soils) (Boudjaja et al., 2010). Characterized by a high density of flow on the slopes, the main troughs oriented South-West-North-East drain areas of spreading of very low slope. The breaks of slopes between the top parts of the reliefs and the foothills are the fissured cliffs with important joints that appear in the landscape feed the slopes in blocks, gravel and pebbles resulting from the collapse of the overhangs (Boudjaja et al., 2010).

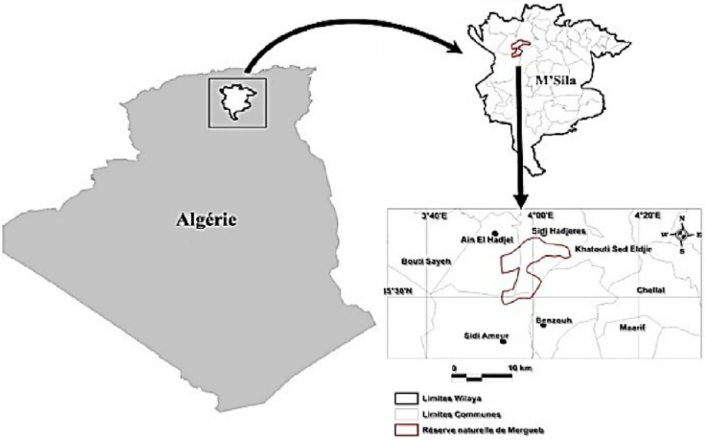


Fig. 1. Map of the protected area: «EL – MERGUEB» with the geographical situation of the study area: (Ain El- Hadjel) (Google Earth).

Vegetation survey and analysis

The present study was carried out from March 2015 through February, 2017. Three different stations (with its different habitats), were selected after general survey of the studied area. The phyto-sociological method were used to study vegetation within the selected three locations. A sampling site was selected systematically according to the distribution of species, habitat differences and the vegetation homogeneity, and also the places of soil profiles. Numbers of stands were determined according to different habitats in each station. Present species were recorded and their cover was evaluated visually as a percentage of the ground surface. The vegetation parameters included listing of all species and life forms. Species identification and nomenclature (Maire, 1926, 1928, 1952, 1987), (Quezel, Santa, 1962, 1963). The nomenclature of species adopted is that of the "Med-Cheklist". Plant cover was estimated and the Specimens of each species were collected, identified and supplemented in laboratory of research in Ecology and Environment, University of Bejaia. The chorological types of the various taxa identified were attributed according to the indications of the flora consulted. In the analysis of floristic records, special attention has been paid to endemic and or rare species. It fluctuates between 300 to 400 m² for forest vegetation and between 5 and 10 m² in terms of rupicolous vegetation. We have made the floristic list of our three stations (A, B and C). The surveys were carried out during field campaigns during the three years from 2015 to 2017. The floristic surveys were conducted according to subjective sampling in all vegetation types in the study area.

Site description (stations)

The choice of stations was based on the durability and persistence of the dominant species. As this index showed, this physiognomy revealed three homogeneous and floristically representative and ecological sites: Formation with: *Artemisia herba-alba* L. (Site A) and formation *Salsola vermiculata* L. and *Anabasis articulata* L. (Site B), *Artemisia campestris* L. and *Zizyphus lotus* L. (Site C). These species formations were selected to carry out this work. In each of these formations, floristic surveys were carried out respectively 13, 10 and 10 associated with soil profiles (Fig. 2).

- **Station A**: *Artemisia herba-alba* L./Ain el Hajal, the geographical coordinates are: (03°56'23,8"E - 35°36'12,6"N). the highest point of this facies is 605 m. The Artemisia stands are welcome, with a large aerial biomass and a fairly diverse floristic procession. The average height of Artemisia feet does not exceed 70 cm.

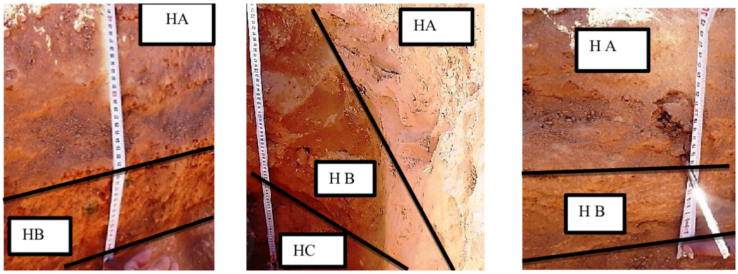
- **Station B**: *Salsola vermiculata* L. and *Anabasis articulata* L. (Maader Nahia/Region (Ain eh Hajal). This station is located in the following geographical coordinates: (03° 56'29,9"E - 35°35'38,6"N). The vegetation is of average height between 30 and 50 cm and the floristic procession is rich. The plant cover is quite important.
- **Station C**: *Artemisia campestris* L. and *Zizyphus lotus* L. stands occupy the following geographical coordinates: (03°58'08,7"E - 35°35'05,7"N). This site is located near the forest house (Ain eh Hajal) at 575 m altitude; the mixed formation in this site is degraded and its height hardly exceeds 30 cm, with a fairly weak floristic procession.

Soil samples and analyses

The sampling adopted in this study consists of exploiting the main soils that characterize the site of our study. Each type of soil is represented by two and representative profiles of each experimental station. The selection of these profiles was based on the soil map of the region of El-Mergueb. In each profile, the samples were taken on two depths (0–40 and 40–60 cm). The choice of these depths is dictated by the fact that they are the parts of the soil where the roots thrive by fixing the plant on its support and feeding to different factors of plant growth: heat, water, all nutrients; it is also where the concept of soil fertility is well connected. Out of a total of (06) to (09), soil samples per station were analyzed, which corresponds to either (21) soil samples in total. The soil profile thus produced is strictly associated with the type of plant formation corresponding to it. Facies with *Artemisia herba-alba* L., facies with *Salsola vermiculata* L., *Anabasis articulata* L. and *Artemisia campestris* L., *Zizyphus lotus* L.

The soils were thus collected at each station (A), (B) and (C) in the fields of Ain el Hadjal regions, namely: Mergueb soula (Fenced station): (Station A), Maader Nahia (Station B), side of the forest house (Station C) (Fig. 3).

Fig. 2. The three experimental station (A), (B) and (C).



(1) Soil profile A. (2) Soil profile B. (3) Soil profile C.

Fig. 3. Soil profiles carried out within 3 experimental stations.

The present study deals with the analysis of soil samples from three different sites in El-Mergueb, which were collected in the period 2015–2017 from Ain El Hadjel commune situated towards south east 35°40′26″ nord, 3°52′54″ est of M'sila city. This study was primarily focused on analyzing the physical and chemical parameters from 21 representative sampling and the analytical results were expected to be representative for the entire field. All samples were collected in a plastic bucket. The analysis were carried at the laboratory of research in Ecology and Environment, University Abdurrahman Mira de Bejaia, Algeria.

Samples collected were thoroughly mixed on a piece of clean cloth, air dried and the lumps were broken using wooden pestle and mortar (Tandon, 1993). The soil profile thus produced is strictly associated with the type of plant formation corresponding to it. Facies of *Artemisia herba-alba* L., facies of: *Salsola vermiculata* L., *Anabasis articulata* L. and *Artemisia campestris* L., *Zizyphus lotus* L. The characterization of each horizon whose objective is to determine the apparent characters of each profile was carried out (color, texture, length, depth, effervescence and the recovery rate) (AFNOR, 1998; Baize, Jabiol, 2011) at the level of each station using the chart of Munsell of color (Pendleton, Nickerson, 1951). The quality of each of the main sand, silt and clay fractions in each soil sample was determined; a 2 mm sieve was used to separate the present sand by wet sieving through a set of nested sieves. The silt and clay in each sample were determined by the pipette that measured the weight percent of sample method (Robinson, 1922). The pH values were determined using pH meter as described by Jackson (1967). For this, 20 g soil sample was mixed with 40 ml distilled water in 1:2 ratio. The suspension was stirred intermittently with glass rod for 30 minutes and left for one hour. The combine electrode was inserted into supernatant and pH was recorded. The electrical conductivity of the soil sample was determined on a digital electrical conductivity bridge for which 20 g soil was added in 40 ml distilled water. The suspension was stirred intermittently for half an hour and kept it for 30 minutes without any disturbances for complete dissolution of soluble salts. The soil was allowed to settle down and then conductivity cell was inserted in the solution to take the reading to record the EC values. The soil organic matter (SOM) was determined by using the modified Walkley-Black method (Chapman, Pratt, 1961). The quantity of organic carbon in the soil was estimated by using the modified Walkey-Black method (Walkey, 1947) as described by Jackson (1962). 1 g finely ground dry soil sample was passed through 0.5 mm sieve without loss and was taken into 500 ml conical flask. To this, 10 ml of 1N potassium dichromate and 20 ml concentrated H_2SO_4 were added and the contents were shaken for a minute and then allowed to set aside exactly for 30 minutes and then 200 ml distilled water, 10 ml phosphoric acid and 1 ml diphenylamine indicator were added. The solution was titrated against standard ferrous ammonium sulphate till color changes from blue violet to green. The blank titration was also carried without soil. Calcium carbonate content ($CaCO_3$%) was determined volumetrically using calcimeter according to Horváth et al. (2005). The CEC was determined according to Sparks (1996). The active carbonate ($CaCO_3$% active) stirred the soil sample with a known amount of ammonium oxalate (N/5) filter. This solution is traced to permanganate before and after its contact with soil sample. The difference between the two titrations corresponds the amount of calcium of carbonate that reacted to oxalate of ammonium (Drouinea modified by Galet, 1951). Bases exchangeable: saturation by a solution of ammonium acetate 1N adjusted to pH = 7. The content of total nitrogen was determined with a modified Kjeldahl's method (Bremner, 1960). The determination of cations has been performed by atomic absorption and flame photometer (Petard, 1993). Cationic exchange capacity (CEC): determined by the sodium acetate method (Petard, 1993). The typology is based essentially on a joint study of edaphic descriptors and individualized plant groups. This type of investigation has a particular interest in pastoral matters. It is the basis of development of model. In other words, this classification allows the manager to better understand the potential of the types of environment with a view to their rational use (pastoral management, reforestation, etc.). The basic data are constituted by a double-entry table or the variables that are arranged in columns and the statements in rows (Bounar, 2014).

Results

Pedology data

After three years of experimentation in the study area in two different seasons (cold and dry), the soil samples were examined (Table shows that the average of clay obtained is moderate and low for the three stations (A),(B) and (C). It ranges from 21 to 25%; also, it is low for the third horizon of station (C) with 11%. For silts, the average obtained is between 29 and 38% for the three stations and 21% for the third horizon of the station (C). The average obtained for the sands are moderate inside our experimental site and range from 30 to 36% for the three stations; however, the station (C), at the third horizon, has a reduced quantity of 29%. The results for the percentage (%) of clay in the three stations confirm the results discussed by Gaouar (2005) when he made his analysis of the typology soil of the nature reserve: El-Mergueb. Also, Pouget (1980) confirms that inside the degraded facies and/or more degraded the content of clay is low. This condition is due to soil degradation. The textures obtained explain the nature of the type of associated vegetation including the station (A), the steppe of Artemisia *Herba-alba* L. is strongly related to areas where storm water or (rainwater) is more or less concentrated at station (B). Steppe of *Salsola vermiculata* L. and *Anabasis articulata* L. is strongly related to low-area texture on a crust and tangentially calcareous gypsum substrate. Station (C) that presents the steppe of *Artemisia campestris* L. and *Zizyphus lotus* L. occupies the texture characterized by the best lands constitute the richest paths with the presence of beds of alluvial oueds (Kaabeche, 2005) (Table 1). An examination of soil samples (Table 2) shows that the values for pH range from 8.4 to 8.61, indicating that the soils are alkaline and under such conditions, the solubility of minerals decreases creating nutrient deficiencies in the soils according to Capot-Rey (1955). The content of organic matter (OM) value ranges from 0.43 to 1.37% except at station (C) with an average of 1.37%. According to Pouget (1980) and Djebaili (1984), the average organic matter is 1 to 2% for steppe soils. The average of total limestone is between 19 and 24% in the three stations showing a calcareous character; according to Capot-Rey (1954) and Djili (2000), the average of the active carbonate is moderate, ranging from 13 to 19% for the

Table 1. The averages obtained after particle size analysis in the laboratory.

Samples of soils	**Clay(%)**	**Silt(%)**	**Sand (%)**	**Soil texture**
SAH1 E	23	38	30	Sandy clay loam
SBH1 E	22	34	33	Sandy clay loam
SCH1 E	24	32	35	Sandy loam
SAH2 E	25	29	32	Sandy loam
SBH2 E	21	34	36	Sandy loam
SC H2 E	19	30	34	Sandy loam
SCH3 E	11	21	29	Loamy sand

Notes: S(A) – station A; S(B) – station B; S(C) – station C; E – three sampling campaigns; H– horizon of each depth.

Table 2. The average obtained after physicochemical analyses.

Sample(s)	**pH H_2O**	**pH KCl**	**$CaCo_3$% total**	**$CaCo_3$% active**	**OM%**	**C%**	**N%**	**C/N**	**EC(mS/cm)**
SAH1 E	8.49	7.55	23	18	0.98	0.57	0.6	0.95	0.98
SBH1 E	8.47	7.82	21	14	0.43	0.25	0.83	0.31	1.65
SCH1 E	8.43	7.8	20	18	1.37	0.8	1.04	0.76	1.45
SAH2 E	8.42	7.81	22	15	0.86	0.5	0.35	1.42	1.38
SBH2 E	8.4	7.78	23	17	0.94	0.55	0.52	1.05	1.21
SCH2 E	8.41	7.56	19	13	0.77	0.45	0.97	0.46	1.20
SCH3 E	8.61	7.66	24	19	0.86	0.5	0.9	0.55	0.91

Notes: pH H_2O – pH water; pH KCl – pH chloride of potassium; $CaCo_3$% – total carbonate ; $CaCo_3$% – active carbonate; OM% – organic matter; C% – total carbon; N% – total nitrogen; C/N – the carbon: nitrogen (C:N) ratio; EC –electric conductivity.

Table 3. The averages obtained after Cation Exchange Capacity Analysis [CEC (meq/100g)].

Samples/ Site	Na+(meq/100g)	Mg+(meq/100g)	K+(meq/100g)	Ca+(meq/100g)	T (meq/100g)
SAH1E	0.02	2.59	0.07	4.07	6.75
SBH1E	0.02	4.17	0.06	4.92	9.17
SCH1E	0.04	3.17	0.06	3.17	6.44
SAH2E	0.01	1.84	0.04	3.25	5.14
SBH2E	0.01	1.95	0.05	3.75	5.76
SC HE	0.01	3,17	0.03	2.33	3.61
SCH3E	0.01	1.4	0.02	2.1	3.53

Notes: Na+ (meq/100g) – exchangeable sodium; Mg+ (meq/100g) – exchangeable magnesium; K+ (meq/100g) – exchangeable potassium; Ca+ (meq/100g) – exchangeable calcium; T (meq/100g) – total of exchangeable cations.

three stations; according to Bonneau et al. (1979), if pH > 7, soils will be rich in active carbonate. Electrical Conductivity average value ranges from 0.91 to 1.65 mS/cm (Table 2). Electrical conductivity is used to estimate the soluble salt concentrations in soil and is commonly used as a measure of salinity. Soil with EC below 0.4 mS/cm are considered marginally saline or non-saline, while soils above 0.8 mS/cm are considered severely saline. In the soils under analysis, the average were found to be non-saline, except for the second horizon of the station (A) and the second horizon of station (C) and first horizon of the station(B) having values 1.38—1.65 and 1.45 mS/cm (Table 2). The measured electrical conductivity reveals unsalted soils for most of our stations except for the station's horizon of 02. For the total carbon, average is fairly low for the three stations and the average of total nitrogen except for the station's horizon one (C), the average of which is 1.04% which is moderate according to Djebaili (1984). The carbon/nitrogen ratio, which shows the fertility of the soil cover, is also low, with an average of all stations being 1% lower, except for two of the stations (A) and (B) being 1.05 and 1.42% considered moderate(classification of C/N ratios in soils is defined in the UNDP project/FAO Gui72/004). The organic carbon (%) ranges from 0.25 to 0.8%, considered to be low in all three stations (Table 2). The average of exchangeable cation in all stations is between low to moderate The average of the two exchangeable Sodium and Potassium cations range < 0.5, for the averages of the two exchangeable cations Magnesium and Calcium is between 2.1 to 4.92 meq/100g and between 1.4 to 4.17 meq/100g, which prove the saturation of Calcium in the soils It is the case of the majority of degraded and overgrazing steppe soils in arid regions with herbaceous vegetation according to Pouget (1980) and Kaabeche (1990, 2005), where the capacity of exchange cations of our experimental site is less than 10 meq/100g. (Table 3), is an impermeable and of low-thickness substratum (Killian, 1961) (Table 3).

Floristic data

Species richness

The number of taxa counted in each station studied is 107 species and subspecies of 80 genera and 26 botanical families of vascular plants for the station (A) (Table 4). The station (B) contains a remarkable floristic diversity with 112 species belonging to 36 genera and 29 botanical families (Table 4) and station (C) includes a floristic list of 58 plant species distributed over 51 genera and 19 botanical families (Table 4). This number represents about 8% of the total Algerian flora estimated at 3139 species (Quezel, Santa 1962, 1963). Of the total flora recorded at the park level, Asteraceae, Poaceae, Fabaceae, Liliaceae, Geraniaceae, Brassiceae, Plantaginaceae and Apiaceae are the most represented with a total of more than 5 species (Table 1). These families account for nearly 40% of the park's total species richness. Our results are consistent with those of Kaabeche (1990) and Djebaili (1984). These are wealthy places and the reserve are among the most diverse ecosystems in the country, having arid and semi-arid zones. This floristic richness places the reserve into a geographical position

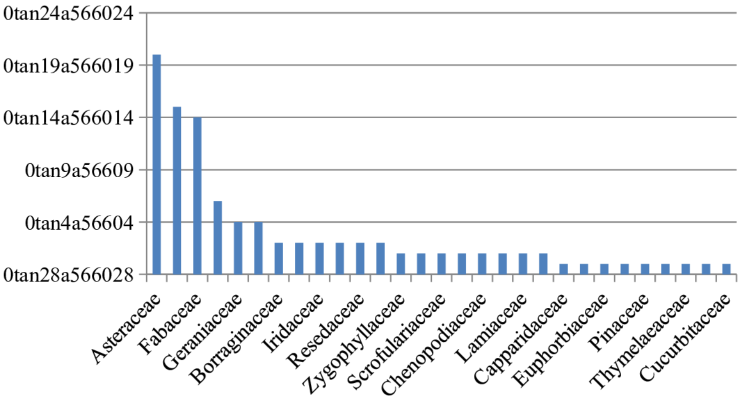


Fig. 4. Floristic diversity of station A.

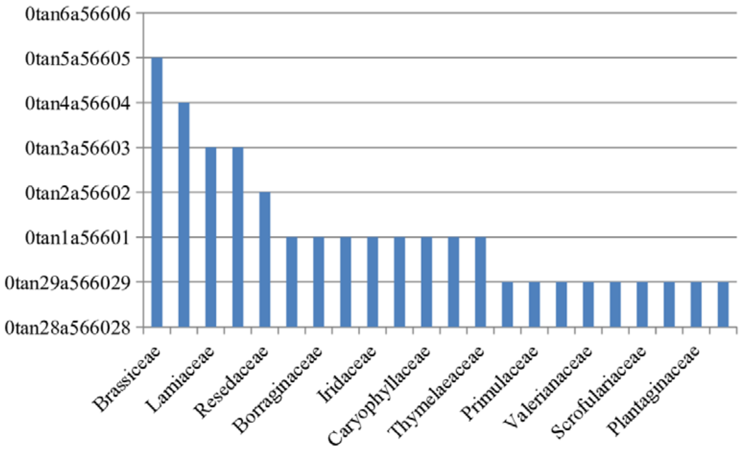


Fig. 5. Floristic diversity of the station B.

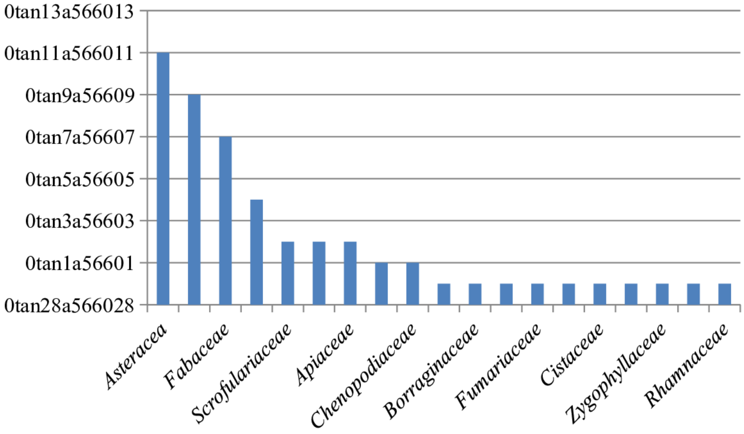


Fig. 6. Floristic diversity of station C.

that opens directly on the two atlases (Tellien and Saharan) and exposes it consequently to the North-South influences. The diversity of habitats, resulting from a climatic and edaphic heterogeneity and a relatively weak exploitation of the environment compared to other ecosystems, saw its legal status as a nature reserve.

Rare and endemic flora

At the level of endemic species, we counted (32) taxa, a rate of 11.55% compared to the total species of the El-Mergueb reserve and more than 3% compared to the estimated total endemic flora of the country to 549 species (Quezel, 1964). This rate of endemism is relatively close to that recorded in several parks in central and eastern Algeria, such as (Belezma) Park Batna with 32 species, Gouraya (Bejaia) with 27 endemic taxa (Djurdjura) with 35 species (Meribai, 2006) and (Kala) Taref with 75 species (Stevenson et al., 1988). *Alyssum scutigerum* L., *Anacyclus cyrtolepidioides* L., *Anvillea radiata* L., *Centaurea involuvrata* L., *Echium trygorrhizum* L., *Pomel linaria* L., *Laxiflora desf* L., *Scorzonera undulata* Vahl. L., *Stipa tenacissima* L., *Thymelaea microphylla* L., *Trigonella monspeliaca* L., *Xeranthemum inapertum* L. The scarcity analysis, based on the data from (Quezel, Santa, 1962, 1963), identified nearly 24 species reported as rare or very rare. The EL-Mergueb reserve is one of the important sources to more than twenty Algerian taxa listed on the Red List of the International Union for the Conservation of Nature (IUCN, 1980).

Statistical analysis

Factor analysis is defined in the literature as a method of searching for interdependent relationships between several variables to define a given phenomenon, reducing the amount of information contained in the initial variables and establishing a smaller set of dimensions

Table 4. Number of families of different species studied.

Family of plant	**Number of endemic species**	**Proportion (%)**	**Number of rare species**	**Proportion (%)**
Asteraceae	09	28.12	07	29.16
Fabaceae	04	12.5	02	8.33
Poaceae	03	9.37	02	8.33
Brassicaceae	03	9.37	02	8.33
Crassulaceae	02	6.25	02	8.33
Resedaceae	02	6.25	01	4.16
Lamiaceae	02	6.25	01	4.17
Scrofulariaceae	01	3.12	01	4.17
Apiaceae	01	3.12	01	4.17
Liliaceae	01	3.12	01	4.17
Chenopodiaceae	01	3.12	01	4.17
Boraginaceae	01	3.12	01	4.17
Thymelaeaeceae	01	3.12	01	4.17
Rubiaceae	01	3.12	01	4.17
Total	32	99.95	24	100

(called factors), aimed at minimizing the loss of information and analyzing the interdependence of the latter. The basic principle of factor analysis is to maximize the variance between the statistical units concerned and to find the central lines (components) of inertia (variation) of the point cloud (Gabor, 2010). To this end, our study uses a multivariate statistical analysis, CFA (Factorial Analysis Correspondence), which is a technical description of contingency tables and is primarily used to propose an analysis of a dataset of plant groups at each station that depended on edaphic variables. The typological results by CFA, highlighting the relationships between edaphic factors and vegetation, are illustrated in (Fig. 5). To better understand the interdependence of the parameters studied, the relationship between floristic records and their own edaphic factors is illustrated in Figures (Figs 5, 6). Floristic records, edaphic factors and plant species are represented by symbols accompanied by their codification.

The figure above shows the gradient of degradation of most groups of plants inside our experimental stations. Group B is the most ecologically stable by its floristic richness and edaphic parameters (Fig. 7); group C occupies the positive part with the edaphic environment. Moreover, the gradient of degradation inside the environment of the group A is noted to be situated the highest in the negative part, where the floristic richness is very low with a poor edaphic cover.

The spatial distribution of species is specified through a correspondence analysis (CA) performed on the stations species matrix (03 stations A, B and C taxa) (Fig. 8). The CA releases a system of factorial axes for making plane representations of all the columns and rows of the matrix. Each factor expresses some information about the analysis. The structures drawn by the clouds of variables are often significant and facilitate the demonstration of phenomena and ecological factors. The eigen values of axis 1 is 0.08 and of axis 2 is 0.075,

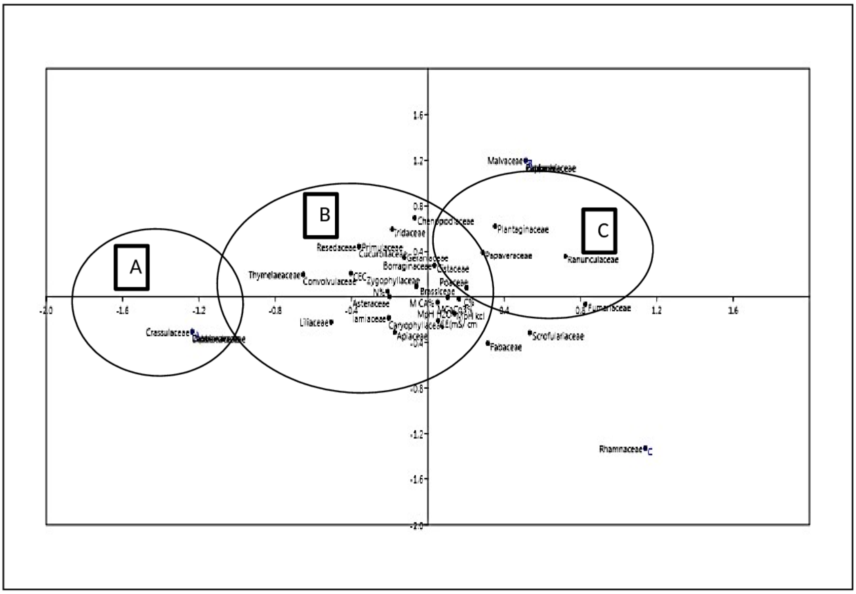


Fig. 7. Results of the factor analysis of the correspondence.

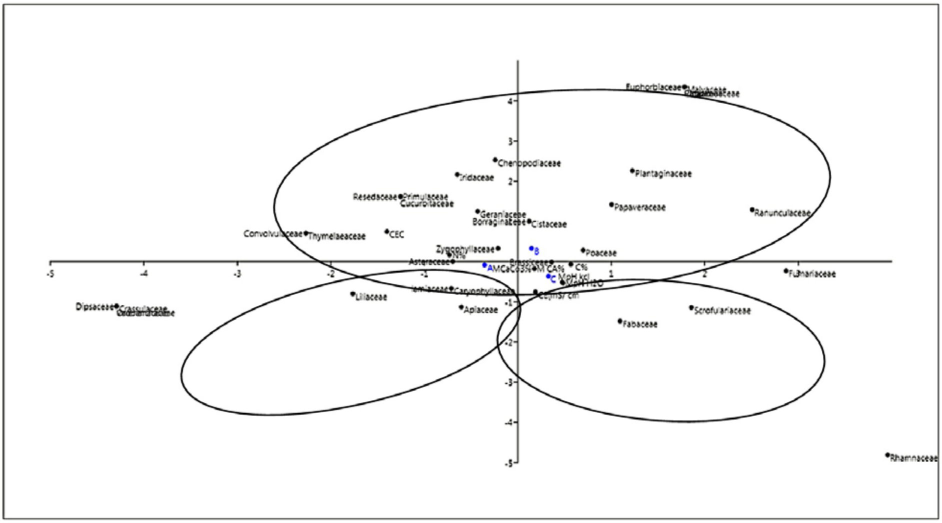


Fig. 8. Global analysis of vegetation data in relation to edaphic parameters.

so the dependence between the rows and the columns is significant. In other words, there is a significant difference between stations, edaphic parameters and plant distribution. Figure 6 shows the distribution of plants and the edaphic parameters according to the stations. The inter-class inertia is expressed on axis F1 (52.19%) and axis F2 (47.80%); the information on the two main axes (Fig. 6) showed the spatial effect on the distribution of taxa. Axis 1 appears in two stations – station B and C – in positive position and in station A in negative position. Axis 2 in positive position finds station B (Table 5).

Table 5. Signification of axes Inert and pure values.

Axes	Eigenvalue	% of total	Cumule
1	0.0827744	52.194	52.194
2	0.0758142	47.806	100

First group (Circle B): Corresponds to station (B) taxa and describes the taxa and edaphic parameters associated with this station *Alyssum scutigerum* L., *Anacyclus cyrtolepidioides* L., *Anvillea radiata* L., *Centaurea involuvrata* L., *Echium trygorrhizum* L. with: pH(H_2O) : 8.4–8.47 ; pHKcl : 7.78–7.82; $CaCo_3$% : 21–23%; T_{CEC} (meq/100g) : 5.76–9.17 (Tables 2, 3). **Second group** (Circle C): Corresponds to the station (C) taxa and describes the taxa and edaphic parameters associated with this station *Enarthrocarpus clavatus* L., *Muricaria prostrate* L., *Psychine stylosa* L., *Stipa tenacissima* L., *Thymus hirtus* L. with pH(H_2O : 8.41 –8.61; pHKcl : 7.56–7.8; $CaCo_3$% : 19–24%; T_{CEC}(meq/100g) : 3.37–5.54) (Tables 2,3). **Third group** (Circle A): Corresponds to station A taxa to describe the taxa and soil parameters related to this station *Linaria laxiflora* L., *Psychine stylosa* L., *Reichardia tingitana* L., *Reseda alba* L., *Salsola vermiculata* L., *Schismus barbatus subsp calycinus* L., with pH (H_2O : 8.42–8.49; pHKcl : 7.78–7.82; $CaCo_3$% : 22% –23; T_{CEC}(meq/100g) : 5.14–6.75) (Tables 2, 3).

Discussion

In general, it appears from these results after phyto-edaphic analysis that the contribution and the specific frequency of the developed stations are different from those of the undeveloped stations. The surface of the soil appears as a succession of dyes separated by very moderate bulges that dominate most of the experimental area (Boudjadja et al., 2010). Soils often encrusted are of small thickness. In some areas, the limestone crust that covers them is continuous and does not allow any plant rooting (Boudjadja et al., 2010). Gamoun et al. (2011) state that the sandy soil is more productive than the limestone soil in addition to creating hardness, limestone layers can reduce water penetration and reduce plant access to water. Janišová (2005) showed that the influence of dominant environmental factors is reflected sensitively in changes of vegetation composition. Therefore, differences in plant composition and proportion of species present in each family in all studied station are due to the differences in environmental factors. It should be noted, however, that both management and land use have a profound effect on vegetation. Assessment of the enclosure of some steppes area in northern Algeria by Boucherit et al. (2017) showed that grazing management could improve plant cover and diversity during the time. So in all studied stations, future management can improve vegetation conditions from cover and plant biodiversity. Because of the differentiation of altitude and micro-macro topography and climate, the vegetation ecosystems of the El-Mergueb region, there are distinguished different types of plant communities, and diverse species compositions and landscape complexes. Rahmonov et al. (2013) have also obtained similar results for the vegetation separation of lowland and mountainous areas in Tajikistan. The typology of soils acquired by the combined action of three factors: the climate, vegetation, and anthropogenic pressure, the partial pedological cover accentuates the impoverishment of his superficial horizons of soils. With these conditions, soils within our experimental area may be only poorly developed (Gaouar, 2005; Soil survey manual, 1937). As grazing animals trample and remove live vegetation and litter mass, production is reduced, especially in arid ecosystems (Gamoun et al., 2010). When production becomes low, leaves litter and organic carbon reduction are inevitable and this can affect soil properties. Greater organic matter can support more species, as seen on the station B. Moreover, the strong presence of different species of botanical families such as Asteraceae, Poaceae and Fabaceae indicate the strong grazing inside our study stations. According to Hirche et al. (2011), the main factor of degradation is overgrazing in arid and semi-arid region of Algeria and our study area is suffering from this phenomenon. So some resistance plant family like Asteraceae are present in the plant composition of station B and C in the first place. Previous studies established the same conclusion about the degradation of the steppe in Algeria (Hirche et al., 2011; Aidoud et al., 2006; Slimani et al., 2010) and Morocco (Benbrahim et al., 2004).

Our results proved the notion: The nature reserve of El-Mergueb is a unique steppe ecosystem and contains a specific biotope not existing throughout the Maghreb, which gives it an international dimension.

Conclusion

The study area is a region geographically located within a steppe ecosystem whose plant and soil diversity is remarkable, presenting a key indicator for this bioclimatic stage. Land degradation, anthropogenic pressure and climatic constraints have accentuated the decrease of the floristic richness and denuded the soil cover, of which this last wealth persists and resists. According to our interpretation, the presence of certain plant species has a climate-related soil requirement whose adaptation is mandatory. The area is rich in endemic species but under pasture and climatic constraints, the vulnerability of species has become a major problem as per their numbers in each station. The low quantity of organic matter in the soils and other mineral elements under a total limestone level dominates majority of the study stations and along with decrease in rainfall, these form the major causes of disruption of growth and development of plants. Also, inside this type of ecosystem, plants can resist to develop with a minimum of climatic conditions and edaphic cover. The protected area of El-Mergueb characterized by a great vulnerability is where erosion dominates; it is accentuated by the negative action of arid climate. Our research proposes to protect the endemic species inside our site, which make the biodiversity inside this protected area unique, particularly in the homogeneous and heterogeneous facies to be representative of green space between two different climatic floors of the ecosystem.

Acknowledgements

I acknowledge assistant editor Eva Orbánová for her valuable contributions to this manuscript. I appreciate the criticism and suggestions that highly improved the quality of this manuscript provided by anonymous reviewers. I thank colleagues who offered crucial help in the field and the lab, at laboratory of Ecology and Environment (LREE), Abdurrahman Mira University of Bejaia, Algeria; International Desert Research Center (IDRC), University of Tehran, Iran; Department of Sciences of Nature and Life, Faculty of Sciences, Mohamed Boudiaf University of M'sila, Algeria; Institute of Ecology and Environmental Sciences (IEES) of Paris, France.

References

AFNOR (1998). French standard NF ISO 11259, X31-001, Soil quality. Simplified description of the soil. France.

Aïdoud, A., Le Floc'h, É. & Le Houérou H.N. (2006). The arid steppe rangelands of Northern Africa. *Science et Changements Planétaires/Sécheresse*, 17(1), 19–30.

Aubert, H. & Pinta M. (1978). Trace elements in soils. *Soil Sci.*, 125(5), 334.

Baize, D. & Jabiol B. (2011). *Guide for soil description*. Paris: INRA.

Benbrahim, K.F., Ismaili, M., Benbrahim, S.F. & Tribak A. (2004). Problèmes de dégradation de l'environnement par la désertification et la déforestation: impact du phénomène au Maroc. *Science et Changements Planétaires/ Sécheresse*, 15(4), 307–320.

Bonneau, M. & Souchier B. (Eds.) (1979). *Pédologie 2.* Constituants et propriétés du sol. Paris: Masson. DOI: 10.7202/1000421ar.

Boucherit, H., Benabdeli, K. & Benaradj A. (2017). Biological recovery the steppe of Hammada scoparia after enclosure in the region of Naama (Algeria). *Ekológia (Bratislava)*, 36(1), 52–59. DOI: 10.1515/eko-2017-0005.

Boudjadja, A., Kaddour, B.H. & Pauc H. (2010). Increasing the value of surface water storage for protected wildlife, particularly the emblematic Cuvier's Gazelle, in Mergueb Reserve (M'sila, Algeria). *Revue d·Ecologie - la Terree et la Vie*, 65(3), 243–253. http://hdl.handle.net/2042/55842

Bounar, R. (2014). *Etude des potentialités biologiques, cartographie et aménagement de la chaîne des Babors dans la démarche du développement durable*. http://www.univ-setif.dz/Tdoctorat

Bremner, J.M. (1960). Determination of nitrogen in soil by the Kjeldahl method. *J. Agric. Sci.*, 55, 11–33. DOI: 10.1017/S0021859600021572.

Breuste, J., Schnellinger, J., Qureshi, S. & Faggi A. (2013). Urban ecosystem services on the local level: urban green spaces as providers. *Ekológia (Bratislava)*, 32(3), 290–304. DOI: 10.2478/eko-2013-0026.

Capot-Rey, R. (1955). Les sols d'Algérie d'après Mr J.H. Durand. *Annales de Géographie*, 64(341), 56–58. www.persee.fr/issue/geo_0003-4010_1955_num_64_341

Chapman, H.D. & Pratt P.F. (1961). *Methods of analysis for soils, plants and waters*. Los Angeles: University of California.

Czudek, R. (2001). *Some key issues in the context of the future of protected areas and wildlife management in Africa*. Contribution to the forestry outlook study for Africa. Rome: Food and Agriculture Organization of the United Nations.

Djebaili, S. (1984). *Steppes algériennes, phytosociologie et écologie*. Alger: OPU.

Djili, K. (2000). *Contribution à la connaissance des sols du Nord d'Algérie. Création d'une banque de données informatisées et utilisation d'un système d'information géographique pour la spatialisation et la valorisation des données pédologiques*. Thèse d'état en sciences agronomiques. El- Harrach: INA.

Gabor, M.R. (2010).Descriptive methods of data analysis for marketing data – theoretical and practical considerations management & marketing challenges for knowledge society. *Management and Marketing*, 5(3), 119–134. http://www.managementmarketing.ro/pdf/articole/198.pdf

Galet, M.P. (1951). Le dosage du calcaire actif et l'appréciation du pouvoir chlorosant des sols. *Bull. d'Off. Inst. du Vin*, 24, 37–42.

Gamoun, M., Tarhouni, M., Ouled Belgacem, A., Hanchi, B. & Neffati M. (2010). Effects of grazing and trampling on primary production and soil surface in North African rangelands. *Ekológia (Bratislava)*, 29(2), 219–226. DOI: 10.4149/ekol_2010_02_219.

Gamoun, M. (2013). Vegetation change in variable rangeland environments: the relative contribution of drought and soil type in arid rangelands. *Ekológia (Bratislava)*, 32(1), 148–157. DOI: 10.2478/eko-2013-0013.

Gaouar, A. (2005). *Management plan of the site of El- Mergueb, hydrogeology, project DGF.NUD-ALG/G 35*. In the national program of classification of the protected areas in Algeria.

Hilbert, D.W., Hughes, L., Johnson, J., Lough, J.M., Low, T., Pearson, R.G., Sutherst, R.W. & Whittaker S. (2007). *Biodiversity conservation research in a changing climate*. Australia: CSIRO.

Hirche, A., Salamani, M., Abdellaoui, A., Benhouhou, S. & Valderrama J.M. (2011). Landscape changes of desertification in arid areas: the case of south-west Algeria. *Environ. Monit. Asses.*, 179(1–4), 403–420. DOI: 10.1007/s10661-010-1744-5.

Horváth, B., Opara-Nadi, O. & Beese F. (2005). A simple method for measuring the carbonate content of soils. *Soil Sci. Soc. Am. J.*, 69, 1066–1068. DOI: 10.2136/sssaj2004.0010.

International Union for Conservation of Nature (IUCN) (1980). *Lists of rare and endangered plants of the Mediterranean basin states*. Athens: IUCN.

Jackson, M.L. (1962). Interlayering of expansible layer silicates in soils by chemical weathering. *Clays and Clay Minerals*, 11(1), 29–46. DOI: 10.1346/CCMN.1962.0110104.

Janišová, M. (2005). Vegetation-environment relationships in dry calcareous grassland. *Ekológia (Bratislava)*, 24(1), 25–44.

Kaabeche, M. (1990). *Vegetation groupings of the Boussada region (Algeria), synthesis test on the Maghreb steppe vegetation*. PhD thesis, Ferhat Abess University of Setif, Algeria. www.univ-setif.dz/Tdoctorat/

Kaabeche, M. (2003). *Conservation de la Biodiversité et gestion durable des ressources naturelles: étude sur la réhabilitation de la flore locale au niveau de la réserve « El - Mergueb » (Wilaya de M'sila, Algerie)*. Algerie: Ministere de L'Agriculture et du Developpement Rural Direction Generale des Forets.

Kaabeche, M. (2005). *Management plan of the site of El- Mergueb, hydrogeology, project DGF.NUD-ALG/G 35*. In the national program of classification of the protected areas in Algeria.

Killian, L. (1961). *Natural and artificial improvement of a pasture in an Algerian Reserve "Le Mergueb"*. Northern Africa.

Le Houérou, H.N. (2005a). *Atlas of climatic diagrams for the isoclimatic Mediterranean regions*. Montpellier: Copymania.

Le Houérou, H.N. (2005b). *The isoclimatic mediterranean biomes. Bioclimatology, diversity and phytogeography*. Montpellier: Copymania.

Leadley, P., Pereira, H.M., Alkemade, R., Fernandez-Manjarrés, J.F., Proença, V., Scharlemann, J.P.W. & Walpole M.J. (2010). *Biodiversity scenarios: projections of 21st century change in biodiversity and associated ecosystem services*. Montreal: Secretariat of the Convention on Biological Diversity.

Maire, R. (1926). *Phytogeographical map of Algeria and Tunisia*. Gouverneur General Algérie

Maire, R. (1928). Origin from the flora of the mountains of Northern Africa. *Memo. Soc. Biogéogr*, 2, 187–194.

Maire, R. (1952). *Flora of North Africa (Morocco, Algeria, Tunisia, Tripolitania, Cyrenaica and the Sahara)*. Vol. 16. Paris: Lechevalier.

Maire, R. (1952,1987). *Flora of North Africa (Morocco, Algeria, Tunisia, Tripolitania, Cyrenaica and Sahara)*. Vol. 1 to 16. Paris: Lechevalier.

Merdas, S., Menad, A., Mostephaoui, T. & Sakaa B. (2017). Plant community structure and diversity under grazing gradient in arid Mediterranean steppe of Algeria. *Journal of Materials and Environmental Sciences*, 8, 4329–4338. DOI : 10.26872/jmes.2017.8.12.456.

Meribai, Y. (2006). *Study of vegetation in Djurjura National Park phytosociology and proposal development*. Magister thesis, Ecology: Phytosociology. Algeria: National Institute of Agronomy.

Pendleton, R.L. & Nickerson D. (1951). Soil colors and special Munsell soil color charts. *Soil Sci.*, 71(1), 35–44.

Petard, J. (1993). *The analyses methods*. Vol. 1. Soils analyses. Paris: Orstom.

Pouget, M. (1980). *Les relations sol-végétation dans les steppes Sud-Algéroises*. Paris: Orstom. http://www.documentation.ird.fr/hor/fdi:00344

Quezel, P. & Santa S. (1962). *New flora of Algeria and southerly desert regions*. Paris: CNRS.

Quezel, P. & Santa S. (1963). *Nouvelle flore de l. Algérie et des régions désertiques méridionales*. CNRS: Paris.

Quezel, P. (1964). L'endémisme dans la flore d'Algérie. *Compte Rendu Sommaire des Séances de la Société de Biogéographie*, 361, 137–149.

Rahmonov, O., Majgier, L., Andrejczuk, W., Banaszek, J., Karkosz, D., Parusel, T. & Szymczyk A. (2013). Landscape diversity and biodiversity of Fann Mountains (Tajikistan). *Ekológia (Bratislava)*, 32(4), 388–395. DOI: 10.2478/eko-2013-0037.

Robinson, G.W. (1922). A new method for the mechanical analysis of soil and other dispersions. *J. Agric. Sci.,* 12, 306–321. DOI: 1017/S0021859600005360.

Slimani, H., Aidoud, A. & Roze F. (2010). 30 Years of protection and monitoring of a steppic rangeland undergoing desertification. *J. Arid Environ.*, 74(6), 685–691. DOI : 10.1016/j.jaridenv.2009.10.015.

Soil survey manual (1937). *Bureau of plant industry soils, and agriculture engineering*. Washington: United State Department of Agriculture.

Sparks, D.L. (1996). *Methods of soil analysis.* Part 3: Chemical methods. Madison: Soil Science Society of America, American Society of Agronomy.

Stephen, M.P. (2015).*The impact of ecosystem services knowledge on decisions*. Partial Fulfillment of the Requirements for the Degree of Doctor of Philosophy Specializing in Natural Resources.

Stevenson, A.C., Skinner, J., Hollis, G.E. & Smart M. (1988). The El Kala National Park and Environs, Algeria: An Ecological Evaluation. *Environ. Conserv.*, 15, 335–348. DOI: 10.1017/S0376892900029830.

Tandon, H.L.S. (1993). *Methods of analysis of soils, plants, waters, and fertilizers*. New Delhi: Fertiliser Development and Consultation Organisation.

United Nations Framework Convention on Climate Change (UNFCCC) (2007). Climate change in developing countries: impacts, vulnerabilities and adaptation.

Walkley, A. (1947).An critical examination of a rapid method for determining organic carbon in soils. Effect of variations in digestion conditions and of inorganic soil constituents. *Soil Sci.*, (63), 251–264.

Wetang·ula, G.N. (2009). Biodiversity, conservation and loss: international legal mechanism for management. In *Short Course IV on Exploration for Geothermal Resources, organized by UNU-GTP, Ken Gen and GDC* (pp. 1–8). Lake Naivasha, Kenya, November 1–22, 2009.

Ekológia (Bratislava) Vol. 38, No. 4, p. 353–366, 2019
DOI:10.2478/eko-2019-0026

SIMULATION OF DECLINE OF NORWAY SPRUCE (*Picea abies* L. Karst.) FORESTS IN GORGAN MOUNTAINS (UKRAINIAN CARPATHIANS): CASE STUDY USING FORKOME MODEL

TARAS PARPAN[1], IHOR KOZAK[2,3], YURIY SHPARYK[1], MYROSLAVA MYLENKA[3], IVAN BALANIUK[4]

[1]Ukrainian Research Institute for Mountain Forestry, Ivano-Frankivsk, Ukraine; e-mail: tarasparpan@gmail.com
[2]Department of Landscape Ecology, John Paul II Catholic University in Lublin, Konstantynow 1H, 20-708 Lublin, Poland; e-mail: modeliho@kul.lublin.pl
[3]Department of Biology and Ecology, Vasyl Stefanyk Precarpathian National University, Shevchenko str., 57, 76010, Ivano-Frankivsk, Ukraine; e-mail: kozakihor58@gmail.com, mulenka.m@gmail.com
[4]Department of Accounting and Auditing, Vasyl Stefanyk Precarpathian National University, Shevchenko str., 57, 76010, Ivano-Frankivsk, Ukraine; e-mail: ifbalaniuk@gmail.com

Abstract

Parpan T., Kozak I., Shparyk Y., Mylenka M., Balaniuk I.: Simulation of decline of Norway spruce (*Picea abies* L. Karst.) forests in Gorgan Mountains (Ukrainian Carpathians): case study using FORKOME model. Ekológia (Bratislava), Vol. 38, No. 4, p. 353–366, 2019.

The FORKOME model used in the article contains elements of forest and ecological approaches and was specially developed for the conditions of the Gorgans. The modeling was performed based on the single simulation results and statistically averaged forecast of 200 simulations ("Monte Carlo") in order to show the tendency of changes and their correspondence with single simulations. The forecast of the forest dynamics was conducted at the 100th anniversary period with "control" and "warm-dry" scenarios. It has been revealed that the rapid decrease in biomass in the first decade was caused by Norway spruce decline. It was revealed that in the control scenario, the most active biomass growth during the 100th year forecast was shown by Silver fir (*Abies alba* Mill.) and beech (*Fagus sylvatica* L.). The fir reacted relatively sensitively to the warming and decrease in rainfall. The relationships between tree species and the influence of biomass of Norway spruce tree on the biomass of the whole forest tree stand were analyzed.

Key words: modeling, biomass, number of trees, forecast.

Introduction

For the past two centuries, the extensive planting of Norway spruce (*Picea abies* L. Karst.) has been a characteristic method of forest management for Central Europe. Norway spruce monocultures take up considerable areas in the Ukrainian Carpathians (Holubets, 1978). Nowadays, under changing climatic conditions, modified forest ecosystems have been rapidly losing their vitality and resistance against destructive abiotic and biotic impacts (Spiecker et al., 2004; Shparyk, 2014, 2019). At the present time, a decline in Norway spruce forests is made visible on the area of 19,300 ha of forest (3% of whole Norway spruce in the Ukrainian Carpathians) with a wood volume of 5.8 million m^3 (Parpan et al., 2014).

Norway spruce mortality was evenly recorded in different regions across all slope expositions (Mandre, 1992). The infestation intensity increased slightly with Norway spruce representation in the stands and was higher in stands older than 80 years (Kozak, Menshutkin, 2001; Grodzki et al., 2014), and defoliation of trees influences the increment and destruction of tree stand structure.

At that time, Norway spruce mortality was recorded in Ukraine as a national problem (Parpan et al., 2014). Studies on the type and scope of diseases and pests and on the methods of sanitary measures of Norway spruce forests in the area of Carpathian were conducted (Turis, Shanta, 2013). Causes of Norway spruce forests drying in the Precarpathians are considered; groups of factors that make this negative process are identified and characterized (Debryniuk, 2011). Conversion of an even-aged secondary Norway spruce into uneven-aged mixed stands in the Carpathian Mountains, as in whole Central Europe, is internationally thought as a main challenge of the recent mountain forest management (Lavnyy, Schnitzler, 2014). The simulation of conversion strategies for a 62-year even-aged secondary Norway spruce site located in Transcarpathian region of the Ukrainian Carpathians was conducted using the growth simulator SIBYLA (Fabrika, 2005).

One of the reasons of decline of Norway spruce forest is global warming. For the past 100 years (1906–2005), the average global temperature has increased by approximately 0.74 °C (IPCC, 2007). During the same period, the average temperature in Europe increased even more by 0.95 °C. According to the forecasts formulated by the Intergovernmental Panel on Climate Change (IPCC), a further significant increase in the average annual temperature in Europe is expected by the end of the 21st century. It is estimated that for the next 100 years, temperatures in Europe will increase from 2.0 to 6.3 °C (IPCC, 2007). The sum of precipitation and the frequency of extreme weather events are also expected to increase (Lindner et al., 2010). At that time, Norway spruce stands in Ukrainian Carpathians grows in the warm temperature belt (Parpan et al., 2014).

Forests are particularly sensitive to climate change, because the long life of trees does not allow for quick adaptation to environmental changes (Linder et al., 2010). Owing to climate change, there are a number of factors affecting forest ecosystems that can function independently or in combination. These factors are temperature changes and rainfall. The occurrence of meteorological phenomena, although they are natural processes occurring in the atmosphere, negatively affects forests, causing very dangerous consequences (Durlo, 2011). The changes taking place in the environment have occurred so quickly that the Norway spruce has not managed to adapt to them.

The forecasting of dynamics of decline Norway spruce forest stands requires the use of special tools, that is, properly designed computer models. Therefore, an attempt was made to use the FORKOME model to carry out prognostic simulations of possible changes in factors affecting the dieback of Norway spruce. Autocorrelation and cross-correlation function were added to the FORKOME model and can also be considered as coefficients of mutual influence (Jørgensen, 1994). Interesting is the analysis of the relationships (correlations) between decline Norway spruce and others tree species in the forest tree stands.

Taking these reflections into consideration, the main objective of this article is to perform the forecasts of biomass and number of trees in decline Norway spruce stands in Gorgans (Ukraine) in control (actual temperature and precipitation) and in warm-dry scenario (with increase 200˚ degree-days temperature and decrease 200 mm precipitation). Additional aim was to conduct cross-correlation analysis between biomass of tree species in the decline forest stand using the FORKOME computer model.

Material and methods

Specificity of decline of Norway spruce stand is analyzed in research plot situated in the Ukrainian Gorgans (Ludvykivka Department, Vygoda State Forestry Enterprise, Ivano-Frankivsk Regional Forestry and Hunting Administration). This plot was located in compartment 14, subcompartment 42 (N: 48047′30,5″, E 23043′53,7″) on the northern slope of the mountain at an inclination of 18º and altitude of 800 m a.s.l. in wet mesotrophic Silver fir – Common beech – Norway spruce forest type. The brown soil covering the Carpathian flysch is a characteristic for plot, which was covered by Norway spruce, fir (*Abies alba* Mill.), sycamore (*Acer pseudoplatanus* L.), and birch (*Betula pendula* Roth.) trees. The average diameters (DBH1,3), heights (H), and age are presented in Table 1.

Table 1. Basic characteristics of the all trees in research plot.

Species	**DBH (cm)**			**H (m)**			**Age (years)**			**N**
	mean	**min**	**max**	**mean**	**min**	**max**	**mean**	**min**	**max**	
Picea abies	31.1	7.1	72.4	27.5	4	45	77	20	120	146
Abies alba	25.3	6.35	80.1	20.5	5	44	56	20	119	78
Acer pseudoplatanus	34.5	15.9	57.7	27.1	9	41.5	84	26	119	15
Betula pendula	29	21.9	36.6	26	20.7	32.5	72	33	112	6

The dimension of the research plot was 50×60 m. Such size is often used in gap models (Bugmann, 2001). After conducting 200 test simulations for each study plot, it was found that the area observed corresponded to the district with the area of 200×300 = 60.0 ha. The statistical analysis of the obtained data (the Shapiro–Wilk test performed using STATISTICA 13.1.) showed that the DBH of the examined trees had normal distribution with marked right-sided asymmetry and (the regression analysis) allow to create prediction equation for *P. abies* biomass changes in control and in warm-dry climate conditions.

The trees observed on the study plots were numbered and their DBH, height, crown radius, age, as well as X and Y coordinates were recorded. Hemispherical photographs (in 9 sites on the study plot) were taken as required for verifying the leaf area index (LAI) calculations performed by FORKOME. A digital SLR camera Canon EOS 5D with Sigma 8mm f/3.5 DG EX FISH EYE with 180° viewing angle was used. Gap Light Analyzer software (Frazer et al., 2000) was used for hemispherical analyses.

All the data collected from the study plots were entered into the model FORKOME (Kozak et al., 2012), and simulations of stand development for a period of 100 years were carried out (such a long simulation period was needed for stabilizing relationships among Norway spruce, fir, and beech which appeared in the process of forecasting on the research plots).

The model FORKOME belongs to the group of gap models that simulate the long-term dynamics (succession) of forest complexes, taking directly into account the influence of the most important environmental factors, such as temperature, precipitation, light, and soil nutrient content, on the main processes taking place in the stands: regeneration, growth, and decline in individual trees. The design and structure of FORKOME were described in detail in earlier publications, and FORKOME model was verified for Ukraine, Poland, and Sweden (Kozak, Menshutkin, 2002; Kozak et al., 2007, 2012). In particular, a large number of modifications and new solutions to the model were introduced in block of natural disasters.

FORKOME represents a patch-model family that simulates the forest association succession, allowing single tree research, and herein, it is divided into blocks. The following two kinds of analysis are possible with FORKOME: (1) statistical analysis with mean value and standard variation and (2) sensitivity analysis in a series of autocorrelation and cross-correlation function calculations. This model sets site, species, climate, and felling parameters, and result-saving and additional analysis by other computer methodology and programmers is also possible. The Monte Carlo statistical method allows simulation of up to 200 variants for each scenario. The model delivers the average amount and average biomass of trees with the standard variation in each year. Autocorrelation and cross-correlation functions are included to improve the sensitivity of the analysis of the forest ecosystems. The number of trees and their biomass are important parameters in these calculations, and available charts present the relationships between these parameters and temperature and humidity for each species and for the entire association.

In the FORKOME model, the "growth bloc" simulates the actual annual tree growth on the studied area. Although each tree has its own genetically coded method of growth, its growth process is also influenced by its environment. The FORKOME model's trees are also described by species-specific growth function, with the main parameters of DBH, height (H), and age, together with the external conditions of the individual patch area. This approach simplifies the growth simulation and allows growth-function activation and implementation in current conditions. The basic simulation consists of the tree diameter calculation, where the annual diameter increase ranges from the minimal value of 0 to the maximum value for each species under ideal conditions (Kozak et al., 2012).

The influence of external conditions is factored into the annual tree volume increase process. The actual tree increase $\delta(D^2H)_{real}$ results from the optimal increase $\delta(D^2H)_{opt}$ and tree growth inhibiting conditions *f1, f2,….. fj*, where the value of each tree-growth-inhibiting factor ranges from 0 to 1.

Light availability is the most important external factor inhibiting tree growth, and in FORKOME, this is calculated with consideration of light radiation loss. This loss is caused by the total shading by the leaf area of higher trees.

The available light function describes the amount of light available for specific tree leaves. Trees are divided into the following 3 types based on their light tolerance index: sun tolerant, medium, and shade tolerant.

The tree-growth-inhibiting light index is called the light reaction function, and this is calculated in two different ways depending on tree-light tolerance index. This model's thermal conditions are described by the addition of annual effective temperatures above 5^{o}, and the temperature index inhibiting the tree growth can be calculated (Botkin, 1993).

The FORKOME model also considers leaf transpiration, and this depends not only on the meteorological conditions but also on the tree species, as in other patch models. There also exist relationships between tree species and ground water level and between tree growth speed and the availability of ground water implemented in the model structure. The bloc is created by the basic water balance equation (Kozak et al., 2012).

A further tree-growth-inhibiting index is the SITE INDEX that describes the ratio of steam-occupied area to the maximum available area (Botkin, 1993).

A tree can perish in the following two ways in the FORKOME model: (1) randomly or (2) if it does not reach minimum diameter size. The model asserts that if the tree does not increase its diameter every year for 10 years, then there is only a 1% chance that the tree will survive that period. The MORTAL statistical probability for annual tree death is 0.386.

The FORKOME model is able to confirm the tree's minimal increase. If the minimum value is not exceeded, then it is assigned its random probability from 0 to 1, and when that value is greater than 0.386, the tree is removed.

Autocorrelation and cross-correlation functions are included to improve the sensitivity analysis of the forest ecosystems. The number of trees and their biomass are important parameters in these calculations, and the available charts present the relationships between these parameters and temperature and humidity for each species and for the entire association.

Autocorrelation function was calculated based on the equation:

$$AC(X,\tau) = \frac{\sum_{t=0}^{t_{\max}-\tau} \left\{ \left[X(t) - \overline{X} \right] * \left[X(t-\tau) - \overline{X} \right] \right\}}{\left| \sum_{t=0}^{t_{\max}} \left[X(t) - \overline{X} \right]^2 \right|} * \frac{t_{\max}}{t_{\max} - \tau}$$

AC(X, t) is the autocorrelation coefficient for X;
X is the parameter for autocorrelation analysis;

X is the arithmetic mean for X;
t is the time in the model;
τ is the time interval.

The current version of the FORKOME model has been verified using the data from 2010 to 2014 and up to 2016. After running the model on the data for 2010, the forecast to 2014 and to 2016 was realized. The model represented 90% real-field data from 2014 and 87% from 2016 year and showed good verification (prognosticity) for the stochastic models. After that, the prognosis was made for the next 94 years (total time of simulations run equal 100 years).

The special "DRYING" block has been added to the FORKOME model. While working on the prediction of drying of Norway spruce forests, the tree can take three basic states: healthy, drying, and overthrowing. Healthy and drying trees have the "dry" parameter that determines the percentage of drying in the crown. A healthy tree with a dry value of 0% has no signs of dryness. In the prediction process, when a tree is classified as dry, its state changes ("is_dry: = true"). From this moment, the tree participates in the process of decline. This block is responsible for controlling the tree's drying.

In addition, the current version of the FORKOME model has improved the appearance of trees and their textures in 3D visualization, which depicts the stages of shrinkage and decay of tree branches. Throughout the simulation process, changes have been tracked for each tree from its occurrence and annual growth to drying and falling out of the composition of the tree stand.

Results

General view of the test area at the beginning, in 2010 (Fig. 1a), and after 4 and 6 years of simulation, in 2014 (Fig. 1b) and in 2016 (Fig. 1c), respectively, with FORKOME model interface (Fig. 1d) was presented. At the beginning (Fig. 1a), there were 53 dry Norway spruce, but at the 6 years of model simulation time, its amount increased to 94 individuals and 20 trees were fall.

Within the analyzed plot, initially (in 2014), there were 245 trees. The trees belonging to Norway spruce (93 alive and 53 dry), fir (72 alive and 6 dry), sycamore (14 alive and 1 dry), and birch (6 alive) were recorded (Table 2). The FORKOME model has been verified using the 2014 and 2016 data and represents 90% field data concerning biomass in 2014 and 87% in 2016.

As a result of simulation, the proportion of Norway spruce trees considerably decreased. The sum of basal area value for Norway spruce was finally smaller (1.9 m^2/ha^{-1}) than for fir (26.2 m^2/ha^{-1}). In terms of numbers, Norway spruce trees almost decreased compared to those in the beginning (146 Norway spruce trees) of simulation and reached 42. The number of fir trees increased to 125 individuals (Table 3).

Regarding the sum of basal area value, a distinct domination of fir is visible, supplemented with young and many beech trees (Table 3). For this forest plot, the Monte Carlo simulation runs were conducted in control conditions and in warm-dry climate changes scenario.

Table 2. Initial characteristics of alive and dry trees in the FORKOME model.

Species	**Tree number**	**Basal area (m^2/ha^{-1})**	**Age (years)**
Picea abies	146	26.5	77
Abies alba	78	19.1	56
Acer pseudoplatanus	15	4.9	82
Betula pendula	6	1.4	72

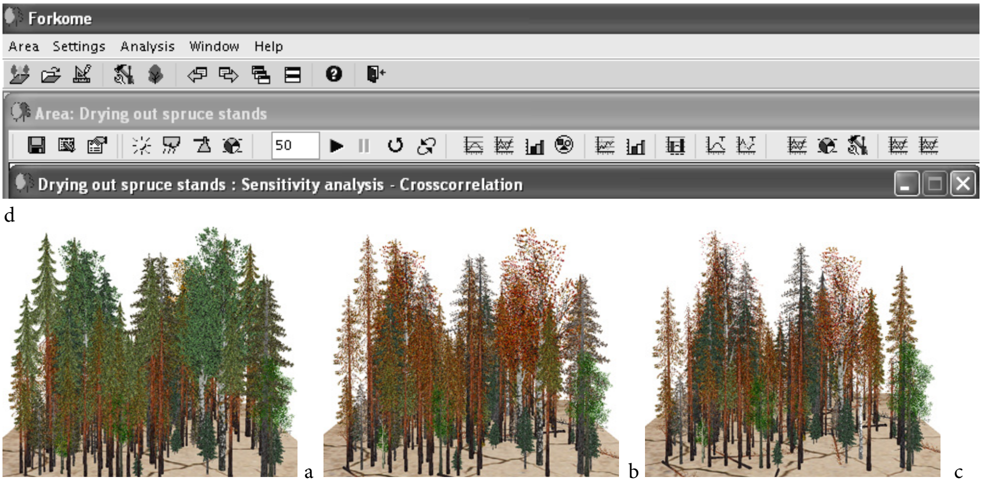


Fig. 1. View of the test plot at the beginning, in 2010 (a); at the 4th year of simulation, in 2014 (b); and at the 6th year of simulation, in 2016 (c), of the forecast with FORKOME model interface (d).

Table 3. Parameters of tree stand in the 100th year of simulation by the FORKOME model.

Species	**Tree number**	**Basal area (m^2/ha^{-1})**	**Age (years)**
Picea abies	42	1.9	61
Abies alba	125	26.2	84
Fagus silvatica	75	3.1	56
Acer pseudoplatanus	5	3.2	168

In Monte Carlo realization (200 simulations) for the control conditions, the FORKOME model predicted that the biomass of Norway spruce trees decreased from 153.57 ± 5.6 t/h in the first years of model time to 32.0 ± 3.2 t/ha in 10 years (Fig. 2). After that the model predicts that Norway spruce would be at the same level up to 70 years and decrease to 11.3 ± 2.6 t/h in 100 years. After the decrease in fir biomass in 10 years from 116.36 ± 4.7 t/h in the first year of model time to 59.5 ± 2.2 t/ha in 10 years, the model predicts that in 100 years, the fir biomass increased to 152.0 ± 4.2 t/ha. The FORKOME model predicts an increase in of beech biomass (to 38.5 ± 3.6 t/h) in the simulation time. The biomass of sycamore will be at the level of 22.8 ± 1.7 t/h in 100 years and that of birch decreased to zero in 10 years of simulation times (Fig. 2).

In the control, the model predicted a decrease (from 93 at the beginning of simulation to 26 in 10 years) in the tree numbers of alive Norway spruce trees. After that the Norway spruce numbers increased to 42 individuals in 100 years. The number of fir trees decreased from 72 at the beginning to 44 in 10 years and increased to 119 individuals in 100 years. The numbers of beech trees increased to 84 individuals in 100 years (Fig. 3).

The prediction of simulation of dead trees showed intensive dying of Norway spruce and also fir trees, which was visible up to 10 years of simulation time.

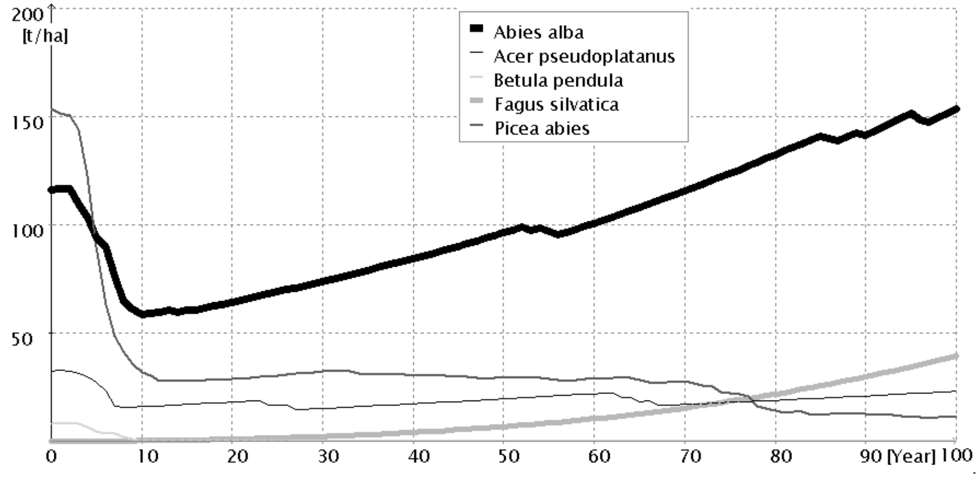


Fig. 2. Prediction of the biomass in control scenario.

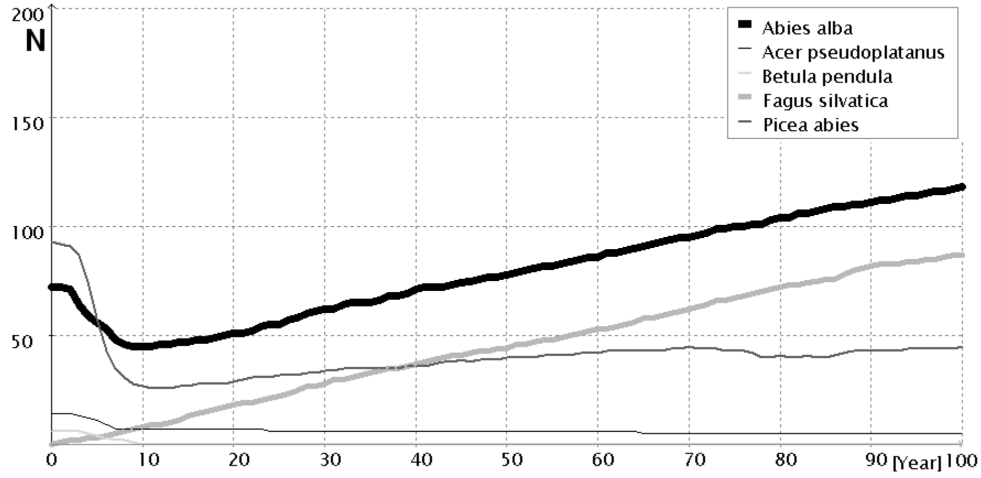


Fig. 3. Prediction of the number of trees in control scenario.

In warm-dry scenario, the model predicted that the biomass of Norway spruce trees decreased from 153.57 ± 5.6 t/h in the first years of model time to 0.62 ± 0.09 t/h in 16 years. After decreasing, the Norway spruce biomass will be at the minimum level of 0.5± 0.1 t/h in 100 years. The model also predicts the decrease in fir (from 116.4± 4.4 t/h in the first year of model time to 17.3± 1.2 t/h in 10 years) and its increase in biomass to 27.1 ±2.4 t/h in 100 years. In the warm-dry climate changes scenario, the model predicted an increase in beech biomass (from zero at the beginning of simulation to 36.14 ± 2.6 t/h biomass in 100 years).

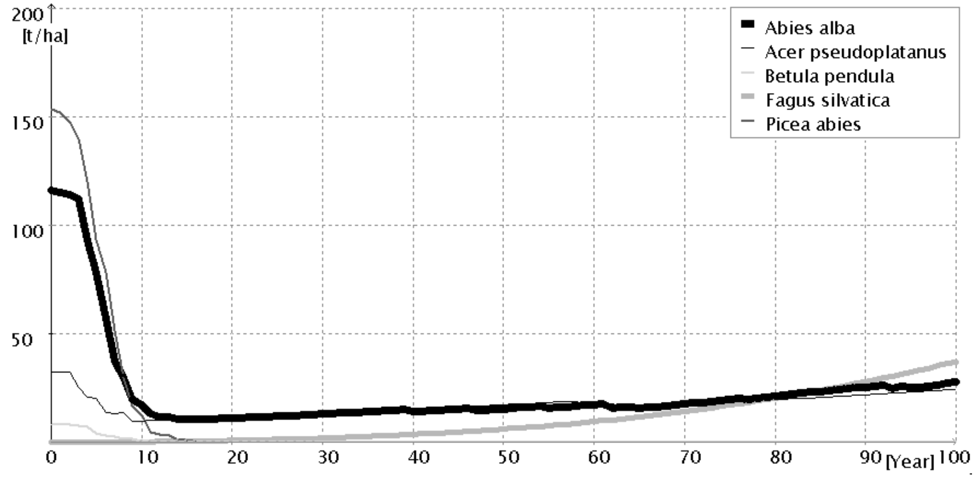


Fig. 4. Prediction of the biomass in warm-dry climate changes scenario.

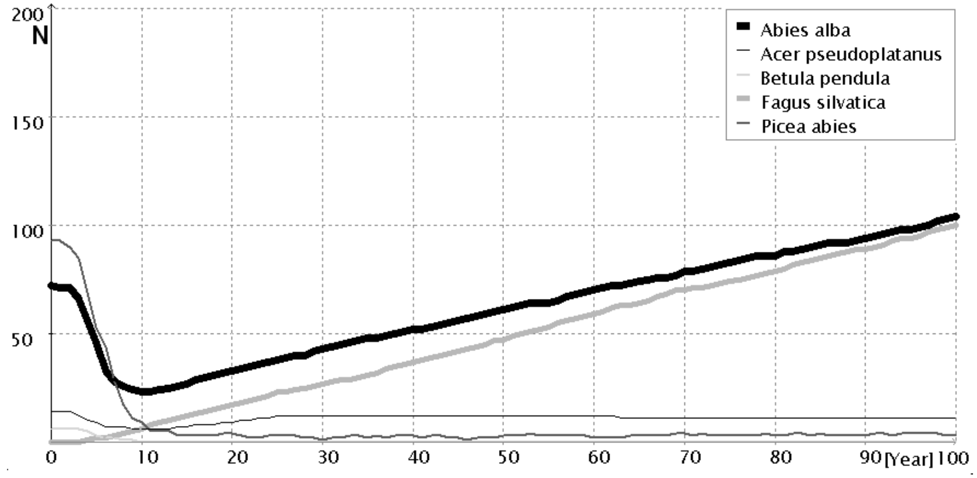


Fig. 5. Prediction of the amount of trees in warm-dry climate changes scenario.

The biomass of sycamore oscillates from 32.36 ± 2.1 t/h at the beginning to 24.12± 1.7 in the 100 year. The biomass of birch decreased to zero in 10 year of simulation times (Fig. 4).

In warm-dry scenario, the tree numbers of Norway spruce decreased from 93 individuals in the first year to 6 individuals in 10 years and to 5 in 100 years of simulation time. After decreasing in the first 10 years, the amount of fir individuals increased to 105 in 100 years. The tree numbers of beech increased to 99 individuals in 100 year. On the plot, the number of sycamore individuals changes to 10 individuals in 100 years (Fig. 5).

As a result of simulation in warm-dry scenario, the proportion of Norway spruce trees decreased more compared to the control conditions. The sum of basal area value for Norway spruce was finally smaller (0.008 m^2 ha^{-1}) than that for fir (4.4 m^2 ha^{-1}). In terms of numbers, Norway spruce trees almost decreased compared to those at the beginning of simulation and to those in the control conditions.

The autocorrelation of Norway spruce biomass and Norway spruce tree amount in FORKOME model simulations in control conditions and warm-dry scenario was analyzed. The autocorrelation of Norway spruce concerning the biomass of trees is more positive than that concerning the amount of trees in control conditions (Fig. 6a). Generally, the autocorrelation of Norway spruce concerning the biomass and the amount of trees was not more changed in warm-dry scenario (Fig. 6b) compared to the control conditions. The autocorrelations of Norway spruce in control conditions and in warm-dry scenario changed a little ($\tau = 3$ for the biomass and $\tau = 2$ for the amount of trees in control conditions; and $\tau = 3$ for the biomass and for the amount of trees in the warm-dry conditions on the level more than +0.75). The autocorrelations for birch and sycamore were similar to the autocorrelation of Norway spruce.

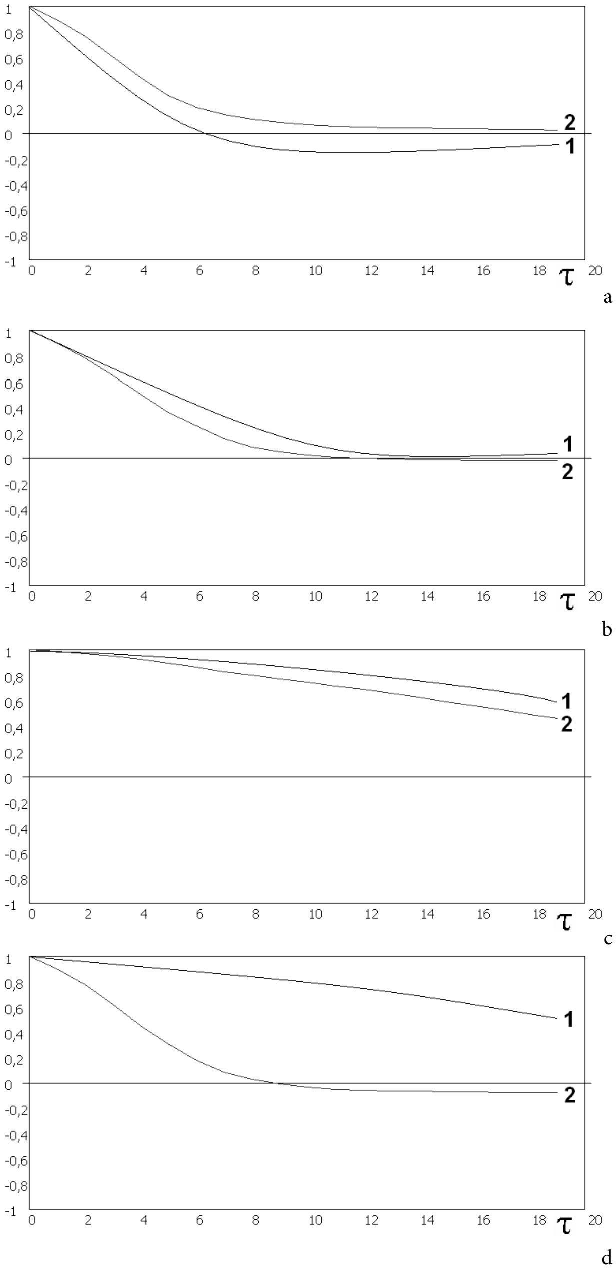


Fig. 6. Autocorrelation of amount of trees (1) and biomass (2) for Norway spruce in control conditions (a) and in warm-dry scenario (b) and for fir in control conditions (c) and in warm-dry scenario (d).

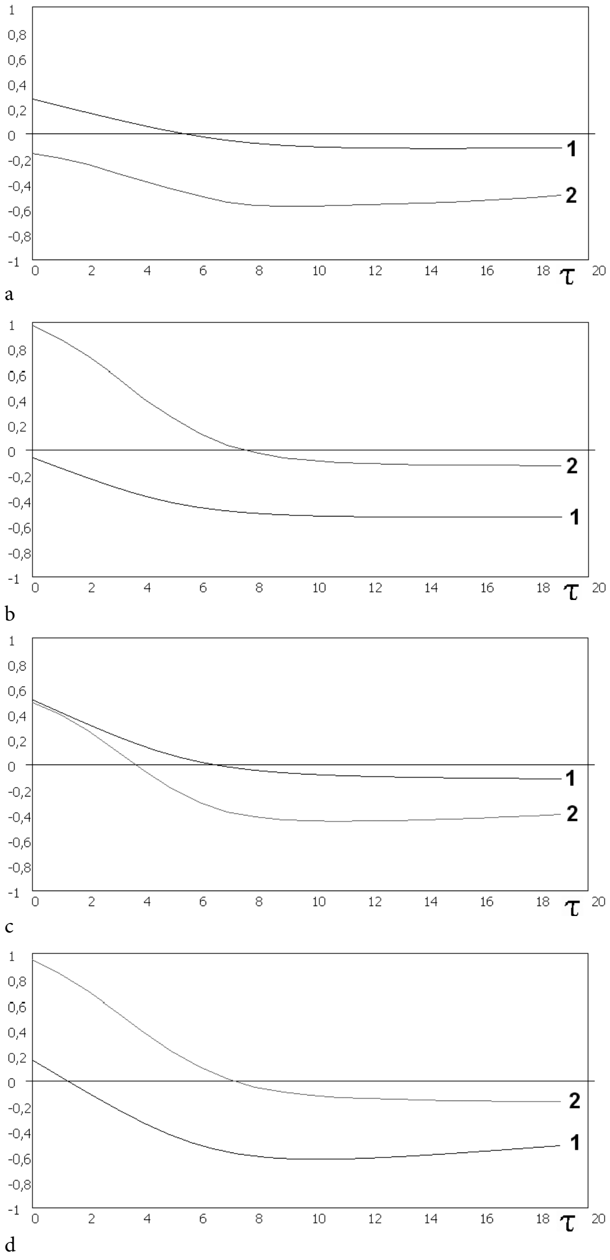


Fig. 7. Cross-correlation between Norway spruce and fir amount of trees (1) and biomass (2) in control conditions (a) and in warm-dry scenario (b) and between Norway spruce and total stand community in control conditions (c) and in warm-dry scenario (d).

For fir in control conditions (Fig. 6c), the autocorrelations of biomass and amount of trees were strong (τ = 10 for biomass and τ = 14 for amount of trees on the level more than +0.75). Only the autocorrelations of biomass of fir trees were less (τ = 3 on the level more than +0.75) in warm-dry climate changes scenario (Fig. 6d).

The cross-correlation between Norway spruce and fir, Norway spruce and beech, Norway spruce and sycamore, and Norway spruce and birch was analyzed using the FORKOME model. In the control scenario, the cross-correlation between Norway spruce and fir biomass of trees was negative (Fig. 7a). It can mean that the relation between Norway spruce and fir were competitive. In warm-dry climate changes scenarios, the cross-correlation between Norway spruce and fir biomass of trees changed from strong positive (τ = 2 on the level more than +0.75) to weak negative (Fig. 7b) and the cross-correlation changed from +0.9 to −0.1.

The cross-correlation is similar between the Norway spruce biomass in comparison with total tree stand biomass and also between tree amounts of Norway spruce in comparison with total tree amount on the stand community. This confirms a small influence of

Norway spruce on all stand community in control conditions (Fig. 7c) and in warm-dry scenario (Fig. 7d).

The results of species biomass simulations during 100 years have been processed in statistical program. The multiple regression analysis for the biomass of Norway spruce, fir, beech, birch, and sycamore was conducted using Statistica 13.1 program.

After statistical processing of the results of simulation data, a mathematical equation has been obtained, which characterizes the connections that appear between the biomass of selected species. Here we only remark that the collected data have been analyzed for the dependent indicators in order to exclude cases that could violate the established regression equation. In the final result, after eliminating nonsignificant samples, the regression equation was determined. All indicators have been checked for the probability test $p \leq 0.05$ in order to exclude those that showed a lack of statistical accuracy.

The statistically significant results with $p \leq 0.05$ have been obtained for all analyzed parameters, and zero values of p have been obtained in the results of regression analysis for the dependent variable Norway spruce biomass.

The analysis of the standardized residuals for the dependent variable showed a lack of values of greater than ±3 sigma, indicating a lack of significant data deviations.

For the analyzed control conditions, the prediction equation for Norway spruce biomass was given as follows: *Picea abies* = -4,89563624765-1,34717188084* "*Fagus silvatica*"+0,246197902646*"*Abies alba*"+10,654894521* "*Betula pendula*"+1,11558300704*"*Acer pseudoplatanus*".

In warm-dry climate scenario, the prediction equation for Norway spruce was given as follows: *Picea abies* = -5.92447769934-0.373233236774* "*Fagus silvatica*"+1.35232848217*"*Abies alba*"+2.88879103231* "*Betula pendula*"-0.759108581137*"*Acer pseudoplatanus*".

These equations can be used for the calculation of Norway spruce biomass in situation where the FORKOME model will not be available and to compare with biomass prediction changes in the FORKOME Monte Carlo simulation run.

Discussion

The article presents the perspectives for the application of FORKOME computer model in prognosis of the impact of natural disturbances on Norway spruce forests. The FORKOME model, in its latest version, includes a recently created and constantly developed block of forest drying. As a result, it is possible to simulate potential changes caused by decline, and thus it predicted the impact of this disturbance agent on forest conditions and stand regeneration by conducting simulations on the issues concerned.

The model FORKOME uses both general mathematical relationships functioning within a forest ecosystem and empirical ones, characteristic for tree stands within analyzed plots. That is, it combines ecological models as well as empirical ones (Brzeziecki, 1999; Bugmann, 2001). On the basis of the results generated using the FORKOME model, the hypothesis that in 10 years of simulation time run, the Norway spruce may not be the dominant species within the analyzed area regarding the number of trees as well as their biomass seems reasonable. This indirectly confirms the Norway spruce in Gorgans area is in damage of forest associations there. The results obtained also suggested that even smallest climate changes, especially in air temperature and rainfalls, can cause decrease in the Norway spruce biomass and amount of trees. It is consistent with the data presented in literature. For example, Debrynuk (2011) reported that in connection with further climate change, the consequence of which is the gradual increase in the average annual temperature of the air and decrease in rainfall, increasing the length of the growing season, the drying of Norway spruce in the forest region will continue. In beech and fir-beech forest types that are intensified by natural regeneration beech forest, the best conditions for growth will be for sycamore and Silver fir. Similar results were presented in our FORKOME model simulations. It means that we can introduce promising species into existed forest stand composition, which can provide the highest stability and productivity of phytocoenosis, as proposed by Debrynuk (2011).

Unfortunately, in the countries of Western Europe, there were only small remains of natural forests. In this regard, virgin forests in the Ukrainian Carpathians and their research are extremely important for the close-to-nature forestry management throughout Europe. The goal of the approaching nature of forestry is the forestry method by which various goals can be achieved. Stable, close to natural, forests can in the long run meet different expectations.

In the Ukrainian Carpathians, computer simulation using the FORKOME model is perspective. The FORKOME model predicted decreases in Norway spruce biomass and amount of trees and increase in Silver fir and Common beech. This once again emphasizes the relevance of introducing sustainable forest management methods that increase the sustainability of forests.

Parpan et al. (2014) also confirmed that secondary Norway spruce forests' decline is very intensive in the Ukrainian Carpathians now on the area of 19,300 ha of forests with a wood volume of nearly 5.8 million m^3. The main reason is Norway spruce stand stability deterioration because of global warming, Norway spruce planting in the not typical for Norway spruce forest sites, environmental pollution, the massive spread of the Norway spruce diseases and pests, and the Norway spruce stands' damages by windfalls and snow. Excessive distribution of the Norway spruce decline areas and Norway spruce forests stability deterioration together with Norway spruce forests increment decreasing require emergency management activities based on the forest typology, the forest monitoring results, and the economic conditions analysis.

This will help to slow up the rapid degradation of Norway spruce stands, what reduces their protective, water-reserve, ecological and recreational functions, and what leads to a wood resources loss and a fire risk increases in these forest stands.

Conclusion

A sharp decrease in the biomass of decline Norway spruce stand was showed in the first decade for 100 years simulation in control conditions and warm-dry scenario. The biomass of Norway spruce trees decreased from 153.57 t/h in the first years of model time to 11.3 t/h in 100 years in control conditions and to 0.5 t/h in warm-dry scenario. The Norway spruce was especially sensitive to changing climatic conditions. Its share and biomass were decreased in control conditions and in warm-dry scenarios. The fir, beech, and sycamore were characterized by a much better adaptation to warmer and drier conditions for the next decades.

Improvement in the FORKOME model provides grounds for arguing about the possibility and expediency of working out forecast predictions for the development of dwindling Norway spruce stands. Scientific research is based on the need for earlier warnings about threats to forest ecosystems in order to minimize the costs of their restoration. Thus, the urgency and necessity of investigations of the current sanitary condition of Norway spruce trees and their forecasting with the use of computer simulation software are emphasized.

Improving the health condition of Norway spruce forests is possible by gradually changing the structure and species composition. The correct solution will be to introduce species that are more resistant, such as fir, and deciduous trees, such as beech, that were presented in the article and in the FORKOME model simulations.

The condition for proper management of mountain Norway spruce forests is to understand their natural disturbances of dynamics at the stand level. Such information is the basis on which sustainable forest practices can be developed.

References

Botkin, D.B. (1993). *Forest dynamics: an ecological model*. Oxford, New York: Oxford University Press.

Brzeziecki, B. (1999). *Ecological model of the forest stand. Principles of construction, parameterization, examples of application (in Polish)*. Ph.D. Thesis, Warszawa.

Bugmann, H. (2001). A review of forest gap models. *Clim. Change*, 51, 259–305. DOI: 10.1023/A:1012525626267.

Debrynuk, Yu M. (2011). Die-back of the fir forests: causes and consequences (in Ukrainian). *Науковий вісник НЛТУ України*, 21(16), 32–38.

Durło, G. (2011). The possibility of adaptation of spruce forests in Beskid Śląski Mts. to changing climatic conditions (in Polish). *Prace i Studia Geograficzne*, 47, 227–236.

Fabrika, M. (2005). *Simulator biodynamiky lesa SIBYLA, koncepcia, konštrukcia a programove riešenie*. Ph.D. Thesis, Technicka univerzita vo Zvolen.

Frazer, G.W., Canham, C.D. & Lertzman K.P. (2000). Gap Light Analyzer (GLA), Version 2.0: Image processing software to analyze true-colour, hemispherical canopy photographs. *Bull. Ecol. Soc. Am.*, 81, 191–197. DOI: 10.1890/0012-9623(2000)081[0190:TT]2.0.CO;2.

IPCC (2007). Climate Change 2007: Synthesis report. In Core Writing Team, R.K. Pachauri & A. Reisinger (Eds.), *Contribution of working groups I, II and III to the fourth assessment report of the intergovernmental panel on climate change*. Geneva: IPCC.

Grodzki, W., Starzyk, J. & Kosibowicz M. (2014). Impact of selected stand characteristics on the occurrence of the bark beetle *Ips typographus* (L.) in the Beskid Żywiecki Mountains (in Polish). *Leśne Prace Badawcze*, 75(2), 159–169. DOI: 10.2478/frp-2014-0015.

Holubets, M. (1978). *Spruce forests of Ukrainian Carpathians (in Ukrainian)*. Nauka.

Jørgensen, S.E. (1994). *Fundamentals of ecological modelling*. Amsterdam, London, New York, Tokyo: Elsevier.

Kozak, I., Chłódek, D., Zawadzki, A., Kozak, H. & Potaczała G. (2007). Simulation of spruce forest reconstruction in the Polish Bieszczady using the FORKOME model (in Polish). *Leśne Prace Badawcze*, 2, 7–26.

Kozak, I. & Menshutkin V. (2001). Investigation of spruce forest dynamics in the Bieszczady Mountains using a computer modelling. *Ekológia (Bratislava)*, 20(4), 371–378.

Kozak, I. & Menshutkin V. (2002). Predictions of spruce forest dynamics in the Polish Bieszczady and Ukrainian Bieskidy using the computer modelling. *Baltic Forestry*, 8(1), 28–34.

Kozak, I., Mikusiński, G, Stępień, A., Kozak, H. & Frąk R. (2012). Modelling forest dynamics in a nature reserve: a case study from south-central Sweden. *J. For. Sci.*, 58(10), 436–445. DOI: 10.17221/28/2012-JFS.

Lavnyy, V. & Schnitzler G. (2014). Conversion felling in the secondary spruce stands experiences in Germany (in Ukrainian). *Proceedings of the Forestry Academy of Sciences of Ukrain, Collection of Research Papers*, 12, 73–78.

Lindner, M., Maroschek, M., Netherer, S., Kremer, A., Barbati, A., Garcia-Gonzalo, J., Seidl, R., Sylvain Delzon, S., Corona, P., Marja Kolstro, M., Lexer, M. & Marchetti M. (2010). Climate change impacts, adaptive capacity, and vulnerability of European forest ecosystems. *For. Ecol. Manag.*, 259, 698–709. DOI: 10.1016/j.foreco.2009.09.023.

Mandre, M., Annuka, E. & Tuulmets L. (1992). Response reactions of conifers to alkaline dust pollution. Changes in the pigment system. *Proc. Est. Acad. Sci. Biol. Ecol.*, 4, 156–173.

Parpan, V.I., Shparyk, Y.S., Slobodyan, P., Parpan, T., Korshov, V., Brodovich, R., Krynyckyi, G., Debrenyuk, Y., Kramarets, V. & Cheban I. (2014). Forest management peculiarities in secondary Norway spruce (*Picea abies* (L.) H. Karst.) stands of the Ukrainian Carpathian (in Ukrainian). *Proceedings of the Forestry Academy of Sciences of Ukraine, Collection of Research Papers*, 12, 178–185.

Shparyk, Y.S. (2014). Form diversity and the health condition of Norway spruce (*Picea abies* (L.) Karst.) in the main forest types of the Ukrainian Carpathians (in Ukrainian). *Forestry and Forest Melioration*, 125, 87–96.

Shparyk, Y.S. (2019). Ecological results of Norway spruce forests' decline in main forest types of the Ukrainian Carpathians (in Ukrainian). *Proceedings of the Forestry Academy of Sciences of Ukraine*, 18, 145–153. DOI: 10.15421/411915.

Spiecker, H., Hansen, J., Klimo, E., Skovsgaard, J.P., Sterba, H. & von Teuffel K. (2004). Norway spruce conversion - options and consequences. *European Forest Institute Research Report*, 18, 25–62.

Turis, E.V. & Shanta O.I. (2013). Status and perspectives of the content of derived conifer forest in the territory of National Park "Enchanted land" (in Ukrainian). *Науковий вісник Ужгородського університету: Серія: Географія, Землеустрій, Природокористування*, 2, 186–190.

Ekológia (Bratislava) Vol. 38, No. 4, p. 367–381, 2019
DOI:10.2478/eko-2019-0027

FITTING COMPETING MODELS OF THE POPULATION ABUNDANCE DISTRIBUTION: LAND SNAILS FROM NIKOPOL MANGANESE ORE BASIN TECHNOSOLS

OLGA KUNAKH, DARYA KOVALENKO

Oles Honchar Dnipro National University, pr. Gagarina, 72, 49010 Dnipro, Ukraine; e-mail: kunah_olga@ukr.net
Bogdan Khmelnitsky Melitopol State Pedagogical University, 20, Hetmanska Street, 72318, Melitopol Zaporizhzhya, Ukraine; e-mail: dashuliakovalenko30@gmail.com

Abstract

Kunakh O., Kovalenko D.: Fitting competing models of the population abundance distribution: land snails from Nikopol manganese ore basin technosols. Ekológia (Bratislava), Vol. 38, No. 4, p. 367–381, 2019.

This paper examines the temporal dynamics of terrestrial mollusks of the Nikopol manganese ore basin technosols. The research was carried out at the Research Centre of the Dnipro Agrarian and Economic University in Pokrov (Ukraine). Sampling was carried out in 2012–2014 on four variants of artificial soil: formed on red-brown clays, on loess-like loams, on gray-green clays, and on humus-rich layer. The distribution of the number of individuals in a mollusk population was described by broken stick, Motomura, log-normal, Zipf, and Zipf-Mandelbrot models. It was shown that the series of models that best describe mollusk abundance distribution is specific for a particular species and technosols type and generally is invariant over time.

Key words: land snails, technosols, distribution, population.

Introduction

Habitat is characterized by the availability of a certain range of resources and ecological conditions for a given species, which creates conditions to occupy, survive and reproduce in a given territory (Hall et al., 1997). Finding appropriate statistical distributions for the analysis of animal populations at different hierarchy levels is an important subject of investigation in ecology (Zipkin et al., 2014). The detection of ecological properties, which makes an area fitting for the existence of a species, is important to study the habitat selection (Calenge, Basille, 2008). The investigations of mollusk communities from different biotopes, which differ in vegetation cover, soil type, and moisture level, were devoted to study the habitat selection by land snails (Millar, Waite, 1999; Martin, Sommer, 2004; Müller et al., 2005; Weaver et al., 2006). The importance of soil factors in the spatial distribution, abundance and diversity of mollusk communities has been revealed (Nekola, Smith, 1999; Juřičková et al., 2008; Szybiak et al., 2009). The soil properties, such as the calcium content, pH, the soil texture (Ondina et al., 2004, Juřičková et al., 2008), soil moisture (Nekola, 2003) and the content of exchangeable cations and aluminum (Ondina et al., 1998), have a significant impact on the mollusks population. The best predictor of land snail species composition is the content of carbonate calcium in topsoil (Juřičková et al., 2008). The litter moisture was shown as being the leading factor affecting the microspatial distribution of the terrestrial mollusks (Książkiewicz-Parulska, Ablett, 2017). The phytoindication data application is useful for the evaluation of land snails' ecological properties (Horsák et al., 2007; Dvořáková, Horsák, 2012; Yorkina et al., 2018, 2019). The habitat preferences of the land snail *Vertigo geyeri* in Poland and Slovakia were effectively explained by Ellenberg phytoindication scales (Schenková et al., 2012). Human disturbance at the habitat and landscape scales benefited *Cepaea nemoralis* through the creation of suitable habitats (Rosin et al., 2017).

The open cast mining is one of the most intense disturbances on terrestrial ecosystems. Gastropods are sensitive to habitat disturbance (Kappes et al., 2009). A considerable diversity of the land snails community of a partially reclaimed abandoned coal mine site was revealed (Arruda, 2014). Snails were shown as being highly sensitive to microclimatic fluctuations and structural changes of the man-made structures (Kappes et al., 2012). The habitat transformation on reclaimed land is able to induce a rapid and considerable evolutionary differentiation of the *C. nemoralis* (Schilthuizen, 2013).

The abundance estimates are used to determine the population status for the assessment of the impact on environmental factors and to monitor population trends (Zipkin et al., 2014). Different statistical distributions were used for population abundance fitting such as the Poisson distribution (Caraco, 1980; Kunakh et al., 2018), the negative binomial distribution (Beauchamp, 2011; Wood, 1985), the geometric distribution (Okubo, 1986), and the power law distribution (Bonabeau et al., 1999). The mechanisms behind the patterning of the intra-population abundance distribution of the land snail *Vallonia pulchella* were summarized. For the analysis of the snail population abundance, the following models were proposed: broken sticks model, niche preemption model, log-normal model, Zipf model, and Zipf-Mandelbrot models (Kunakh et al., 2018).

The aim of our work is to examine the temporal dynamics of terrestrial mollusks of the Nikopol manganese ore basin technosols, as well as to find regularities in nature of statistical laws, which explain the distribution of the snail populations.

Material and methods

The research was carried out at the Research Centre of the Dnipro Agrarian and Economic University in Pokrov (Ukraine). The Research Centre (47°38'55.24"N.L., 34°08'33.30"E.L.) for the study of optimal regimes of agricultural reclamation was established in 1968–1970 (Fig. 1). Sampling was carried out in 2012–2014 on four variants of artificial soil (technozems): formed on red-brown clays (RedBrown), on loess-like loams (Loess), on gray-green clays (GrayGreen) and on humus-rich layer (Pedozem) (Fig. 2). According to WRB 2007 (IUSS Working group WRB, 2007), the examined soil belong to the RSG Technosols (Yorkina et al., 2018). From 1995 to 2003, a long-term legume-cereal agrophytocenosis grew on the site, after which the process of naturalization of the vegetation began. In each variant of the technosols, the test polygon was laid, within which sampling was done consisting of 7 transects of 15 samples each. Test points form a regular grid with a mesh size of 3 m. Thus, the total test point number is 105. The test point size is 0.5 × 0.5 m. A quadrat was fixed on the soil surface prior to taking the snail individuals. The snail individuals were manually collected from the soil samples and plants. Each site within the polygons was examined three times a year: in the spring (early May), summer (end of June) and autumn (late September – early October).

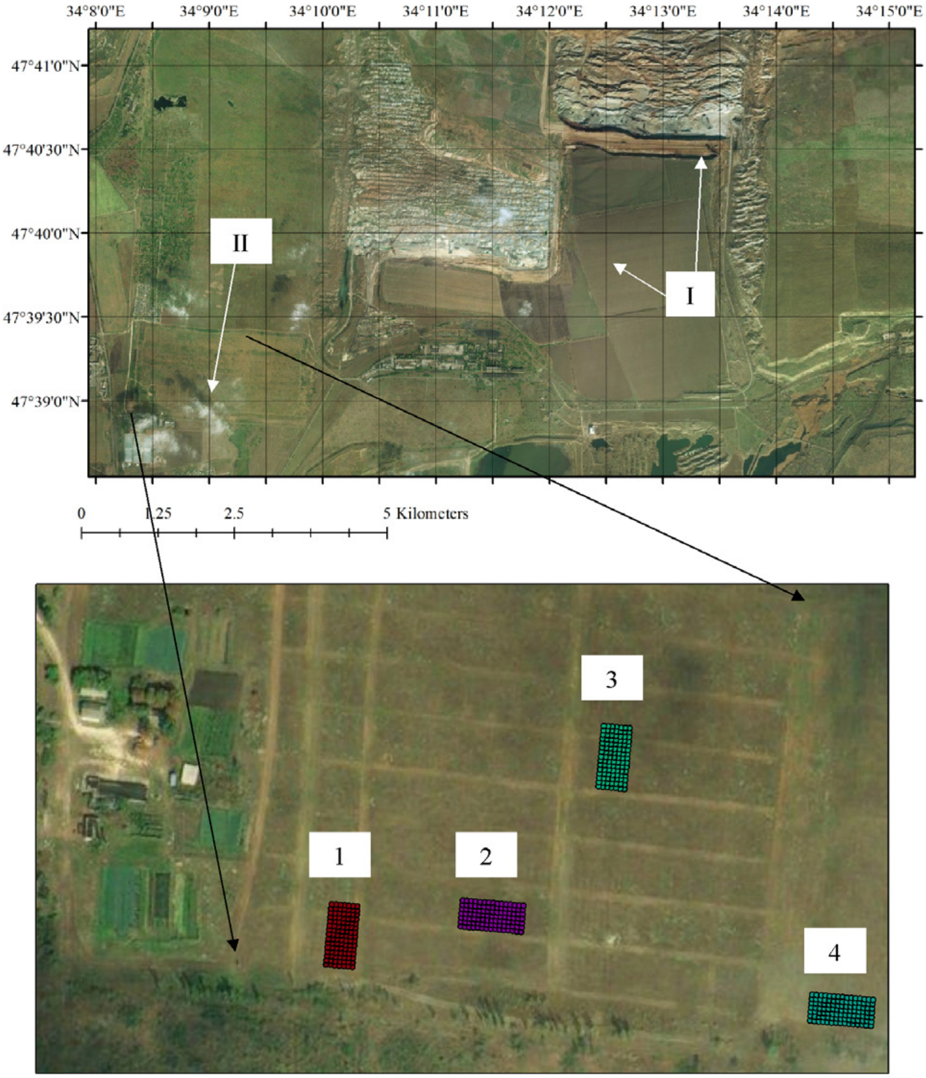


Fig. 1. Location of the test polygons within the Nikopol manganese ore basin.
Notes: I. – Zaporizhia pit mining of manganese ore; II. – an experimental field with different technosols variants: 1 – technosols on loess-like loams; 2 – technosols on red-brown clays; 3 – technosols on gray-green clays; 4 – technosols on humus-rich layer (pedozem).

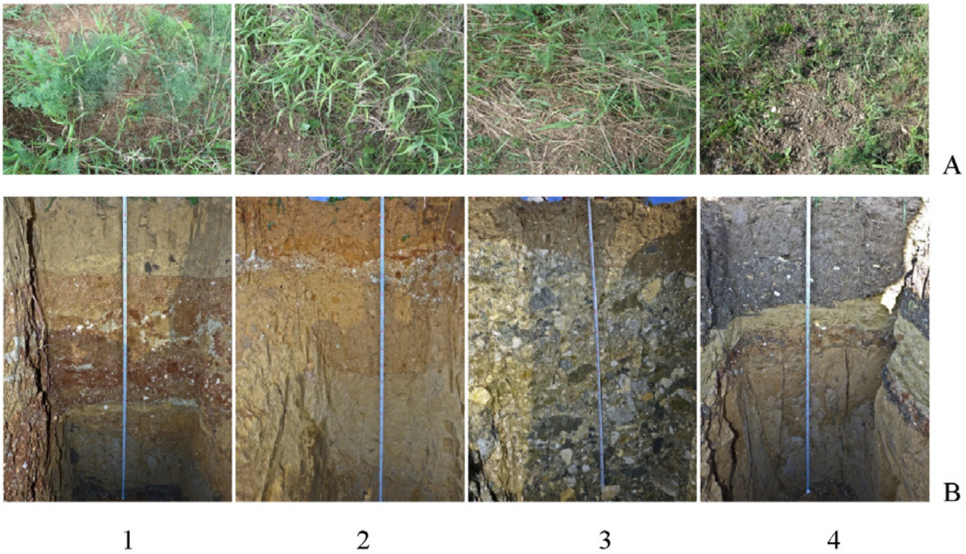


Fig. 2. Plant cover (A) and soil section of the test polygons.
Notes: 1 – technosols on loess-like loams; 2 – technosols on red-brown clays; 3 – technosols on gray-green clays; 4 – technosols on humus-rich layer (pedozem).

The distribution of the number of individuals in a mollusk population may also be described by the dependence "rank-abundance", which are often used in community ecology (Whittaker, 1965). In this case, the individual sample plot sites with mollusks may be regarded as equivalents of individual species in the community (Kunakh et al., 2018). For the analysis in this case, the following model can be used:

broken stick model (MacArthur, 1957):

$$\hat{a}_r = \frac{N}{S}\sum_{k=r}^{S}\frac{1}{k}\ ; \qquad (1)$$

Motomura model (the Whittaker niche preemption model) (Motomura, 1932):

$$\hat{a}_r = N\alpha(1-\alpha)^{r-1}; \qquad (2)$$

log-normal model (Preston, 1948, 1962):

$$\hat{a}_r = \exp[\log(\eta)+\log(\sigma)\Phi]\ ; \qquad (3)$$

Zipf model (Zipf, 1949):

$$\hat{a}_r = N\hat{p}_1 r^{\gamma}\ ; \qquad (4)$$

Zipf-Mandelbrot model (Mandelbrot, 1983):

$$\hat{a}_r = cN\ (r+\beta)^{\gamma}\ ; \qquad (5)$$

where $\hat{a}_r$ is the expected abundance of species of rank *r*; *S* is the number of species; *N* is the number of individuals; Φ is a standard normal distribution function; $\hat{p}_1$ is the estimated proportion of the most abundant species; α, μ, σ, γ, β and c are the parameters in each model.

The degree of adequacy of the model was evaluated using Akaike›s information criterion (AIC) and the Bayesian information criterion (BIC). The best model has the lowest AIC and BIC. Statistical calculations were performed with the help of the Statistica 7.0 program and the project for statistical computations R (www.r-project.org) using *vegan* (Oksanen et al., 2018).

Results and discussion

The four species of terrestrial mollusks were found within the investigated technosols: *Brephulopsis cylindrica* (Menke, 1828), *Monacha cartusiana* (O.F. Muller, 1774), *Chondrula tridens* (O.F. Muller, 1774), *Helix lucorum* Linnaeus, 1758. The abundance of the *Brephulopsis cylindrica* in the studied habitats varies from 3.68 ± 0.43 to 74.55 ± 4.46 ind./m^2 (Fig. 3). The population abundance is statistically significantly different between the studied types of technosols (Kruskal-Wallis test: $H = 1812.7$, $p < 0.001$). The most favorable conditions for this snail species are formed in the gray-green clays and loess-like loams (Fig. 1). The less favorable habitats are formed in the pedozems and most extreme is in the red-brown clays. During the study period, the statistically significant trend of the *B. cylindrica* abundance decline in years was reveled (Kruskal-Wallis test: $H = 52.3$, $p < 0.001$). A common feature is the tendency to reduce the abundance of *B. cylindrica* during the year (Kruskal-Wallis test: $H = 24.6$, p = $p < 0.001$). But deviation from the specified pattern can be observed depending on the technosol type and from year to year. Thus, in 2012, for all technosol types, *B. cylindrica* population decreased during the year. The local minimum of population size was observed in the 2013 summer on loess-like loams and gray-green clay, and vice versa; at the same time, there was a local maximum of *B. cylindrica* population size on pedozems and red-brown clays. In 2013, in technosol types except loess-like loams, the local maximum of the *B. cylindrica* abundance was detected. In 2013, the local maximum of the population abundance was observed in the spring. Deviations from the general trend of decline for the year occurred in the conditions of total low abundance of snails in the spring.

Most often the log-normal and the Zipf-Mandelbrot models are the best for the analytical fitting of empirical data on *B. cylindrica* abundance patterns (Table 1). The Zipf-Mandelbrot model is the best in a slightly greater number of cases than the log-normal model. The Zipf-Mandelbrot model gives the best results in a considerable number of cases for *B. cylindrica* population in biotopes on loess-like loams, and almost parity between these models is observed for other technosol types. For the spring season, there is a slight advantage of the log-normal model for fitting the *B. cylindrica* abundance. For the summer season, there is parity in the descriptive abilities of the models, and Zipf-Mandelbrot model has priority in the autumn.

Thus, distributions of the *B. cylindrica* abundance in the technosol types, which are significantly different in terms of the environmental conditions that is indicated in the total level of the abundance and features of population dynamics can be described by the two models – log-normal and Zipf-Mandelbrot. The Zipf model is very close to the Zipf-Mandelbrot model and quite likely to reasonably choose one of them, the larger data set is needed. The Motomura (geometric series) model describes a system with a high level of competitive relations. The broken stick model on the contrary, describes the situation with a random distribution of system components, which is possible under the conditions of abundant resources and the

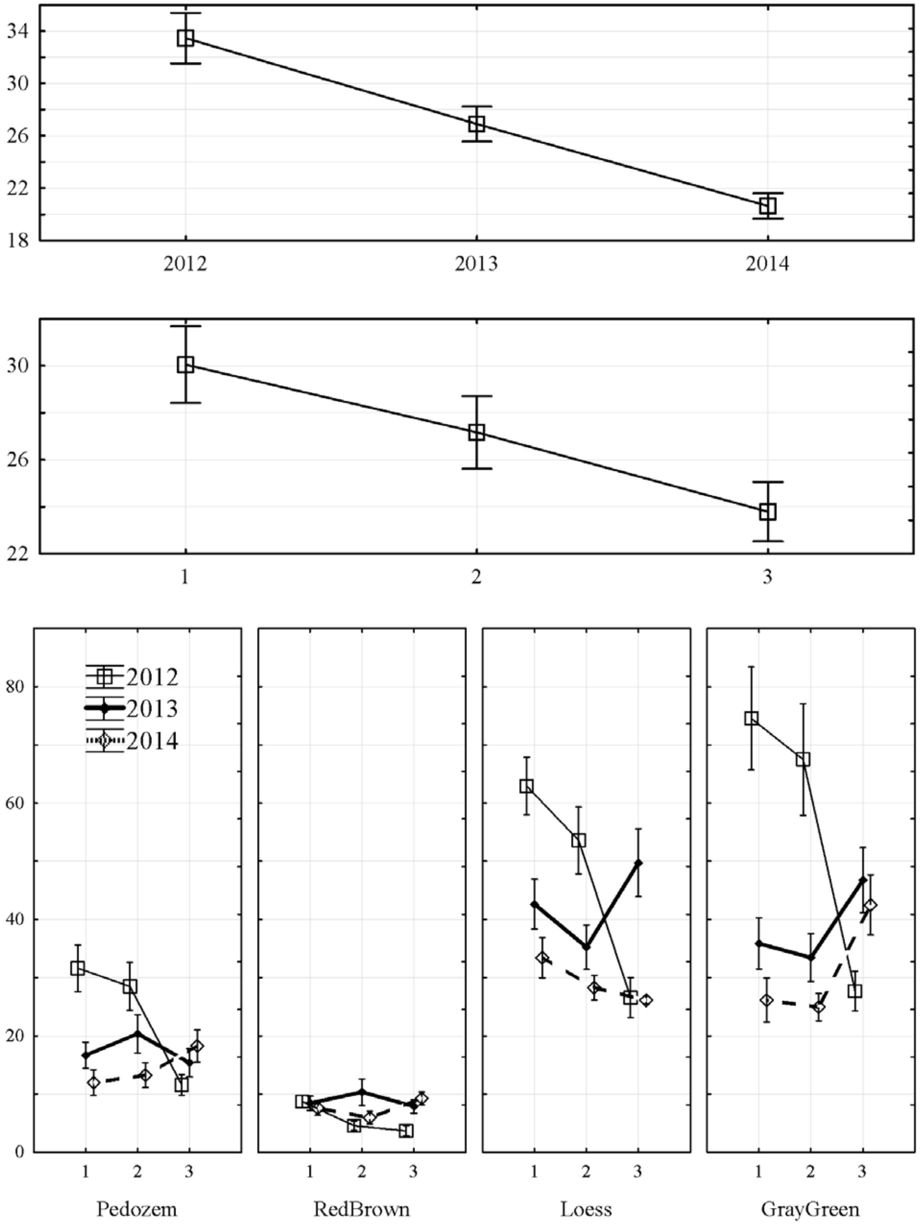


Fig. 3. Dynamics of the *Brephulopsis cylindrica* population abundance in different technosol types (ind./m^2, mean ± 95% confidence interval).
Notes: 1 – spring; 2 – summer; 3 – autumn.

absence of significant competition. Thus, the obtained results indicate that such alternatives as considerable competition in the population *B. cylindrica*, and complete lack of competition may be rejected.

T a b l e 1. The number of the best models that describe the distribution of *Brephulopsis cylindrica* abundance in the different types of technosols or the seasons of the year.

The technosols types or seasons	**Model**	
	Lognormal	**Zipf-Mandelbrot**
Technosol type		
Loess-like loams	3	6
Gray-green clays	4	5
Red-brown clays	5	4
Humus-rich layer	4	5
Season		
Spring	7	5
Summer	6	6
Autumn	3	9
Total	16	20

T a b l e 2. The number of the best models that describe the distribution of *Monacha cartusiana* abundance in the different types of technosols or the seasons of the year.

The technosols types or seasons	**Model**			
	Lognormal	**Mandelbrot**	**Preemption**	**Null**
Technosol type				
Loess-like loams	4	3	1	1
Gray-green clays	4	5	–	–
Red-brown clays	7	–	2	
Humus-rich layer	6	2	1	–
Season				
Spring	5	5	2	–
Summer	8	2	1	1
Autumn	8	3	1	–
Total	21	10	4	1

The Zipf-Mandelbrot and log-normal models describe systems that are characterized by complex organization and nonlinear response to external effects. A competitive relationship between individuals of one species or different species in community can act as one of the mechanisms of such organization formation. We can assume that the role of a competitive relationship is variable and can change from the state of complete control of the population dynamics (as required by the Motomura model) to the absence of such control (as required by the broken stick model).

The Zipf-Mandelbrot model can also describe a system characterized by fractal properties. The fractal properties were shown for populations of terrestrial molluscs. This allows you to hypothesize the existence of a hierarchical organization of spatial structure of populations of terrestrial molluscs, which are formed on technosols.

The abundance of the *Monacha cartusiana* in the studied habitats varies from 2.25 ± 0.27 to 18.64 ± 1.08 ind./m^2 (Fig. 4). The population abundance is statistically significantly different between the studied types of technosols (Kruskal-Wallis test: $H = 324.14$, $p < 0.001$).

The loess-like loams and red-brown clays form the most favorable conditions for this species. The gray-green clays are less favorable habitats. The pedozems are the most extreme. During the period of investigation, the greatest *Monacha cartusiana* abundance was observed in 2012, and the minimum – in 2013. The interannual differences of the abundances are

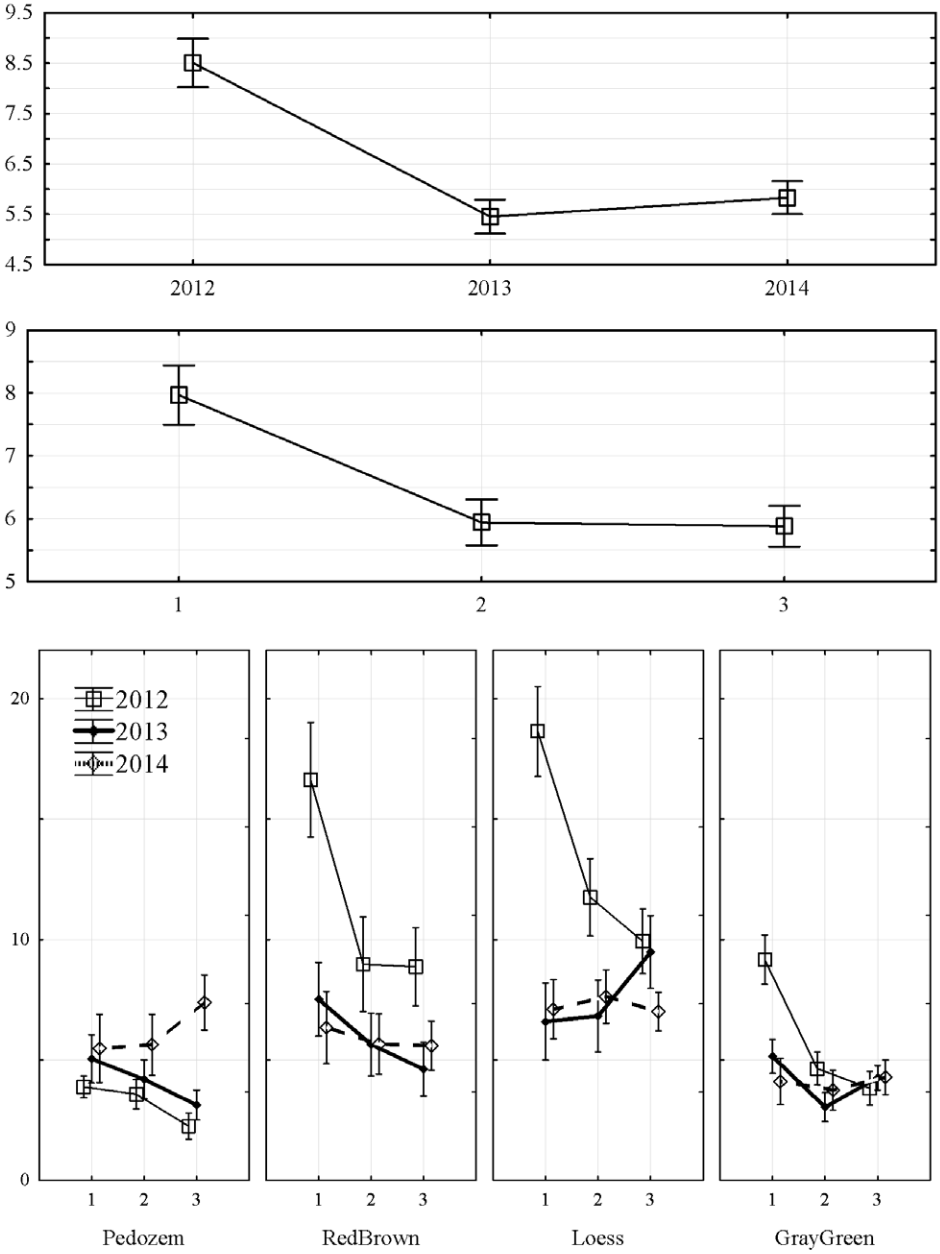


Fig. 4. Dynamics of the *Monacha cartusiana* population abundance in different technosol types (ind./m^2, mean ± 95% confidence interval).
Notes: 1 – spring; 2 – summer; 3 – autumn.

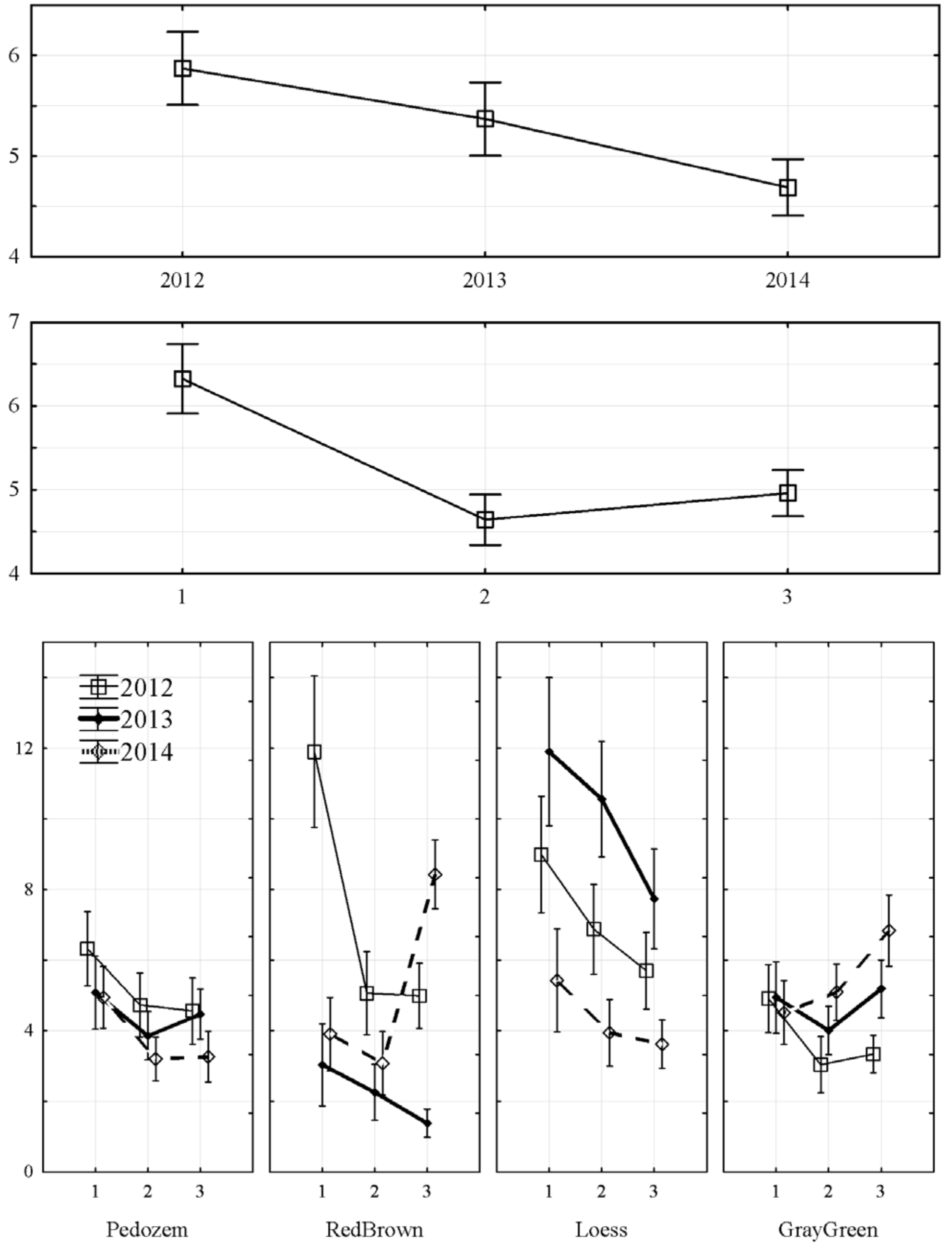


Fig. 5. Dynamics of the *Chondrula tridens* population abundance in different technosol types (ind./m^2, mean ± 95% confidence interval).
Notes: 1 – spring; 2 – summer; 3 – autumn.

Table 3. The number of the best models that describe the distribution of *Chondrula tridens* abundance in the different types of technosols or the seasons of the year.

The technosol types or seasons	**Model**			
	Lognormal	**Mandelbrot**	**Zipf**	**Null**
Technosol type				
Loess-like loams	1	2	5	1
Gray-green clays	–	2	7	–
Red-brown clays	–	3	6	–
Humus-rich layer	–	1	8	–
Season				
Spring	–	4	7	1
Summer	1	1	10	–
Autumn	–	3	9	–
Total	1	8	26	1

Table 4. *Helix lucorum* population abundance in different technosol types (ind./m^2, mean ± 95% confidence interval).

Technosol type	**Рік**	**Season**		
		Spring	**Summer**	**Autumn**
Red-brown clays	2012	–	–	–
	2013	–	–	–
	2014	0.04 ± 0.04	–	–
Loess-like loams	2012	1.60 ± 0.25	0.46 ± 0.15	0.69 ± 0.16
	2013	0.53 ± 0.17	1.07 ± 0.22	0.72 ± 0.17
	2014	0.53 ± 0.14	0.30 ± 0.13	0.30 ± 0.13
Humus-rich layer	2012	0.04 ± 0.04	–	–
	2013	–	0.34 ± 0.11	–
	2014	–	–	–
Gray-green clays	2012	0.38 ± 0.13	0.19 ± 0.08	0.19 ± 0.08
	2013	0.11 ± 0.07	0.27 ± 0.10	0.50 ± 0.13
	2014	–	0.04 ± 0.04	–

statistically significant (Kruskal-Wallis test: H = 125.9, $p < 0.001$). For the total sample size of the role of seasonal changes during the year is statistically significant (Kruskal-Wallis test: H = 38.60, $p < 0.001$). The general trend is to reduce the population abundance from spring to summer and population stabilization from summer to autumn. The population abundance between summer and autumn is not statistically significantly different (Kruskal-Wallis test: = 1.53, $p = 0.37$). The clearest trend of downsizing for the year was observed in 2012 in all technosol types. In 2013, the decline during the year took place only in pedozems and red-brown clays. This year, the growth of the population abundance took place in the autumn on loess-like loams and red-brown clays. In 2014, to the overall low *M. cartusiana* abundance, there were fluctuation oscillations of the population that occurred during the year in all technosol types studied. Thus, the smaller the overall *M. cartusiana* population abundance, the less precise is the trend of the population dynamic during the season. For red-brown clays and loess-like loams, there were trends towards a decreasing of total abundance for the season with a sharp drop in the population between spring and summer. The stable abundance level of molluscs during the year is characterized for pedozems. The summer minimum of the *M. cartusiana* abundance and the subsequent compensatory increase in the autumn were revealed for the gray-green clays.

The lognormal model has the most explanatory capabilities to fit the distribution of the *M. cartusiana* abundance from the considered statistical models (Table 2). According Akaike criterion, Zipf-Mandelbrot model demonstrates a higher explanatory capacity in fewer cases, but fairly regularly. In some cases, the distribution of the *M. cartusiana* abundance is best described by Motomura or broken stick models.

Only the most general trends of explanatory capacity of the statistical distribution models of the abundance allow to distinguish the population within technosol types. Thus, significant dominance of the lognormal model explanatory capacity is characteristic for the population on the red-brown clays and pedozems. Almost complete parity of the lognormal and Zipf-Mandelbrot models was revealed for population on loess-like loams and gray-green clays. The *M. cartusiana* population distribution in the spring can be best described by the Zipf-Mandelbrot and lognormal models. The summer and autumn distributions can be best described by the Zipf-Mandelbrot model. In the seasonal aspect, there is a shift from such a distribution, which can mainly be explained by the Zipf's law in the spring and summer to such that is mainly be explained by the Motomura law in autumn. Thus, the *M. cartusiana* population dynamics over time was accompanied by changes in population characteristics of the statistical distribution of this species abundance. The population distribution during the abundance sharp increase can be described by the Zipf's model. The possibility of outbreaks is more typical for the habitat in loess-like loams, which can be considered as the most favorable environment for the existence of this species.

The abundance of the *Chondrula tridens* in the studied habitats varies from 1.38 ± 0.20 to 11.90 ± 1.06 ind./m^2 (Fig. 5). The population abundance is statistically significantly different between the studied types of the technosols (Kruskal-Wallis test: H = 81.04, $p < 0.001$). The loess-like loams form the most favorable conditions for this species. The *Ch. tridens* population density is much smaller in the red-brown clays, and the lowest in the pedozems and gray-green clays (the population density in the last two technosols are not statistically significantly different from each other, Kruskal-Wallis test: H = 0.58, $p = 1.00$).

The decline of the *Ch. tridens* population abundance was observed during the period of studies over the years (Kruskal-Wallis test: H = 21.66, $p < 0.001$). In the seasonal aspect, the population dynamics was characterized by a spring maximum and a summer minimum. The autumn raising was negligible in comparison with the summer population level. The abundance seasonal differences were statistically significant (Kruskal-Wallis test: H = 34.78, $p < 0.001$). The differences between abundances in summer and in autumn were not statistically significant (Kruskal-Wallis test: H = 1.86, $p = 0.18$). It should be noted that there are significant deviations from the established seasonal pattern of *Ch. tridens* population variability depending on the technosol type and every year. Throughout the year, the downward trend of the *Ch. tridens* population dynamic was revealed for loess-like loams. The similar pattern with less amplitude variation was also confirmed for pedozems. The considerable variability of seasonal dynamics was reveled for red-brown clays. In 2012, after the spring outbreak, in the summer a dramatic decline was detected with the further stabilization in the autumn. In 2013, monotonic decline was observed throughout the year with a very low starting abundance in the spring. In 2013, the abundance outbreak occurred in the autumn after a summer minimum.

The Zipf model is the best to describe the *Ch. tridens* abundance distribution (Table 3). *Helix lucorum* is a large synanthropic land snail (Korábek et al., 2018), which has been recently reported from a studied territory (Balashov et al., 2013). Mollusk *H. lucorum* was regularly found in the loess-like loams and was often found in 2012 and 2013 in the gray-green clays. In this, the species was found twice in the pedozems, and only once in the red-brown clays (Table 4).

Table 5. The number of the best models that describe the distribution of *Helix lucorum* abundance in the different types of technosols or the seasons of the year.

The technosol types or seasons	**Model**			
	Preemption	**Zipf**	**Lognormal/Zipf**	**Null**
Technosol type				
Loess-like loams	3	4	–	2
Gray-green clays	4	–	1	2
Red-brown clays	–	–		1
Humus-rich layer	–	–	1	1
Season				
Spring	2	2	–	3
Summer	4	–	1	2
Autumn	1	2	1	1
Total	7	4	2	6

Table 6. The number of the best models that describe the distribution of mollusks abundance in the different types of technosols or the seasons of the year.

Species	**Model**				
	Lognormal	**Mandelbrot**	**Null**	**Preemption**	**Zipf**
Brephulopsis cylindrica	16	20	–	–	–
Chondrula tridens	1	8	1	–	26
Helix lucorum	2	–	6	7	4
Monacha cartusiana	21	10	1	4	–
Total	40	38	8	11	30

Distribution of the *H. lucorum* abundance can be best described by the Motomura or the broken stick models (Table. 5). The Zipf model best explains the mollusk population distribution in loess-like loams. The Motomura model is the best for population from the biotopes on gray-green clays. In the seasonal aspect, there is a certain predominance of one or another model to describe the abundance distribution of the *H. lucorum*. The broken stick model has an advantage in the spring, Motomura model has an advantage in the summer. The broken stick, Zipf, lognormal and Motomura models demonstrate the almost equal opportunity for abundance distribution fitting. Thus, a significant predominance of descriptive abilities of the two models – Motomury and a broken stick – indicates a certain level of statistical invariance of the *H. lucorum* distribution in a range of environmental conditions, which are formed in technosols. The loess-like loams and gray-green clays form consistently favorable conditions for the existence of mollusks *H. lucorum*. In two other types (pedozems and red-brown clays), the representatives of this species occur sporadically.

The Zipf, lognormal or Zipf-Mandelbrot models have the best ability to explain the abundance distribution of the mollusks populations (Table 6). Mollusks species have some specific composition of models that best describe the abundance distribution of their population in technosols. Thus, the lognormal and Zipf-Mandelbrot models are the best for the most abundant mollusks *Brephulopsis cylindrica* and *Monacha cartusiana*. The Zipf model is the best, and to a lesser extent – the Zipf-Mandelbrot model for less abundant species *Chondrula tridens*. The broken sticks or Motomura models are the best for the rare species *Helix lucorum*.

Table 7. The number of the best models that describe the distribution of mollusks abundance in the different time periods.

Time period	Model				
	Lognormal	**Mandelbrot**	**Null**	**Preemption**	**Zipf**
Year					
2012	12	16	2	4	9
2013	17	10	2	4	10
2014	11	12	4	3	11
Season					
Spring	12	14	4	4	9
Summer	16	9	3	5	10
Autumn	12	15	1	2	11
Total	40	38	8	11	30

Table 8. The number of the best models that describe the distribution of mollusks abundance in the different technosols.

Technosol	Model				
	Lognormal	**Mandelbrot**	**Null**	**Preemption**	**Zipf**
Loess-like loams	8	11	4	4	9
Gray-green clays	11	8	1	1	8
Red-brown clays	9	12	2	4	7
Humus-rich layer	12	7	1	2	6
Total	40	38	8	11	30

The list and the proportion of the models that can best describe the distribution of the mollusk abundance, are almost invariable in time for years (Table 7). In the seasonal aspect the Zipf-Mandelbrot model relatively more often takes precedence in spring or in autumn and lognormal model – in summer. The Zipf, Motomura and broken stick models are not dependent on the time aspect.

A particular range of models that best describes the distribution of mollusks abundance is specific for each technosol type (Table 8). Lognormal model is often effective to describe the mollusk population in the red-brown clays and in the pedozems. The Zipf-Mandelbrot model is best to explain the distribution in loess-like loams and gray-green clays. The broken stick model is often suitable to describe populations in loess-like loams. The Motomura model is also effective in loess-like loams and in gray-green clays, and Zipf model is suitable for population in loess-like loams and pedozems.

Conclusion

The series of models that best describe mollusk abundance distribution is specific for a particular species and technosol type, and generally is invariant over time. Species with high overall level of population abundance that are characterized by distributions are well explained by lognormal model or Zipf-Mandelbrot model. It is believed that these models mark the steady state populations of mollusks, which allows to build complex relationships that combine both competition between individuals of the population, interspecific competition and the possibility of implementing reproductive potential under favorable ecological environment conditions. The abundance of the facultative populations, formed mainly by migration in terms of periodic extinctions micropopulations, is often described by the Motomura model or the broken stick model.

References

Arruda, J.A. (2014). The land snails of a partially reclaimed abandoned coal mine site. *Trans. Kans. Acad. Sci.*, 117(1–2), 15–20. DOI: 10.1660/062.117.0103.

Balashov, I.A., Kramarenko, S.S., Zhukov, A.V., Shklyaruk, A.N., Baidashnikov, A.A. & Vasyliuk A.V. (2013). Contribution to the knowledge of terrestrial molluscs in southeastern Ukraine. *Malacologica Bohemoslovaca*, 12, 62–69.

Beauchamp, G. (2011). Fit of aggregation models to the distribution of group sizes in Northwest Atlantic seabirds. *Mar. Ecol. Prog. Ser.*, 425, 261–268. https://www.jstor.org/stable/24874645

Bonabeau, E., Dagorn, L. & Freon P. (1999). Scaling in animal group-size distributions, *PANAS*, 96, 4472–4477. DOI: 10.1073/pnas.96.8.4472.

Calenge, C. & Basille M. (2008). A general framework for the statistical exploration of the ecological niche. *J. Theor. Biol.*, 252(4), 674–685. DOI: 10.1016/j.jtbi.2008.02.036.

Caraco, T. (1980). Stochastic dynamics of avian foraging flocks. *Am. Nat.*, 115(2), 262–275. www.jstor.org/stable/2460597.

Dvořáková, J. & Horsák M. (2012). Variation of snail assemblages in hay meadows: Disentangling the predictive power of abiotic environment and vegetation. *Malacologia*, 55(1), 151–162. DOI: 10.4002/040.055.0110.

Hall, L.S., Krausman, P.R. & Morrison M.L. (1997). The habitat concept and a plea for standard terminology. *Wildl. Soc. Bull.*, 25(1), 173–182. https://www.jstor.org/stable/3783301

Horsák, M., Hájek, M., Tichý, L. & Juřičková L. (2007). Plant indicator values as a tool for land mollusc autecology assessment. *Acta Oecol.*, 32(2), 161–171. DOI: 10.1016/j.actao.2007.03.011.

IUSS Working Group WRB (2007). *World reference base for soil resources 2006, first update 2007*. World Soil Resources Reports No. 103. Rome: FAO.

Juřičková, L., Horsák, M., Cameron, R., Hylander, K., Mikovcová, A., Hlaváč, J.C. & Rohovec J. (2008). Land snail distribution patterns within a site: The role of different calcium sources. *Eur. J. Soil Biol.*, 44, 172–179. DOI: 10.1016/j.ejsobi.2007.07.001.

Kappes, H., Jordaens, K., Hendrickx, F., Maelfait, J.P., Lens, L. & Backeljau T. (2009). Response of snails and slugs to fragmentation of lowland forests in NW Germany. *Landsc. Ecol.*, 24(5), 685–697. DOI: 10.1007/s10980-009-9342-z.

Kappes, H., Clausius, A. & Topp W. (2012). Historical small scale surface structures as a model for post-mining land reclamation. *Restor. Ecol.,* 20, 322–330. DOI: 10.1111/j.1526-100X.2011.00800.x.

Korábek, O., Juřičková, L., Balashov, I. & Petrusek A. (2018). The contribution of ancient and modern anthropogenic introductions to the colonization of Europe by the land snail *Helix lucorum* Linnaeus, 1758 (Helicidae). *Contrib. Zool.*, 87(2), 61–74. DOI: 10.1163/18759866-08702001.

Książkiewicz-Parulska, Z. & Ablett J.D. (2017). Microspatial distribution of molluscs and response of species to litter moisture, water levels and eutrophication in moist, alkaline ecosystems. *Belg. J. Zool.*, 147(1), 37–53. DOI: 10.26496/bjz.2017.3.

Kunakh, O.N., Kramarenko, S.S., Zhukov, A.V., Kramarenko, A.S. & Yorkina N.V. (2018). Fitting competing models and evaluation model parameters of the abundance distribution of the land snail *Vallonia pulchella* (Pulmonata, Valloniidae). *Regulatory Mechanisms in Biosystems*, 9(2), 198–202. DOI: 10.15421/021829

MacArthur, R.H. (1957). On the relative abundance of bird species. *Proc. Natl. Acad. Sci. USA,* 43(3), 293–295. DOI: 10.1073/pnas.43.3.293.

Mandelbrot, B.B. (1983). *The fractal geometry of nature*. New York: W.H. Freeman and Company.

Martin, K. & Sommer M. (2004). Relationships between land snail assemblage patterns and soil properties in temperate-humid forest ecosystems. *J. Biogeogr.*, 31(4), 531–545. DOI: 10.1046/j.1365-2699.2003.01005.x.

Millar, A.J. & Waite S. (1999). Mollusks in coppice woodland. *J. Conchol.*, 36, 25–48.

Motomura, I. (1932). On the statistical treatment of communities. *Zoological Magazine*, 44, 379–383.

Müller, J., Strätz, C. & Hothorn T. (2005). Habitat factors for land snails in European beech forests with a special focus on coarse woody debris. *European Journal of Forest Research*, 124(3), 233–242. DOI: 10.1007/s10342-005-0071-9.

Nekola, J.C. & Smith T.M. (1999). Terrestrial gastropod richness patterns in Wisconsin carbonate cliff communities. *Malacologia*, 41(1), 253–270.

Nekola, J.C. (2003). Large-scale terrestrial gastropod community composition patterns in the Great Lakes region of North America. *Divers. Distrib.*, 9(1), 55–71. DOI: 10.1046/j.1472-4642.2003.00165.x.

Oksanen, J., Blanchet, F.G., Kindt, R., Legendre, P., Minchin, P.R., O'Hara, R.B., Simpson, G.L., Solymos, P., Stevens, M.H.H. & Wagner H. (2018). *Community ecology package.* R package version 2.5-2. https://CRAN.R-project.org/package=vegan

Okubo, A. (1986). Dynamical aspects of animal grouping: Swarms, schools, flocks and herds. *Advances in Biophysics*, 22 (1986), 1–94. DOI: 10.1016/0065-227X(86)90003-1.

Ondina, P., Mato, S., Hermida, J. & Outeiro A. (1998). Importance of soil exchangeable cations and aluminium content on land snail distribution. *Appl. Soil Ecol.*, 9(1–3), 229–232. DOI: 10.1016/S0929-1393(98)00080-8.

Ondina, P., Hermida, J., Outeiro, A. & Mato S. (2004). Relationships between terrestrial gastropod distribution and soil properties in Galicia (NW Spain). *Appl. Soil Ecol.*, 26(1), 1–9. DOI: 10.1016/j.apsoil.2003.10.008.

Preston, F.W. (1948). The commonness, and rarity, of species. *Ecology*, 29 (3), 254–283. DOI: 10.2307/1930989.

Preston, F. (1962). The canonical distribution of commonness and rarity: Part I. *Ecology*, 43, 185–215. DOI: 10.2307/1931976.

Rosin, Z. M., Lesicki, A., Kwiecinski, Z., Skorka, P. & Tryjanowski P. (2017). Land snails benefit from humanalterations in rural landscapes and habitats. *Ecosphere*, 8(7), e01874. DOI: 10.1002/ecs2.1874.

Schenková, V., Horsák, M., Plesková, Z. & Pawlikowski P. (2012). Habitat preferences and conservation of *Vertigo geyeri* (Gastropoda: Pulmonata) in Slovakia and Poland. *J. Molluscan Stud.*, 78, 105–111. DOI: 10.1093/mollus/eyr046.

Schilthuizen, M. (2013). Rapid, habitat-related evolution of land snail colour morphs on reclaimed land. *Heredity*, 110, 247–252. DOI: 10.1038/hdy.2012.74.

Szybiak, K., Błoszyk, J., Koralewska-Batura, E. & Gołdyn B. (2009). Small-scale distribution of wintering terrestrial snails in forest site: relation to habitat conditions. *Pol. J. Ecol.*, 57(3), 525–535.

Weaver, K.F., Anderson, T. & Guralnick R. (2006). Combining phylogenetic and ecological niche modeling approaches to determine distribution and historical biogeography of Black Hills mountain snails (Oreohelicidae). *Divers. Distrib.*, 12(6), 756–766. DOI: 10.1111/j.1472-4642.2006.00289.x.

Whittaker, R.H. (1965). Dominance and diversity in land plant communities. *Science*, 147, 250–260. DOI: 10.1126/science.147.3655.250.

Wood, C.C. (1985). Aggregative response of common mergansers (Mergus merganser): predicting flock size and abundance on Vancouver Island salmon streams. *Can. J. Fish. Aquat. Sci.*, 42(7), 1259–1271. DOI: 10.1139/f85-157.

Yorkina, N., Maslikova, K., Kunah, O. & Zhukov, O. (2018). Analysis of the spatial organization of *Vallonia pulchella* (Muller, 1774) ecological niche in Technosols (Nikopol manganese ore basin, Ukraine). *Ecologica Montenegrina*, 17, 29–45. www.biotaxa.org/em

Yorkina, N., Zhukov, O. & Chromysheva, O. (2019). Potential possibilities of soil mesofauna usage for biodiagnostics of soil contamination by heavy metals. *Ekológia (Bratislava)*, 38(1), 1–10. DOI: 10.2478/eko-2019-0001.

Zipf, G. K. (1949). *Human behavior and the principle of least effort*. Cambridge: AddisonWesley.

Zipkin, E. F., Leirness, J. B., Kinlan, B. P., O'Connell, A. F. & Silverman, E. D. (2014). Fitting statistical distributions to sea duck count data: implications for survey design and abundance estimation. *Statistical Methodology*, 17, 67–81. DOI: 10.1016/j.stamet.2012.10.002.

Ekológia (Bratislava) Vol. 38, No. 4, p. 382–391, 2019
DOI:10.2478/eko-2019-0028

BREEDING BIOLOGY OF BOOTED EAGLE *Hieraaetus pennatus* (Gmelin, JF, 1788): THE FIRST STUDY IN NORTH AFRICA

ISSAM BOUCHAHDANE[1*], MOHAMED BOUKHEMZA[2], KARIM SOUTTOU[1], AREZKI DERRIDJ[2]

[1]Department of Agronomy, Faculty of Science of Nature and Life, University of Djelfa, 17000 Djelfa, Algeria; e-mail: ibouchahdan@gmail.com
[2]Earth Biology and Ecology Laboratory, Faculty of Biological and Agricultural Sciences, Mouloud Mammeri University of Tizi Ouzou, B.P. 17 R.P., DZ, 15000, Algeria

* Author for correspondence

Abstract

Bouchahdane I., Boukhemza M., Souttou K., Derridj A.: Breeding biology of booted eagle *Hieraaetus pennatus* (Gmelin, JF, 1788): the first study in North Africa. Ekológia (Bratislava), Vol. 38, No. 4, p. 382–391, 2019.

Knowledge and study of breeding birds' biology is essential to understand the dynamic, assess productivity and identify threats. The population of booted eagles (*Hieraaetus pennatus*) was studied during three consecutive breeding seasons (2013–2015) in the Chettabah forest. Each year, arrivals of Booted eagles are recorded during March (15 individuals in 2013, 12 in 2014 and 11 in 2015). For the three years of study, the mean breeding pair's density was 1.23 ± 0.153/km^2, incubation period ranged from 33 to 38 days, whilst the frequency distribution of clutch size was 67.6% for 2 eggs, 27% for 1 and 5.4% for 3. Amongst the 63 eggs monitored, 55 hatched. No significant variation in hatching rate was observed during the 3 years (90% in 2013, 76.19% in 2014 and 84% in 2015). The comparison of breeding parameters with those reported in Spain showed a noticeable similarity. Further investigations should be conducted to improve the provision of information to support sound management interventions of this North African population.

Key words: booted eagle, paired density, breeding biology, forest, North Africa.

Introduction

Raptors are at the top of their food chain and are considered to be biologically important and environmentally sensitive (Olendorff et al., 1989) and ecosystem health indicators (Newton, 1979; Donázar et al., 2002; Sergio et al., 2005). They act as a conservation flagship (Simberloff, 1998).

The booted eagle is a summer resident in southern Europe, although some individuals winter in the Mediterranean Basin (Martínez, Sánchez-Zapata, 1999; Baghino et al., 2007). The booted eagle *Aquila pennata* is mainly a trans-Saharan migratory species, but some individuals overwinter in Southern Europe and North Africa (Zalles, Bildstein, 2000; Thévenot et al., 2003).

The majority of the population migrates to tropical Africa through the Strait of Gibraltar and the Bosphorus (Cramp, Simmons, 1980). Migratory routes remain poorly known, with one study based on one individual tracked by satellite telemetry published to date (Chevallier et al., 2010). The booted eagle *Hieraaetus pennatus* is considered 'rare' in Europe, where it breeds in Western Europe (France, Portugal and Spain) and Eastern Europe (most of them in Russia, Turkey and Ukraine) with several thousands of couples (Garcia-Dios, 2014).

Very few biological aspects of the booted eagle are known, and this species is considered one of the least known of all European raptor (Suárez et al., 2000; Bosch et al., 2005). Although some scientific studies have been focused on the habitats (Suárez et al., 2000; Díaz, 2005; Martínez et al., 2007) in North Africa, studies on raptors, amongst them, the booted eagle, are rare, which are mostly conducted on distribution and inventories, and a few phenological studies are also conducted (Isenmann, Moali, 2000; Thévenot et al., 2003; Isenmann et al., 2005), but there was no work on raptors breeding biology, especially the booted eagle. The present work on the breeding of several pairs of the booted eagle in the mountain ranges of Constantinois (Algeria) provided the first contribution to the study of the breeding biology of this species in North Africa.

Material and methods

Study area

Field data were collected from the Chettabah forest (24 000 h), province of Constantine (36°18 'to 36°21' N; 6°26 'and 6°30' E), which is located in the centre East of Algeria, about 245 km from the Algerian–Tunisian borders, 431 km from the capital Algiers to the west, 89 km from Skikda to the north (the Mediterranean Sea) and 235 km from Biskra to the south (The Sahara).

The Chettabah forest is built on a majestic rock located on both sides of Oued Rhumel; it is thus surrounded by real natural obstacles; the landmarks show that the region is heterogeneous when it comes to its position and its sea level; the region is located at an altitude of 694–1350 m above sea level. The study area has a sub-humid and semi-arid climate, with an average temperature varying between 8° in winter and 34° in summer, at an altitude of 694 m. It is characterised by a very rugged terrain crisscrossed by several seasonal thalwegs and spread over all forests with the presence of numerous ridges (Belgherbi et al., 2018). Its vegetation is dominated by the Aleppo Pine, the Green Oak, the Atlas Pistachio and). Its vegetation is dominated by the Aleppo Pine, the Green Oak, the Atlas Pistachio and the maquis shrublands are of low proportion. (Fig. 1).

Field work

This work was carried out for three consecutive breeding seasons (2013–2015) from the end of March until the end of September. We conducted a bi-monthly phenological monitoring and counting of the booted eagle in the Chettabah forest and a weekly monitoring and counting of breeding couples (2 times per week) in the the zouaoui cliff using binoculars (90 * 90) and a telescope (60 * 80) to locate the nests.

Once the nest is located, we noted the date of its discovery, its geolocation using a GPS (Global Positioning System) (Garmin) and its height from the ground. The nest dimensions (internal diameter and external diameter) were recorded; we also measured the size of eggs (number of eggs per nest), the weight of the eggs using an electronic scale (0.01 g of accuracy) and their length (A) and width (B) using an electronic Vernier caliper.

Hoyet's (1979) formula V = 0.51 x A x B2 allowed us to calculate the eggs volume. The two visits per week allowed us to estimate the incubation period (Del Hoyto et al., 1994) and also to monitor the hatching success (the ratio between the number of eggs laid and the number of hatched eggs) and fledging success, which translates to the

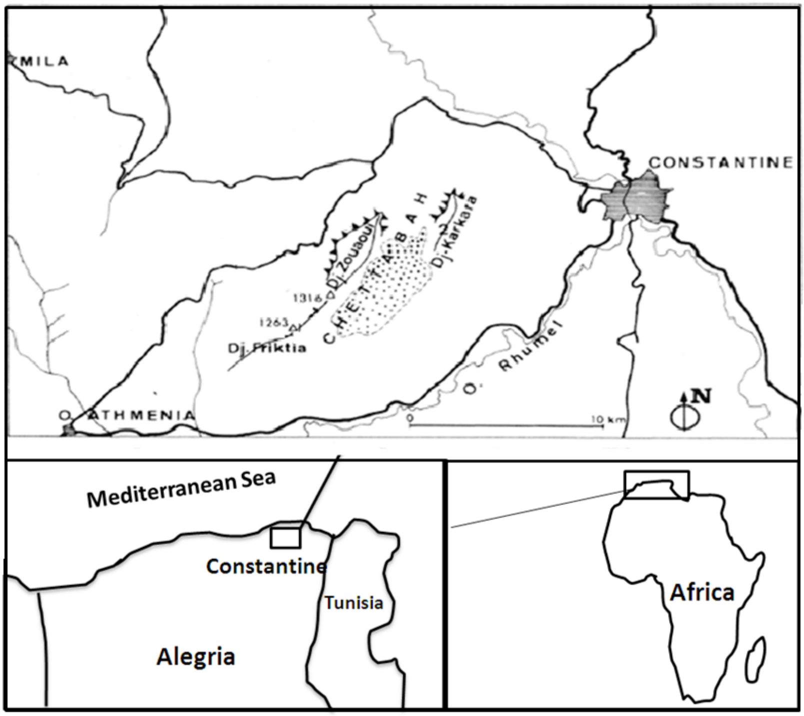


Fig. 1. Geographic location of the study area: Chettabah forest (Marre et al., 1977).

number of fledgling chicks per contribution to the number of eggs hatched (Martínez et al., 2006a). The flight rate was calculated by dividing the number of fledgling chicks by the total number of active nest (García-Dios, 2014). The nests that were inaccessible were followed using a binocular and a telescope to be able to note their clutch size, their hatching success and fledging success.

Statistical analysis

All analyses were performed after verification of normality criteria (Kolmogorov–Smirnov test) and homoscedasticity (Levene's test) of the different dependent and independent variables. The one-way ANOVA (Anlaysis of variance) parametric test was performed to see if there is a 3-year variation in booted eagle abundance as well as nest and egg size and incubation period. Other non-parametric test such as the Kruskal–Wallis test was used to see if the variation depends on the years of hatching success and fledging success, and chi-squared test allowed us to evaluate the distribution of the clutch size according to the 3 years of monitoring. Statistical analyses were carried out using R (R development core 2015).

Results and discussion

Phenology, abundance and pairs density

The population size of the booted eagles in the Chettabah forest remain stable during the 3 years of survey (ANOVA: F = 0.551; df = 2; $p > 0.05$) (Fig. 2).

The booted eagle occupies the Chettabah forest for almost 7 months (from the end of March to the end of September). In this study, the booted eagle arrives in March every year with a large number of individuals (15 individual in 2013, 12 individual in 2014 and 11 individual in 2015). The population size reaches its first peak at the beginning of May with 28 individual in 2013, 35 individual in 2014 and 36 individual in 2015; however, it reaches its second peak at the end of August with a fairly large number of individuals (33 individual in 2013, 38 individual in 2014 and 42 individual in 2015; Fig. 3).

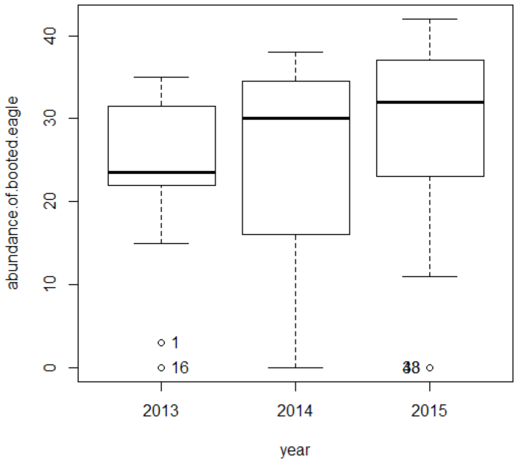


Fig. 2. Box plot representing the variation of booted eagle abundance amongst 3 years of monitoring.

In Algeria, Isenmann et Moali (2000) reported that the species arrives in the middle of March and leaves at the end of September. Our results are similar to those recorded in the south-west of the Mediterranean (south-east Spain) (Martínez et al., 2006b; Jiménez-Franco et al., 2014). In the Chettabah ravin , the number of pairs varied in the range of 11–14 pairs (11 pairs in 2013, 12 pairs in 2014 and 14 pairs in 2015), with a mean density of 1.23 ± 0.153/km^2. Our results are more important than those reported in the south of Spain, Murcia (ranked first site in Europe), with 1.29 ± 0.3/10 km^2 (Martínez et al., 2006b, 2011).

Garzón (1974): 1 pp./0.67 km^2 , Díaz Robledo (Martí, Del Moral, 2003): 11 pp./35 km^2 in the western portion of Malaga, Hiraldo and González Grande (1987): 1 pp./0.80 km^2 (minimum) and 1 pp./15 km^2 (maximum), Garcia Dios in Bosch (2003): 1 pp./3.20 km^2 (Candeleda, Ávila) and 1 pp./2.03 km^2 (valle del Tiétar, excepting Candeleda), Díaz (2006): 18 pp./10 km^2 (Madrid), In Mallorca there is an abundance of 0.048 pp. / km^2 and in Menorca 0.130 pp. / km^2 (Viada, de Pablo, 2009), Abundances have been recorded as 1 territory/10 km^2 in Murcia, 0.8 territories/10 km^2 in Castellon, 0.7 territories/10 km^2 in extremadura, 0.7 territories/10 km^2 in Castile and León 0.6 territories/10 km^2 in Andalusia, Cantabria and la Rioja (Palomino, Valls, 2011).

Abundances have been recorded as 1 territory/10 km^2 in Murcia, 0.8 territories/10 km^2 in Castellon, 0.7 territories/10 km^2 in extremadura, 0.7 territories/10 km^2 in Castile and León 0.6 territories/10 km^2 in Andalusia, Cantabria and la Rioja (Palomino, Valls, 2011).

Description of habitat and nest measurement

A total of 37 nests were monitored, including 28 nests (75.67%) for 3 consecutive years, 11 nests in 2013, of which 08 were measured; 12 nests in 2014, of which 10 were measured;

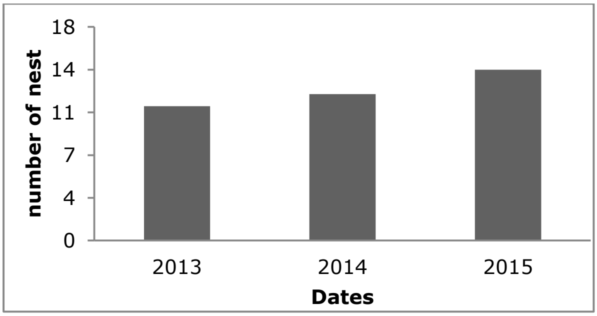


Fig. 3. Number of nests for booted eagle in forest of Chettabah amongst 3 years of monitoring.

and 14 nests in 2015, of which 10 were measured. All the nests were built on branches of pistachio (67.56%) and Aleppo pine (32.43%) (Fig. 3). In Europe, the booted eagle builds its nest on several types of support (e.g. *Pinus sylvestris* (Díaz, 2005), *P. pinaster* (García-Dios, Viñuela, 2000), *P. nigra* (Díaz et al., 1994), *Quercus suber* (Urios et al., 1991), *Fagus sylvatica* (Iribarren, Rodríguez Arbeloa, 1988) and *Eucalyptus eucalyptus* (Suárez et al., 2000). Our results are approximately similar to those announced by Martínez (2002) in Murcia and (Bosch, Prat, 2011) in Catalonia, where *Pinus halepensis* is the preferred support for the nesting of this species. The internal diameter of the nest varies between 20 and 46 cm. The external diameter varies between 63 and 108 cm; the height varies between 125 and 210 m. All the measured parameters do not vary according to the 3 years of monitoring: internal diameter (ANOVA: F = 3.171; df = 2; p = 0.055), external diameter (ANOVA: F = 2.347; df = 2; p = 0.111), height (ANOVA: F = 2.311; df = 2; p = 0.115). Our results confirm what was proved by Zuberogoitia et al. (2011) in Spain (Bizkaia) (Table 1).

Clutch size and egg measurement and period of incubation

The clutch size varies between 1 and 2–3 eggs. The number of nests with a modal clutch size is 2 eggs (67.6%), 1 egg (27%) and 3 eggs (5.4%) among the 3 years. The mean clutch size is 2.7 ± 0.48 in 2013, 2.66 ± 0.48 in 2014 and 2.6 ± 0.50 in 2015. The distribution of this clutch size shows that there is no a significant difference amongst the 3 years (X2 = 1.18; df(degree of freedom)= 4; $p > 0.05$; Fig. 4).

For most species, nest construction involves a considerable investment of time and energy that could be reallocated directly to reproduction if nests were reused, resulting pos-

Table 1. Comparison of booted eagle nest characteristics in North Africa (present study) and Southern Europe (Spain).

Nests characteristics		Mean	Min	Max	SD	Coefficient of variation (%)
Present study (n = 37)	Internal diameter (cm)	34.33	20	46.00	6.52	18.99
	External diameter (cm)	82.33	63	108	11.82	14.35
	Height of nests (m)	164.33	125	210	21.45	13.05
Spain (Zuberogoitia et al., 2011)	Internal diameter (cm)	27.5	-	-	-	-
	External diameter (cm)	72.5	-	-	-	-
	Height of nests (m)	-	-	-	-	-

sibly in earlier clutch initiation or larger clutch sizes (Redmond et al., 2007). Booted eagle clutch size varies between 1 and 2 eggs, and, rarely, 3 eggs (Martínez, 2002). Our results are similar to those found by Díaz Ruiz and Cebollada Baratas (2011) in Guadarrama Mountains (Spain) with a modal clutch size of 2 eggs (67%), 1 egg (25%) and 3 eggs (8%) (Díaz Ruiz, Cebollada Baratas, 2011).

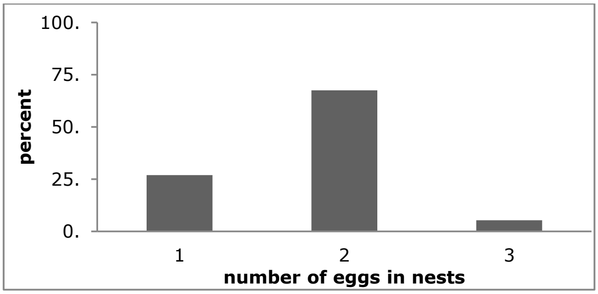


Fig. 4. Frequency distribution of clutch size of booted eagle in North Africa during the 3 years of study.

In the total of 63 eggs, 49 eggs were measured, of which egg length does not vary significantly amongst 3 years of monitoring (ANOVA: $F = 0.148$; $df = 2$; $p > 0.05$).

The length of the eggs varies between 50.78 and 58.20 mm during the 3 years of monitoring, with an average of 63.60 ± 2.09 mm in 2013, 54.97 ± 1.61 mm in 2014 and 54.69 ± 2.36 mm in 2015.

No significant difference was observed in the eggs width amongst the 3 years (ANOVA: $F = 2.016$; $df = 2$; $p > 0.05$). Egg width varies between 40.32 and 49.12 mm, with an average of 55.59 ± 1.86 mm in 2013, 54.91 ± 1.66 mm in 2014 and 56.95 ± 2.43 mm in 2015.

No significant difference was observed in the weight of the booted eagle's eggs amongst the 3 years (ANOVA: $F = 0.842$; $df = 2$; $p > 0.05$). The weight of the eggs varies between 54 and 73.40 g, with an average of 66.51±4.85 g in 2013, 64.28 ± 6.3 g in 2014 66.07 ± 4.18 g in 2015. As for egg volume, it differs significantly among the years (Kruskal–Wallis test: $X2 = 6.53$, $p < 0.05$).

Numerous studies have shown that egg size is an important variable in the history of life and it influences survival and reproductive success (Thomas, 1983; Stokland, Amundsen, 1988; Rutkowska, Cichon, 2005). Our eggs biometric dimensions are similar to what has been confirmed in Spain (García Dios, unpublished data, The Booted Eagle book in Spanish) (Table 2).

Table 2. Comparison of booted eagle's nest sizes in North Africa (present study) and Southern Europe (Spain).

Eggs characteristics		**Mean**	**Min**	**Max**	**SD**	**Coefficient of variation (%)**
Present study (n=49)	Length (mm)	54.7665	50.78	58.2	2.01027	3.67
	Width (mm)	44.1763	40.32	49.12	2.06441	4.66
	Weight (g)	65.5898	54	73.4	5.18752	7.89
Spain Garcia dios 2014 (n =113)	Length (mm)	54.73	-	-	-	-
	Width (mm)	44.26	-	-	-	-
	Weight (g)	60.5	-	-	-	-

The eggs are incubated by the female eagle during a period that varies between 33 and 38 days. No significant difference was observed in the incubation period amongst the 3 years of monitoring (ANOVA: F = 0.200; ddl = 2; p > 0.05). Incubation is an essential part of breeding birds (Deeming, 2002). It is an energetically costly step, and it takes time in the breeding cycle (Vleck, 1981a,b; Reid et al., 2002). Our results confirm what has been proved by various studies that the incubation period of the booted eagle eggs varies between 35 and 38 days (Iribarren, 1975; Cramp, Simmons, 1980; Bezzel, 1985; Iribarren, Rodríguez, 1988; Perrins, 1987; Del Hoyto et al., 1994; Martínez et al., 2005).

4-success and mortality rate

In the total of 63 eggs, 55 eggs were hatched, with an average of 54.22 ± 2.36 (90%) in 2013, 54.11 ± 2.88 (76.19%) in 2014 and 51.77 ± 2.04 (84%) in 2015. No significant difference was observed in the hatching success in functions of the 3 monitoring years (Kruskal–Wallis test: X2 = 0.570, p > 0.05). The average number of fledglings is 52.69 ± 2.15 (88.88%) in 2013, 73.38 ± 2.73 (87.50%) in 2014 and 54.69 ± 2.36 (85.71%) in 2015. The number of fledging was not significantly variable amongst the years of study (Kruskal–Wallis test: X2 = 0.040; p > 0.05). Our results regarding hatching success and fledged success are close to what have been proved by Martínez et al. (2006a,b and 2011) in Spain (Table 3).

The fledgling chick/nest number is 1.297 chicks (n = 37) is close to S^a de Guadarrama, 1.20 chick (Díaz, 2006); is important in Valle del Tiétar, 1.02 chicks (García Dios, Viñuela, 2000); and less important to southern Spain and northeastern Spain: Madrid, 1.46 (Díaz, 2006) and Murcia, 1.57 chick (Sánchez et al., 1995) and 1.65 chick (Martinez et al., 2006a). Alto Tajo: 1.70 chick (Arroyo, 2003), Central Catalonia: 1.82 chick (Bosch, 2003) and North S^a: 1.72 chick (Díaz, 2006). The mortality rate is 11.11% (2013), 12.50% (2014), 17.64% (2015) of an average 13.752% during the study years, (Martínez et al., 2006a) suggested that, The main cause of mortality is predation by carnivores and owls. In our study area, owls were noticed by regurgitation pellets as well by their nocturnal scream, which can be one of the causes of loss of chicks throughout the study period. Our results are significant in contribution to those announced at Téitar Valley with a rate of 30.3% (García Dios, Viñuela, 2000).

Table 3. Comparison of booted eagle's nest success in North Africa (present study) and Southern Europe (Spain).

	Year	**Number of eggs laid**	**Number of eggs hatched (%)**	**Number of young fledged (%)**
Present study	2013	20	90	88.88
	2014	21	76.19	87.50
	2015	25	84	85.71
Spain (Martínez, 2006, 2011)	2002	24	20(81)	19(95)
	2003	27	22(81)	18(82)
	2004	33	24(73)	13(54)
	2007	-	18(90)	9(90)
	2008	-	17(68)	12(86)
	2009	-	24(73)	15(83)

Conclusion

From this study, which turns out to be the first in North Africa, we were able to shed some light on the breeding biology of booted eagles in Algeria. Although pioneering, this work must be developed further in the near future. Indeed, certain aspects of the bioecology of this raptor deserve special attention to allow improving our understanding on the mechanisms driving this population. It would be, therefore, of great importance to initiate studies on multiscale factors (local versus landscape) affecting (i) the nest site selection, (ii) breeding parameters and (iii) nest survival.

References

Arroyo, B. (2003). *Censo de rapaces forestales en el Parque Natural del Alto Tajo*. Bioma TBC y Consejería de Medio Ambiente de Castilla-La Mancha. Informe inédito.

Baghino, L., Bottero, M., Pedemonte, R. & Rapetti C. (2007). La migrazione post-nuziale dell'Aquila minore ad Arenzano nel 2006. *Il Biancone*, 2, 22–26.

Belgherbi, B., Benabdeli, K . & Mostefai K. (2018). Mapping the risk of forest fires in Algeria: application of the forest of Guetarnia in western Algeria. *Ekológia (Bratislava)*, 37(3), 289–300. DOI: 10.2478/eko-2018-0022.

Bezzel, E. (1985). *Kompendium der Vögel Mitteleuropas*. Nonpasseriformes. Wiesbaden: Aula-Verlag.

Bosch, J. (2003). Fenología y parámetros reproductivos del Aguililla Calzada *Hieraaetus pennatus* en Cataluña central (España). *Ardeola*, 50(2), 181–189.

Bosch, J., Borras, A. & Freixas J. (2005). Nesting habitat selection of booted eagle *Hieraaetus pennatus* in Central Catalonia. *Ardeola*, 52(2), 225–233.

Bosch, J. (2011). Population and breeding ecology of booted eagle in Catalonia. In I. Zuberogoitia & J.E. Martínez (Eds.), *Ecology and conservation of European forest-dwelling raptors* (pp. 87–92). Bilbao: Diputación Foral de Bizkaia.

Chevallier, D., Handrich, Y., Georges, J-Y., Baillon, F., Brossault, P., Aurouet, A., Le Maho, Y. & Massemin S. (2010). Influence of weather conditions on the flight of migrating Black storks. *Proc. R. Soc. Lond. B*, 277, 2755–2764. DOI: 10.1098/rspb.2010.0422.

Cramp, S. & Simmons K.E.L. (1980). *Handbook of the birds of Europe, the Middle East and North Africa*. Vol. 2. Hawks to Bustards. Oxford: Oxford University Press.

Deeming, D.C. (2002). Patterns and significance of egg turning. In D.C. Deeming (Ed.), *Avian incubation: behaviour, environment and evolution* (pp. 161–178). Oxford: Oxford University Press.

Del-Hoyo, J., Elliot, A & Sargatal J. (Eds.) (1994). *Handbook of the birds of the World*. Vol. 2. New World Vultures to Guineafowl. Spain: Lynx Edicions.

Díaz, J. (2005). La vida privada del águila calzada. *Quercus*, 227, 14–21.

Díaz, J. (2006). *El Águila Calzada y su conservación en la Comunidad de Madrid*. Fondo para la Investigación y Conservación de los Animales Salvajes y su Hábitat FICAS. Madrid.

Díaz, M., Martí, R., Gómez-Manzaneque, Á. & Sánchez A. (Eds.) (1994). *Atlas de las Aves nidificantes en Madrid*. Sociedad Española de Ornitología y Agencia de Medio Ambiente, Comunidad de Madrid.

Díaz-Ruiz, J. & Cebollada Baratas F. (2011). Monitoring and conservation of the booted eagle *Aquila pennata* in the Sierra de Guadarrama (central España). In I. Zuberogoitia & J.E. Martínez (Eds.), *Ecología y Conservación de las Rapaces Forestales Europeas* (pp. 93–98). Bilbao: Diputación Foral de Bizkaia.

Donázar, J.A., Blanco, G., Hiraldo, F., Soto-Largo, E. & Oria J. (2002). Effects of forestry and other land- use practices on the conservation of cinereous vultures. *Ecol. Appl.*, 12, 1445–1456. DOI: 10.1890/1051-0761.

García-Dios, I.S. & Viñuela J. (2000). Efectos de la gestión forestal sobre el éxito reproductor del Aguililla Calzada (*Hieraaetus pennatus*) en el valle del Tiétar. *Ardeola*, 47, 183–190.

García-Dios, I.S. (2003). Siblicide and cannibalism in the Booted Eagle (*Hieraaetus pennatus*) in the Tietar valley, Central Spain. *Journal of Raptor Research*, 37(3), 261.

García-Dios, I.S. (2014). Booted Eagle (*Hieraaetus pennatus*). In A. Salvador & M.B. Morales (Eds.), *Enciclopedia Virtual de los Vertebrados Españoles*. Madrid: Museo Nacional de Ciencias Naturales.

Garzón, J. (1974). Contribución al estudio del status, alimentación y protección de las falconiformes en España central. *Ardeola*, 19, 279–330.

Hiraldo, F. & González Grande J.L. (1987). *Las Rapaces Ibéricas*. Madrid: Centro de Fotografía de la Naturaleza.

Hoyt, D.F. (1979). Practical methods of estimating volume and fresh weight of bird eggs. *The Auk*, 96, 73–77. DOI: 10.1093/auk/96.1.73.

Iribarren, J.J. (1975). Biologia del Aguila calzada (*Hieraaetus pennatus*) durante el periodo de nidificacion en Navarra. *Ardeola*, 21, 305–320.

Iribarren, J.J. & Rodríguez A.A. (1988). Sobre la biología del Águila Calzada *Hieraaetus pennatus* (Gmelin, 1788) en Navarra. *Publicaciones de Biologia de la Universidad de Navarra, Serie Zoologicay*, 17, 3–27.

Isenmann, P. & Moali A. (2000). *Oiseaux d'Algérie*. Paris: SEOF.

Isenmann, P., Gaultier, T., El Hili, A., Azafzaf, H., Dlensi, H. & Smart M. (2005). *Birds of Tunisia*. Paris: Société d'Etudes Ornithologiques de France.

Jiménez-Franco, M.V., Martínez, J.E. & Calvo J.F. (2014). Patterns of nest reuse in forest raptors and their effects on reproductive output. *J. Zool.,* 292(1), 64–70. DOI: 10.1111/jzo.12085.

Marre, A., Quinif, Y., Lahondere, J.C. & Vila J.M. (1977). Le relief karstique et la géologie du versant Nord du Djebel Zouaoui (Constantine, Algérie). *Méditerranée*, 29(2), 45–54.

Martí, R. & Del-Moral J.C. (Eds.) (2003). *Atlas de las Aves Reproductoras de España. Dirección General de la Conservación de la Naturaleza*. Madrid : Sociedad España de Ornitología.

Martínez, J.E. & Sanchez-Zapata J.A. (1999). Invernada de aguililla calzada (*Hieraaetus pennatus*) y culebrera europea (*Circaetus gallicus*) en España. *Ardeola*, 46, 93–96.

Martínez, J.E., Pagán, I. & Calvo J.F. (2006a). Factors influencing territorial occupancy and reproductive output in the Booted Eagle (*Hieraaetus pennatus*). *Ibis*, 148(4), 807–819. DOI: 10.1111/j.1474-919X.2006.00586.x.

Martínez, J.E., Pagán, I. & Calvo J.F. (2006b). Interannual variations of reproductive parameters in a booted eagle (*Hieraaetus pennatus*) population: the influence of density and laying date. *Journal of Ornithology*, 147(4), 612–617. DOI: 10.1007/s10336-006-0086-0.

Martínez, J.E., Pagán, I., Palazón, J.A. & Calvo J.F. (2007). Habitat use of booted eagles (*Hieraaetus pennatus*) in a Special Protection Area: implications for conservation. *Biodivers. Conserv.*, 16, 3481–3488. DOI: 10.1007/s10531-006-9053-6.

Martínez, J.E., Pagán, I., Jiménez-Franco M.V. & Calvo J.F. (2011). Ecology of the booted eagle in semiarid Mediterranean landscapes. In I. Zuberogoitia & J.E. Martínez (Eds.), *Ecology and conservation of European forest-dwelling raptors* (pp. 226–233). Bilbao: Diputación Foral de Bizkaia.

Martínez, J.F. (2002). *Ecologíadel Águila Calzada (Hieraaetus pennatus) en ambientes mediterráneos*. Tesis Doctoral, Universidad de Murcia, Spain.

Newton, I. (1979). *Population ecology of raptors*. Berkhamsted: T. and A.D. Poyser.

Olendorff, R.S., Bibles, D.D., Dean, J.R., Haugh, J.R. & Kochert M.N. (1989). Raptor habitat management under the U.S. Bureau of Land Management Multiple–use Mandate. *Raptor Research Reports*, 8, 1–80.

Palomino, D. & Valls J. (2011*). Las rapaces forestales en España*. Población reproductora en 2009–2010 y método de censo. Madrid: Seo/BirdLife.

Perrins, C.H. (1987). *Birds of Britain and Europe*. London: HarperCollins.

Redmond, L.J., Murphy, M.T. & Dolan A.C. (2007). Nest reuse by Eastern Kingbirds: adaptive behavior or ecological constraint? *Condor*, 109(2), 463–468. DOI: 10.1650/0010-5422(2007)109[463:NRBEKA]2.0.CO;2.

Reid, J.M., Monaghan, P. & Nager R.G. (2002). *Incubation aviaire: écologie, évolution et énergétique (éd DCDeeming)* (pp. 314–325). Oxford: Oxford University Press.

Rutkowska, J. & Cichon M. (2005). Egg Size, Offspring Sex and Hatching Asynchrony in Zebra Finches Taeniopygia guttata. *J. Avian Biol.*, 36(1), 12–17. DOI: 10.1111/j.0908-8857.2005.03469.x.

Sánchez-Zapata, J.A., Sánchez, M.A., Calvo, J.F. & Esteve M.A. (1995). *Ecología de las Aves de Presa de la Región de Murcia*. Murcia: Servicio de Publicaciones de la Universidad de Murcia.

Sergio, F., Newton, I. & Marchesi L. (2005).Top predators and biodiversity. *Nature,* 436, 192–192. DOI: 10.1038/436192a.

Simberloff, D. (1998). Flagships, umbrellas and keystones: Is single-species management passe in the landscape era? *Biol. Conserv*., 83, 247–257. DOI: 10.1016/S0006-3207(97)00081-5.

Stokland, J.N & Amundsen T. (1988). Initial size hierarchy in broods of the shag: Relative significance of egg size and hatching asynchrony. *The Auk,* 105, 308–315. DOI:10.2307/4087495.

Suárez, S., Balbontín, J. & Ferrer M. (2000). Nesting habitat selection by booted eagles *Hieraaetus pennatus* and implications for management. *J. Appl. Ecol*., 37, 215–223. DOI: 10.1046/j.1365-2664.2000.00503.x.

Thévenot, M., Vernon, R. & Bergier P. (2003). *The birds of Morocco* (an annotated checklist). BOU.

Thomas, C.S. (1983). The relationship between breeding experience, egg volume and reproductive success of the kittiwake Rissa tridactyla. *Ibis,* 125, 567–574. DOI: 10.1111/j.1474-919X.1983.tb03151.x.

Urios, V., Escobar, J.V., Pardo, R. & Gómez J.A. (1991). *Atlas de las Aves nidificantes de la Comunidad Valenciana*. Valencia: Conselleria d´Agricultura i Pesca. Generalitat Valenciana.

Viada, C. & De-Pablo F. (2009). Cens d'aguila Calçada *Hieraaetus pennatus* a Balears al 2009 i estat de Conservacio). *Anuari Ornitologic de les Balears*, 24, 1−15.

Vleck, C.M. (1981a). Energetic cost of incubation in the Zebra Finch. *Condor*, 83, 229−237. DOI: 10.2307/1367313.

Vleck, C.M. (1981b). Hummingbird incubation : female attentiveness and egg temperature. *Oecologia*, 51(2), 199−205. DOI: 10.1007/BF00540601.

Zalles, J.I. & Bildstein K.L. (Eds.) (2000). *Raptor Watch: A global directory of raptor migration sites*. Birdlife Conservation Series No. 9. Cambridge: Birdlife International, Kempton: Hawk Mountain Sanctuary.

Zuberogoitia, I., Castillo, I., Zabala, J., Iraeta, A. & Azkona A. (2011). Population trends of diurnal forest raptors in Biscay. In I. Zuberogoitia & J.E. Martínez (Eds.), *Ecology and conservation of European forest-dwelling raptors* (pp. 70–80). Bilbao: Diputación Foral de Bizkaia. DOI: 10.13140/2.1.2204.5121.

Ekológia (Bratislava) Vol. 38, No. 4, p. 392–400, 2019
DOI:10.2478/eko-2019-0029

MODEL OF THE REPRESENTATIVE GEOECOSYSTEM AT REGIONAL LEVEL

ZITA IZAKOVIČOVÁ, MILENA MOYZEOVÁ

Institute of Landscape Ecology, Slovak Academy of Sciences, Štefánikova 3, P.O.Box 254, 814 99 Bratislava, Slovak Republic; e-mail: zita.izakovicova@savba.sk, milena.moyzeova@savba.sk

Abstract

Izakovičová Z., Moyzeová M.: Model of the representative geoecosystem at regional level. Ekológia (Bratislava), Vol. 38, No. 4, p. 392–400, 2019.

The paper is aimed at the presentation of the new conception of nature protection. The new conception is based on the creation and protection of the representative geoecosystems. The strategic aim of defining the representative geo-ecosystem is: to determine a representative geo-ecosystem for each territorial unit on the given hierarchical level - the regional principle, to determine a representative occurrence for each type of the geo-ecosystem – the typological principle. The list of types of representative geo-ecosystems should serve as an ecologically based systematic framework for new protected areas (according to the analysis of unsufficiently protected representative geo-ecosystems) designations, as well as for methodical proposals of biocentres of the territorial system of ecological stability. The new approach to nature conservation is based not only on the protection of life forms but also on the protection of living conditions.
The paper presents an example of the creation of representative geoecosystems on the regional level –Trnava region.

Key words: representative geoecosystem, region Trnava, territorial system of ecological stability, integrated approach.

Introduction

Healthy ecosystems are the basis of life and are essential to human well-being and societal prosperity. They provide direct and indirect benefits, including food, clean air and fresh water, shelter and medicine, and they mitigate natural disasters, pests and diseases and contribute to climate regulation. Despite the irreplaceable significance of ecosystems for landscape, they are continually threatened and degraded. According to MEA, 60% of the world‘s ecosystems are degraded or used unsustainably; 75% of fish stocks are over-exploited or significantly depleted and 75% of the genetic diversity of agricultural crops has been lost worldwide since 1990. Deterioration and loss in ecosystems jeopardise provision of these services. We lose species, habitats and the wealth and employment derived from nature, and we endanger our own well-being (EC, 2011).

Pressures on the use of ecosystems and their services have increased notably in recent years. This is accompanied by increased exploitation, especially emanating from demographic change, economic development, a consumer lifestyle, and changing societal behaviour. Increased demand for ecosystem products, including food, raw materials and derived energy, and subsequent land use changes and supply of regulating and supporting services will assume the same or even greater importance in the future. Ecosystem conservation, protection of its functions and efficient use of ES are indispensable for sustainable development and for the preservation of human society. Several initiatives have been taken to address the protection of ecosystems and their services, whether on the professional or political scene (Burkhard et al., 2014; De Groot et al., 2010; Constanza, 2016; Haines-Young, Potschin, 2012; etc.).

The aim of the modern concept of nature and landscape protection of the landscape also in Slovakia is to provide a framework for a prosperous and sustainable society through the protection, rehabilitation, development and sustainable use of ecosystems and their services. In addition to the traditional objectives of protection and conservation of natural heritage for aesthetic, cultural and scientific reasons, new trends in nature conservation also focus on utilitarian values of nature (such as ecological stabilization, self-regulation, conservation of production capacity, gene pool utilization, protective, hygiene and recreational effects). Thus, it is not only about the protection of individual ecosystems, but also about ensuring the efficient use of their ecosystem services (Izakovičová et al., 2017).

The aim of this approach is to maintain suitable living conditions for man as a biological, social and economic species (anthropocentrism), but also suitable conditions for life of other living systems - ecosystems (biocentrism). The current level of knowledge unequivocally confirms that the conditions of human life are sustainable only if the living conditions for all other species are maintained, which requires preserving the natural functions and relationships of all - including the abiotic - elements of the landscape (geobiocentrism). That is why all science concepts supporting such ecological policy focus on developing principles for preserving natural functions and relationships in geoecosystems.

Thus, the modern concept of nature protection is based on the model of protection of representative geoecosystems (REPGES). REPGES represent landscape units characterized by a certain diversity of conditions - different bedrock, hydrological, pedological, climatic, morphometric conditions, which also cause the occurrence of various forms of ecosystems and biota i.e. life forms. These land units represent a potential for the development of society.

The aim of the paper is to introduce the concept of protection of representative geoecosystems, which represent the basic spatial units for the provision of ecosystem services. We present the methodical procedure at the regional level on example of the Trnava district.

Theory and methodology

Specific objects and carriers of geoecodiversity are geoecosystems. For most geographers and geosystemologists, the term geosystem is a complex expression of the reality of the landscape. However, for many other natural sciences as well as for the non-professional public, the "geo-" prefix evokes association only with the abiotic part of the landscape, and often only with the bedrock. Therefore, for the avoidance of these doubts, we also accept the possibility of using the "geoeco-" prefix in various terms that emphasize both aspects of the indivisible reality of the landscape, both abiotic conditions and natural systems (Miklós et al., 2006).

The starting point of the strategy of preserving the diversity of conditions and life forms (geoecodiversity) in the above mentioned sense is identifying and defining those geoecosystems that we want to preserve (Odum, 1975; Bunce et al., 1996; Jongman et al., 2006). Thus, the strategic objective of defining representative geoecosystems is

(Miklós et al., 2006):

- identify a representative geoecosystem for each territorial unit at a given hierarchical level - regional principle,
- determine a representative occurrence for each type of geoecosystem - a typological principle.

In other words:

- each territorial unit must have identified a representative geoecosystem, as well as,
- each type of geoecosystem is representative somewhere (there is a territorial unit where the type has a representative occurrence).

The strategic outcome of this procedure is that representative geoecosystems in each region should be declared as skeleton elements of the territorial system of ecological stability (TSES), i.e. according to our legislation for biocentres or biocorridors and interaction elements. Despite the fact that the nature protection in Slovakia is on a high level on a worldwide scale, in terms of the number, size and quality of protected areas, after elaborating the system scheme of representative geoecosystems and comparing it with protected areas, we can state that many representative geoecosystems are not protected and are in danger of being wiped out. Overall, the conservation of ‚attractive' types of REPGES, such as the most interesting rare habitats, exotics, aesthetically impressive communities (e.g. rock communities, karst habitats, alpine habitats) still prevails.

For Slovakia, a system of representative potential geoecosystems (REPGES) was developed at a supra-regional level (Miklós et al., 2006). The aim was to develop a system scheme for the strategy of protecting the diversity of conditions and life forms at the state level, in other words to elaborate a list containing all strategically important geoecosystems of the SR. Subsequently, a methodological procedure for specification of REPGES at regional level was elaborated on the example of Trnava district.

The individual types of REPGES SR were determined on the basis of (Miklós et al., 2006):

- Zonal (bioclimatic) conditions - in the landscape they are mainly expressed by vegetation zones. We characterize them according to bioclimatic conditions, which are comprehensively expressed in 9 zones of potential vegetation.
- Azonal conditions - primarily by quaternary-geological base and relief, secondary by soils and groundwater level. We divided them into 37 types.

In a real landscape, these conditions manifest themselves **comprehensively**, they cannot be separated. Zonal conditions in the region cannot be changed at all, while azonal - soil, water and relief - can be partially changed or influenced by the use of energy.

The methodology of creation of REPGES at regional level was also based on this methodology. REPGES at regional level have been designated on the basis of synthesis:

a) *abiocomplexes* - the basis for the processing of abiocomplexes was a digital spatial database of **abiocomplexes** processed at Slovak Geological Institute Dionýz Štúr within the project Geological maps for the needs of the landscape-ecological basis of integrated landscape management (KEZIMK) with a detail corresponding to the scale of 1:50 000. The database contains spatially delimited (based on 1:10 000) topical synthetic units - (abiocomplexes). The units are described by the following set of parameters relevant to REPGES processing:

- relief: morphographic-morphometric-positional type of relief, average slope,
- geological basis - substrate: lithogenetic characteristics of the substrate, hydrogeological characteristics of the substrate (+ relevant state variables), engineering geological zone (+ relevant state variables), genetic types of quaternary sediments,
- soil: soil subtype, soil type, skeleton, depth, etc.
- climate type,
- groundwater regime for ABK.

Processed parameters were supplemented and revised (simplified, supplemented, reclassified), for the purposes of REPGES creation.

b) *map of potential vegetation* - represents the vegetation that would have developed in the territory if man had not affected the landscape. It is important to know which natural vegetation units were present in the area and how they were spatially distributed. The characterization of potential vegetation was done on the basis of internal data of the Institute of Geobotany of SAS according to the concept of Michalko et al. (1986).

Study area

Trnava district was chosen as a study area. The administrative boundary of the Trnava district was the basis of the delimitation of the study area. The district is located in the western part of Slovakia. It is part of the Trnava region.

It is administratively made up of 45 rural municipalities and 1 city of Trnava, which also serves as a regional town. With an area of 741 km^2, it belongs to medium-sized districts of Slovakia. Trnava is a typical industrial-agricultural region. According to the last population census, 126 382 inhabitants live in the territory.

From the geomorphological point of view, the area consists of two basic geomorphological units - the Danubian Lowland (part of the Trnava Upland and the Danubian Flat) and the Little Carpathians. The core of the territory, i.e. the central and southern part is the Danubian Lowland. The Little Carpathians form the northwestern border of the territory. Geomorphological conditions also determine the characteristics of other landscape-forming components and the occurrence of natural resources and potentials of the territory, which determine the functional use of the territory. The natural resources in the area are dominated by quality soils (chernozem and floodplain soils), which with favorable climate conditions, create a high potential for agricultural development. Agricultural soil of the study area belongs to the best quality and most fertile soils.

There is a high forestry potential in the northern part of the study area. Some of the forests are used for forest management, part of them are protective forests. The forest ecosystems of the study area are characterized by high gene pool and nature conservation value. Most of the forest units of the study area form part of the Little Carpathians Protected Landscape Area (PLA). In the study area there are also numerous protected areas (16 sites) and 4 sites of the *Natura 2000 network.*

From the environmental aspect, it represents a region with specific environmental problems resulting from the development of industry and agriculture (strong degree of contamination of individual components of the environment, degradation processes of ALF due to incorrect land management, inappropriate crop structure, etc.). It is highly anthropized landscape with a very low degree of ecological stability.

Results

In total, 95 basic types of REPGES were designated in the Trnava district, resulting from the synthesis of abiocomplexes and types of potential vegetation. Altogether 26 types of abiocomplexes were allocated in four basic categories (plains, uplands, highlands, mountainous areas) and 11 types of potential vegetation determined according to bioclimatic conditions and 4 types determined according to azonal conditions. Individual types of REPGES have different areal representation. The largest share is occupied by the following REPGES (Izakovičová et al., 2011):

- undulating loess plains with oak forests – 16.75%,
- medium rugged loess upland with oak-hornbeam Carpathian forests – 9.87%,
- undulating loess plains with thermophilic Pontic-Pannonian oak forests – 7.38%,
- flat plateaus of loess uphills with oak-hornbeam Carpathian forests – 7.14%,
- medium rugged loess uphills with oak-hornbeam Pannonian forests – 5.47%,
- undulating plains of sand dunes with oak forests – 5.20%.

For 14 REPGES, the area ranges from 1 to 5%. Areas of other REPGES (up to 70 types) do not reach even 1% of the total area of the district.

During the historical development, the man significantly influenced the structure of the REPGES of the study area, which was manifested mainly by the significant occupation of forest ecosystems (deforestation) and their transformation into agricultural, especially arable land, first into small-block and then large-block arable land. Significant intervention into the structure of REPGES had the intensification of agriculture, which occurred at the time of collectivization and socialization. This meant deforestation and plowing of land. The gradual increase in heavy machinery has led to the elimination of vegetation residues, creating a deforested, ecologically unstable, monotonous, intensively agriculturally exploited landscape. Significant disturbance and significant transformation of REPGES is also evidenced by the naturalness coefficient, which reaches zero value in most of the area of the Trnava loess table, which means that all the natural ecosystems were destroyed and converted to semi-natural or artificial ecosystems. The assessment of the conservation/anthropogenic change of individual REPGES types was carried out on the basis of the naturalness coefficient (Miklós et al., 2006), which expresses the current percentage of natural vegetation within each REPGES type. Up to 17 REPGES types have no natural vegetation and up to 27% of REPGES have no natural vegetation percentage greater than 1%. 30% of REPGES types have abundance of natural vegetation over 50%. The structure of abundance of REPGES types on the basis of naturaless coefficient is expressed in Fig. 1.

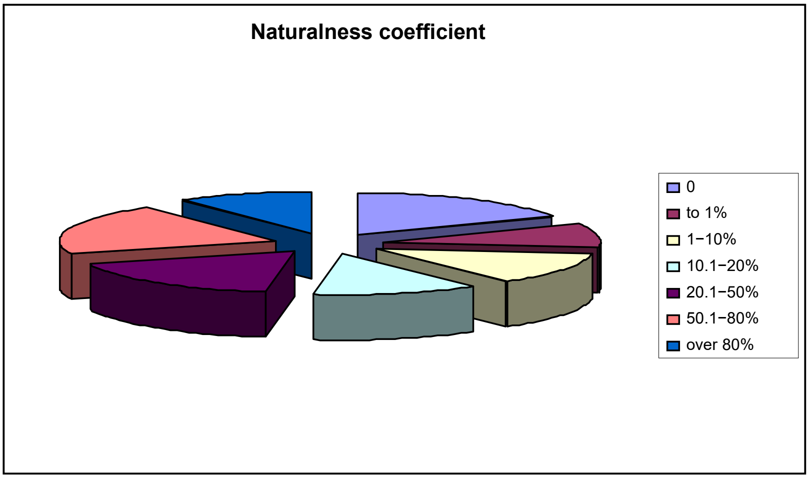


Fig. 1. Naturalness coefficient.

Even now, individual types of REPGES are threatened by the development of human activities. The assessment of the threat of individual REPGES due to stress factors was carried out on the basis of the threat coefficient, which reflects the occurrence of the stressor in the REPGES area and the intensity of its negative effect. 46% of REPGES types are medium endangered, 38.5% are severely endangered and 15.5% are critically endangered, based on this coefficient. REPGES located in the central part of the district on loess tables and loess uplands are critically endangered; REPGES of the highlands and rocks located in the northern part of the district are the least endangered. Most of them are part of the Little Carpathians PLA.

In order to preserve individual types of REPGES, it is necessary to know their degree of protection. Protection of REPGES was evaluated on the basis of the protection coefficient (Miklós et al., 2006), which expresses the proportion of the area of REPGES that falls under the 2nd to 5th degree of protection. The 17 types of REPGES of the study area (17.7%) show a zero degree of protection. It is the most extensive REPGES of loess tables and loess uplands. The highest level of protection has REPGES No. 84: *strongly rugged highland on variegated* *rocks with acidophilous oak forests*, of which up to 87.2% of the area is protected and REPGES No. 88: and No. 89:, where the share of protection is almost 65%. However, their areas are relatively small. Area of REPGES No. 84: *strongly rugged highland on variegated rocks with acidophilous oak forests* reaches 12.89 ha, which in percentage represents only 0.02% of the total area of the district and the area of REPGES no. 88: *medium rugged karst lower upland with calciphile beech forests* and 89 *medium rugged karst lower upland with beech and fir-beech flowery forests* together with about 1000 ha, which represents only 1.35% of the total area of the district. The highest number of REPGES shows a degree of protection in the range of 40–60%, which includes up to 54 types of REPGES (56.25%). The categorization of REPGES according to the protection coefficient is shown in Fig. 2.

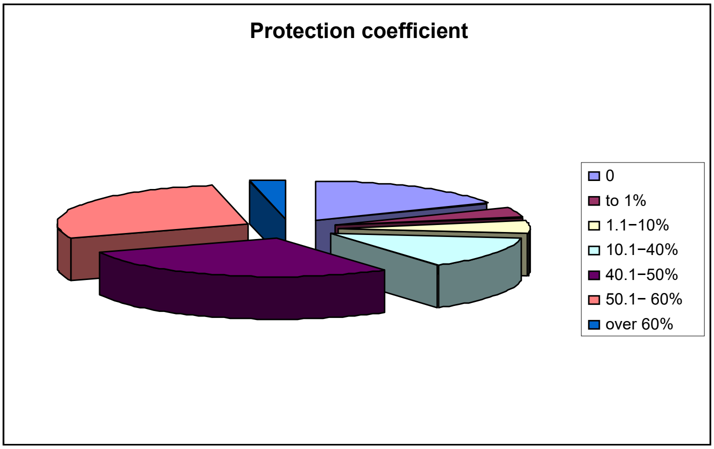


Fig. 2. Protection coefficient.

Discussion

The identification and characterization of the types of representative geoecosystems should serve as a strategic scheme for a systemically built plan for the preservation of representative conditions and forms of life in a given territory. From a practical point of view, it should serve as an ecologically founded systemic basis for designing new protected areas (by analyzing which representative geoecosystems are at risk and what types of representative geoecosystems are inadequately protected), strengthening the management of nature conservation, as well as the systematic designing biocentres and biocorridors of supra-regional and regional level within the creation of the territorial system of ecological stability (TSES) (Miklós et al., 2018). The basis of biocentres should be made by the individual REPGES and the TSES should be designed in such a way as to ensure maximum representativeness and protection of individual ecosystems. According to the modern concept of nature protection, the basis of nature protection is not only the protection of individual life forms but also the protection of their living conditions (Miklós et al., 2006). The concept of ecological networks in the Slovak Republic, known as the territorial system of ecological stability, was also developed on this basis. This, unlike the global concept of ecological networks (Jongman, 1996; Jongman, Kristiansen, 2001; Forman, 1990; Nowicki et al., 1996; Buček, Lacina, 2000; Reháčková, Pauditšová, 2003; Fábos, Ryan, 2004; Izakovičová, Świąder, 2017) does not focus only on protection of life forms, but also on the protection of their conditions. Concept of TSES changes the "classic" idea of the nature conservation based on the division of the landscape to protected and non-protected areas towards a system which maintains the ecological stability of the whole territory by an ecologically suitable spatial structure of the landscape even in the case that it is exploited in different – even in intensive – way. Thus the concept of the TSES is an important tool to secure spatial stability of landscape (Izakovičová et al., 2000).

The TSES is a concise method based on landscape ecological research which modified the ideas of ecological networks towards integrated managment of optimal organisation and utilisation of the landscape as a whole.

The procedure of the TSES contains the basic actions as follows (Izakovičová et al., 2000).

- Delineation of main elements of the TSES: biocentres, biocorridors and interactive elements. Those elements compose the frame of an ecological network. As biocentres should be delineated those biotopes which serves as the basis for food, shelter and site for reproduction, as the biocorridors and interactive elements should be projected chains of biotopes which brake the isolation and ensure the migration and interaction as well as the spatial ecological stability of the landscape.
- Definition and proposal of so called eco-stabilising measures, which should fulfil different practical ecological function as soil and water protection, microclimatic, hygienic, aesthetic and other functions. Among those the agro-technical, agro-ameliorative and forest management measures might be underlined.

At the same time, REPGES are the landscape's basic operational units for assessing ecosystem services. The ES concept focuses on comprehensive research into ecosystems, their functions and the assessment of the benefits that individual ecosystems can provide to society. It is built on an interdisciplinary basis. It seeks to ensure the protection and efficient use of ecosystems and their services so that all ES are used in a harmonious way and that one ES does not develop at the expense of others. Several methods have been developed to assess the ES, whether monetary or non-monetary, participatory, biophysical, etc. The fundamental idea of the ES concept is therefore the usefulness and benefits of nature for society and human well-being. Ecosystem services can very easily be defined as the contribution of ecosystems (living systems) to human well-being. These services are final, as they are the outputs of ecosystems (whether natural, semi-natural or largely altered) that directly affect people's well-being. Their basic attribute is that they retain a link to the associated ecosystem functions, processes and ecosystem structure itself that co-creates them (Haines-Young, Potschin, 2012). Different spatial units are used to assess ecosystem services. The most commonly used and simplest spatial units are land use elements. However, these do not take into account the complex conditions of the landscape system and therefore REPGES may represent new spatial units for the assessment of ecosystem services.

Conclusion

The characteristics of the types of representative geoecosystems should serve as an ecologically founded systemic basis for the designing new protected areas as well as a primary basis for the creating functional territorial systems of ecological stability. As already mentioned, representative geoecosystems in each region should be declared as elements of the skeleton of the territorial system of ecological stability; i.e. according to our legislation: biocentres, biocorridors, interaction elements, according to foreign literature: core areas (Jongman, 1996; Nowicki et al., 1996).

The proposals of TSES should also focus on strengthening the overall spatial stability of individual REPGES and geoecological regions. When selecting elements of TSES, preference should be given to sites with a natural species composition, in the absence of these, it should be proposed to create new sites with natural vegetation so that all types of REPGES are represented and that a representative type of geoecosystem is preserved in each region.

In the Trnava district, there is no element of supra-regional or regional character in up to 21 types of REPGES. Supra-regional elements of TSES are represented in 67 (69.7%) types of REPGES, regional in 64 (66.7%) types of REPGES (overlap of regional and supra-regional elements of TSES was found in 57 types of REPGES).

From the point of view of preservation and protection of REPGES it is necessary to ensure protection of those REPGES, where elements of TSES of different hierarchical order are located. For those REPGES (21 types) where the TSES elements are not currently identified, it is necessary to create them and subsequently to provide them with the necessary protection. It is also necessary to establish a set of ecostabilizing measures that will not only protect individual types of ecosystems, but also ensure the efficient use of their ecosystem services.

Acknowledgements

„This publication was produced through support under the Operational Program Research and Innovation for the project: Assessment of landscape changes and their impact on the environment, NFP313010X649, co-financed by the European Fund regional development ." We are grateful to Mgr. Silvia Chasníková for English proofreading.

References

Buček, A. & Lacina J. (2000). *Geobiocenologie II.* Brno: MZLU.

Bunce, R.G.H., Barr, C.J., Clarke, R.T., Howard, D.C. & Lane A.M.J. (1996). Land classification for strategic ecological survey. *J. Environ. Manag.*, 47, 37–60. DOI: 10.1006/jema.1996.0034.

Burkhard, B., Kandziora, M., Hou, Y. & Müller F. (2014). Ecosystem service potentials, flows and demand – concepts for spatial localisation, indication and quantification. *Landscape Online*, 34, 1–32. DOI: 10.3097/LO.201434.

Costanza, R. (2016). Ecosystem services and theory and practice. In M. Potschin, R. Haines-Young, R. Fish & R. Kerry Turner (Eds.), *Routledge handbook of ecosystem services* (pp. 15–24). London, New York: Routledge.

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, 260–272. DOI: 10.1016/j.ecocom.2009.10.006.

European Commission (2011). *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. http://ec.europa.eu/environment/nature/biodiversity/comm2006/pdf/2020/1_EN_ACT_part1_v7[1].pdf

Fábos, J.Gy. & Ryan R.L. (2004). International greenway planning: an introduction. *Landsc. Urban Plann.*, 68(2–3), 143–146. DOI: 10.1016/S0169-2046(03)00155-5.

Forman, R.T.T. (1990). Ecologically sustainable landscapes: The role of spatial configuration. In I.S. Zonneveld & R.T.T. Forman (Eds.), *Changing landscapes: An ecological perspectives* (pp. 261–278). New York: Springer Verlag.

Haines-Young, R. & Potschin M. (2012). Common International Classification of Ecosystem Services (CICES): 1035 Consultation on Version 4, August-December 2012. EEA Framework Contract No EEA/IEA/09/003.

Izakovičová, Z., Hrnčiarová, T., Miklós, L., Tremboš, P., Ružičková, J., Liška, M., Králik, J., Moyzeová, M., Šíbl, J. & Pauditšová E. (2000): *Metodické pokyny na vypracovanie projektov regionálních ÚSES a miestnych ÚSES*. Bratislava: Združenie Krajina 21.

Izakovičová, Z., Miklós, L., Moyzeová, M., Špilárová, I., Kočický, D., Halada, Ľ., Gajdoš, P., Špulerová, J., Baránková, Z., Štefunková, D., Kenderessy, P., Šatalová, B., Dobrovodská, M., Hrnčiarová, T., David, S. & Krnáčová Z. (2011). *Model reprezentatívnych geoekosystémov na regionálnej úrovni*. Bratislava: ÚKE SAV.

Izakovičová, Z., Bezák, P., Mederly, P. & Špulerová J. (2017). Uplatňovanie konceptu ekosystémových služieb v plánovacej a riadiacej praxi v Slovenskej republike - výsledky projektu OpenNESS na prípadovej štúdii Trnava. *Životné Prostredie*, 51(4), 198–204.

Izakovičová, Z. & Świąder M. (2017). Building ecological networks in Slovakia and Poland. *Ekológia (Bratislava)*, 36(4), 303–322. DOI: 10.1515/eko-2017-0025.

Jongman, R.H.G. (1996). Research priorities: scientific concepts and criteria. In Perspectives on ecological networks. European Centre for Nature Conservation. *Man and Nature*, 1(14), 151–160.

Jongman, R.H.G. & Kristiansen I. (2001). *National and regional approaches for ecological networks in Europe*. Nature and Environment, 110. Strasbourg: Council of Europe Publishing.

Jongman, R.H.G., Bunce, R.G.H., Metzger, M.J., Mücher, C.A., Howard, D.C. & Mateus V.L. (2006). Objectives and applications of a statistical environmental stratification of Europe. *Landsc. Ecol.*, 21, 409–419. DOI: 10.1007/s10980-005-6428-0.

Michalko, J., Berta, J. & Magic D. (1986). *Geobotanická mapa ČSSR*. Bratislava: Veda, vydavateľstvo SAV.

Miklós, L., Izakovičová, Z., Boltižiar, M., Diviaková, A., Grotkovská, L., Hrnčiarová, T., Imrichová, Z., Kočická, E., Kočický, D., Kenderessy, P., Mojses, M., Moyzeová, M., Petrovič, F., Špinerová, A., Špulerová, J., Štefunková, D., Valkovcová, Z. & Zvara I. (2006). *Atlas reprezentatívnych geoekosystémov Slovenska*. Bratislava: ÚKE SAV, MŽP SR, Banská Štiavnica: Esprit, s.r.o.

Miklós, L., Diviaková, A. & Izakovičová Z. (2018). *Ecological networks and territorial systems of ecological stability*. Cham: Springer.

Nowicki, P., Bennet, G., Middleton, D., Rientjes, S. & Wolters R. (Eds.) (1996). Perspectives on ecological networks. European Centre for Nature Conservation. *Man and Nature*, 1(14), 161–192.

Odum, E.P. (1975): *Ecology: The link between the natural and the social sciences*. London-New York-Sidney-Toronto: Holt Rinehart and Winston.

Reháčková, T. & Pauditšová E. (2003). Ecological networks in urban areas - new approaches. *Ekológia (Bratislava)*, 22(2), 108–118.