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TROPHIC STATUS OF BLIDINJE LAKE (BOSNIA AND HERZEGOVINA) BASED ON THE DETERMINATION OF THE TROPHIC STATE INDEX (TSI)

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Abstract

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Blidinje lake is an integral part of the Nature Park Blidinje established in 1995. Blidinje lake is largest mountain lake in Bosnia and Herzegovina (B&H) according to the surface and water supplies. Considering the surface, it is surprising to its low depth and large surface changes.
Eutrophication is defined as the 'biological effect of increasing concentrations of plant nutrients in aquatic ecosystems'. Eutrophication results in increased primary production or the production of aquatic plants. It can adversely affect the suitability of the use of water resources for other purposes. Metabolism of shallow lakes is extremely specific such as classical connections; algal biomass with a load of nutrients in shallow lakes can have catastrophic effects.
Nutrients that come directly or indirectly into water lead to increase in the amount of algae as well as changes in the composition of phytoplankton, where there is suppression of diatoms and green algae by cyanobacteria.
The values of parameters assessing trophic index (Trophic State Index [TSI]) are total phosphorus, chlorophyll a and transparency. This method that generalises numerous data using mathematical equations allows the expression of trophic level.
On the basis of earlier studies on physical and chemical parameters of water quality in Blidinje lake, Carlson trophic index was calculated, and based on that assessment, trophic level is given.

Key words: eutrophication, trophic index, total phosphorus, transparency, chl-a.

Introduction

The Blidinje lake is an integral part of Blidinje Nature Park founded in 1995. It is the largest mountain lake in Bosnia and Herzegovina (B&H) (Spahić, 2001) and occupies 364 km^2 of the surface (Šimunović, Bognar, 2005). The main factors influencing the formation of lakes are devastation of forests and anthropogenic flooding of the abyss.

The lake is under strong influence of meteorological conditions. The amount of water in the lake depends on the amount of rainfall and the rate of water sinking (Musa, 2005). The lake is very shallow with an average depth of 0.3–1.9 m. The ratio of lake surface during high water and surface lakes during low water level is 3.6:2.5 km^2. The transparency of water is 15 cm, because a large amount of suspended particles blurs water (Spahić, 2001).

Blidinje Nature Park is under the influence of the mountain climate, with fresh summers lasting from early June to September. Above 1,700 m above sea level, there are parts with 'everlasting snow'. Precipitation is most prevalent in the fall and early spring, and the mean annual temperatures vary and range from 1.2 to 11.2 °C (Soldo, 2005).

Eutrophication is the biological effect of increasing nutrient concentrations in aqueous ecosystems (Harper, 1992). By increasing the concentration of nutrients, it increases the primary production, that is, the development of phytoplankton. Increased phytoplankton development leads to an increase in the amount of organic substance that is bactericidal, oxygen concentration reduction and increased release of sedimentary nutrients. This nutrient increase allows algae production, resulting in reduced transparency and water staining. This process can be natural or anthropogenic (Welch, 1980).

Nutrients that come directly or indirectly into water lead to an increase in the amount of algae, that is, to increase the amount of cyanobacteria. Carlson (1977) uses algal biomass as the basis for determining the trophic state. The amount of chlorophyll, the transparency and the total phosphorus are three variables that affect each other and algae biomass independently.

On the basis of earlier researches on physicochemical water quality indicators of Blidinje lake and Carlson's TSI, Blidinje lake is classified into certain categories of trophy, as opposed to the OECD (1982) classification which provides only a descriptive estimate of the level of trophy (US EPA, 2000).

Carlson's Trophic State Index (Tidal Index – TSI)

The trophic state is defined as the total mass of living biological material (biomass) in the aquatic area at a given site at a given time. The trophic state arises in response to an increased amount of nutrients (Naumann, 1929). And the amount of nutrients can affect the annual life time, drinking water and mixing water.

Carlson (1977) uses algae biomass as the basis for determining the trophic state. The amount of chlorophyll, the depth and the total phosphorus are three variables that affect algal biomass and thus affect the determination of the trophic state of any aqueous area. Chlorophyll is preferred because it is the most accurate predictor of algal biomass. According to Carlson (1977), total phosphorus can be a better indicator of chlorophyll for predicting the summer trophic state than winter patterns, and transparency should only be used if no other method is available.

The index can be easily calculated and used. Carlson (1977) has formed three equations that mathematically calculate the degree of trophy:

$$\mathrm{TSI\ (SD)} = 10 \times \left(6 - \frac{\ln SD}{\ln 2}\right),$$

where SD is the secchi depth in metres (m)

$$TSI\ (Chl\text{-}a) = 10 \times \left(6 - \frac{2.04 - 0.68 \ln Chl\text{-}a}{\ln 2}\right),$$

where Chl-a is the concentration of chlorophyll a in mg/m^3

$$TSI\ (TP) = 10 \times \left(6 - \frac{\ln \frac{48}{TP}}{\ln 2}\right),$$

where TP is the concentration of total phosphorus in mg/m^3.

Logarithmic data transformation allows the use of statistical parameters (mean, standard deviation and parametric comparison of tests). This makes it easier not only to compare and reduce data but also to communicate because the user does not have to use graphs with logarithmic sections (Carlson, 1977).

In addition to Carlson's equation (1977), he compiled a systematic table according to which the index range is 0–100, although the index does not theoretically have lower and upper bounds.

Carlson defines the trophic index using every doubling of algae biomass as a benchmark for dividing each state, that is, every time the biomass concentration doubles because of a basic scale, a new trophic state emerges. Owing to the reciprocal relationship between the biomass concentration and the transparency measured by the Secchi disk, any duplication of biomass results in a split of transparency values. The zero dot in the table corresponds to the value greater than ever measured by the Secchi disc. The largest transparency of 41.6 m was measured in lake Masyuko in Japan. The next largest integer on the log 2 rank is 64. Then the transparency equation was formed (Table 1).

The scale is presented numerically, not nomenclature. A small number of nomenclature categories leads to loss of information, because a large number of lakes are classified together and susceptible to a change in the trophic state. This scale classifies lakes into more than 100 categories, giving more accurate information on the trophy lake condition (Carlson, 1977).

The trophy index gains value when it can be related to certain events in the water area as shown in the table. Lakes with TSI values of less than 40 are usually classified as oligotrophic, 40–50 as mesotrophic and 50–70 as eutrophic, and if the value is greater than 70, then it is hypertrophic lake. Table 2 shows the classification of the lake according to Carlson's trophic index and the characteristics of a particular category (Carlson 1983, Carlson, Simpson, 1996).

Table 1. TSI and its parameters (Carson 1977).

TSI	Secchi disk (SD)	TP	Chl-a
0	64	0.75	0.04
10	32	1.5	0.12
20	16	3	0.34
30	8	6	0.94
40	4	12	2.6
50	2	24	6.4
60	1	48	20
70	0.5	96	56
80	0.25	192	154
90	0.125	384	427
100	0.064	768	1183

The main advantage of the TSI is that the relationship between the variables can be used to identify certain conditions

Table 2. Lake classification according to Carlson's trophic state index (Carlson, Simpson, 1996).

TSI	Attributes	Water Supply
<30	**Oligotrophy:** Clear water, oxygen throughout the year in the hypolimnion	Water may be suitable for an unfiltered water supply
30–40	Hypolimnia of shallower lakes may become anoxic	
40–50	**Mesotrophy:** Water moderately clear; increasing probability of hypolimnetic anoxia during summer	Iron, manganese, taste, and odour problems worsen. Raw water turbidity requires filtration
50–60	**Eutrophy:** Anoxic hypolimnia, macrophyte problems possible	
60–70	Blue-green algae dominate, algal scums and macrophyte problems	Episodes of severe taste and odour possible
70–80	**Hypereutrophy:** (light limited productivity). Dense algae and macrophytes	
>80	**Oligotrophy:** Clear water, oxygen throughout the year in the hypolimnion	Water may be suitable for an unfiltered water supply

Table 3. Interpretations of deviations of the index values (Carlson, 1983).

Relationship between TSI Variables	Conditions
TSI(Chl) = TSI(TP) = TSI(SD)	Algae dominate light attenuation; TN/TP = 33:1
TSI(Chl) > TSI(SD)	Large particulates, such as *Aphanizomenon* flakes, dominate
TSI(TP) = TSI(SD) > TSI(Chl)	Non-algal particulates or colour dominate light attenuation
TSI(SD) = TSI(Chl) > TSI(TP)	Phosphorus limits algal biomass (TN/TP > 33:1)
TSI(TP) >TSI(Chl) = TSI(SD)	Algae dominate light attenuation but some factor such as nitrogen limitation, zooplankton grazing or toxics limit algal biomass

in the lake that limit algal biomass growth or affect the measured variables. As the relationships between variables derived from regression relationships and correlations are not perfect, the variability between the index values can be expected. However, in some situations, these variations are not random and the factors that interfere with this empirical relationship can be identified (Carlson, 1981). Table 3 gives some possible interpretations of the index deviation (Carlson, 1983).

Material and methods

Determination of sampling locations

Surface water samples were taken at five stations located along the lakeshore (Fig. 1). Surface water sampling was the appropriate method, given the fact that water in shallow lakes is often mixed and there is no vertical stratification (Rocha et al., 2009). Water samples are taken two weeks in the period May to November in 2008. Some parameters have not been monitored for some months (Ivanković et al., 2011).

All water quality parameters are determined in accordance with the APHA methodology (APHA, 1998). Accuracy of each method is prescribed by the APHA methodology. Dense depth (using Secchi disk) was measured directly at the sampling points. Ultraviolet–visible (Shimatzu) was used for analysing phosphate and total phosphorus content (λ = 690 nm). The chlorophyll content was determined by a fluorimetric method (TURNER TD-700, Sunnyvale, CA) at wavelength of 365 nm using 90% acetone as extraction solvent (Jeffrey et al., 1997).

Data analysis

From the data for chlorophyll, the transparency and total phosphorus according to the research by Ivanković (2010) determined the trophic status of Blidinje lake. The trophic state index (TSI) is obtained by incorporating data for these three parameters into the corresponding Carlson (1977) equation. The obtained results enabled the determination of Blidinje lake trophy category based on the criteria given in Tables 1 and 2. (Carlson, 1977). A certain relationship between individual parameters was observed using Table 3 (Carlson, 1983).

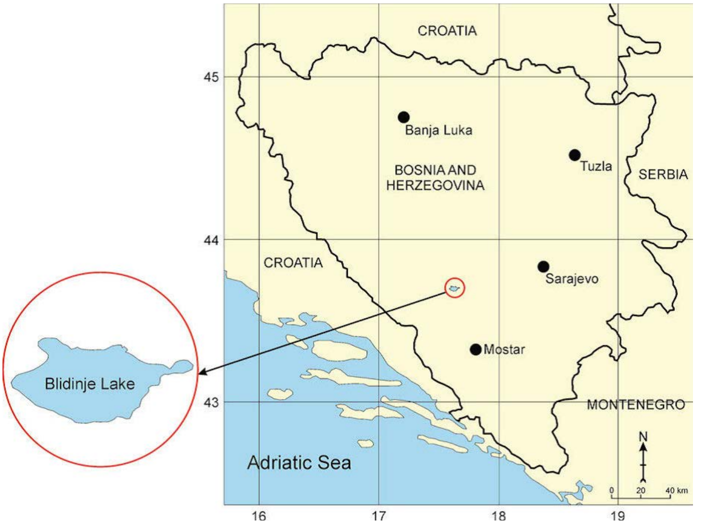

Fig. 1. Location of Blidinje lake on the map of B&H with noticed sampling locations (Ivanković, Hafner, 2012).

Results and discussion

Carlson's Trophic State Index – Secchi Depth

Blidinje lake is a shallow mountain lake of shady water, which varies throughout the day depending on the direction and speed of the wind. At different locations, there are different micro-conditions and different values of the same variables are possible at the same time (Ivanković et al., 2011).

Transparency of Blidinje lake is very low, especially from June to September. The minimum measured transparency at all locations is 10 cm and the largest is 50 cm, measured at locations 4 and 5. The average transparency is 22.81 cm, according to Harper (1992), indicates the eutrophic/hypertrophic status of Blidinje lake, which according to other relevant The indicators of the trophy are not so. There is no big difference in the measured translucency between individual locations. Everywhere, the transparency is reduced, and the reason is the suspended matter, not the density of phytoplankton (Ivanković et al., 2011).

According to Table 1, based on Carlson's Trophic State Index (TSI) values, the data is greater than 60. By entering the data into the Carlson equation, this is confirmed, so the lowest value of TSI (SD) is 69.99, and the highest is 93.18. The average value of TSI (SD) is 82.71. The differences in TSI (SD) depending on the location are very small.

These values of Blidinje lake are in the category of hypertrophic lake (Fig. 2). From June to September, when the transparency is significantly reduced, the TSI (SD) is very high and points to the hypertrophic state of the lake when it is possible to flourish the algae and eliminate fish, but this does not happen because other relevant indicators do not have hypertrophic values.

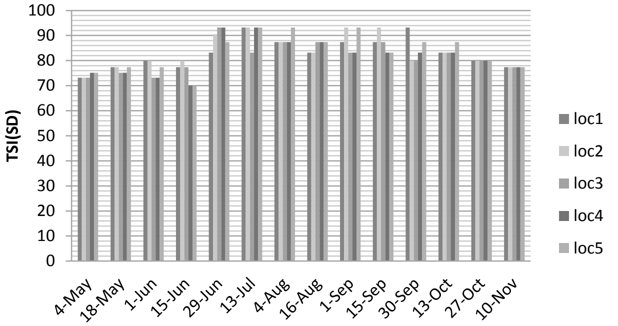

Fig. 2. Trophic state index – secchi depth.

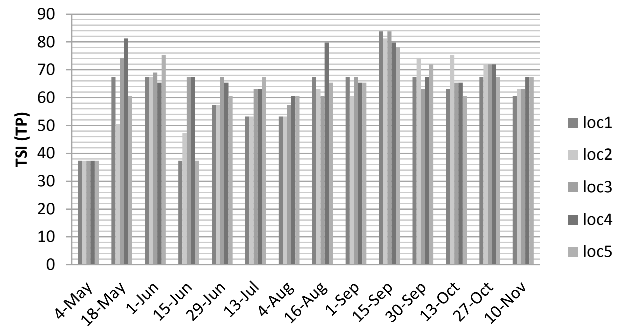

Fig. 3. Trophic state index – total phosphorus (nonfiltered).

Carlson's Trophic State Index – Phosphorus

Values of total phosphorus of non-filtered samples from Blidinje lake vary from 0.01 mg/L (10 μg/L), which is the lowest measured value for all sites at the same time (4 May 2008), up to 0.25 mg/L (250 μg/L), which is the highest measured value at location 3. Values very depending on the location and time in which samples are collected (Ivanković et al., 2011). The highest measured values of total phosphorus of Blidinje lake correspond to hypertrophic waters (>0.1) according to Harper (1992). Most measured values belong to the area of moderately eutrophic waters, whilst some of the least measured values correspond to the mesotrophic waters according to the same classification. According to Brancelj (2002), it is occasionally possible to record high values of total phosphorus even in oligotrophic waters.

According to this data for the total phosphorus of unfiltered samples, the Carlson TSI values for phosphorus is between 30 and 90 (Table 1), which classifies the Blidinje lake in the oligotrophic to hypertrophic (Table 2) categories depending on the location and time of sampling. Owing to the properties of shallow lakes that have a very intense interaction of water–sediment, large variations in total phosphorus values are possible.

By incorporating the data obtained into the Carlson equation for TSI (TP), the lowest value of TSI (TP) was calculated from 37.35 for May 4 at all locations (Chart 2). According to this value, the lake belongs to the category of oligotrophic lake, which occasionally in some locations may remain without oxygen in the summer (Table 2). The highest calculated value for September 15 at locations 1 and 3 is 83.77. This value sets the lake into a group of hypertrophic lakes (Table 2). The average calculated value of the TSI (TP) is 63.29, and according to this, the Blidinje lake belongs to the category of eutrophic lake (Table 2). Carlson›s TSI values for the total phosphorus of unfiltered samples depend on the location and time of sampling.

Phosphates of filtered samples were not recorded on several occasions during May, June and July. The lowest recorded value was 0.01 mg/L (10 µg / L) on several occasions at all sites, and the highest was recorded on May 18 at site 2 and August 15 at site 3 and was 0.23 mg/L (230 mg/L) (Ivanković et al., 2011). The mean value is 0.0384 mg/L (38.4 µg/L). This form of phosphorus is the only form available for algae for photosynthesis processes (Brancelj, 2002). All measured values of dissolved reactive phosphorus correspond to the mesotrophic category (Harper, 1992), except the values obtained for March 18 at location 2 and July 15 at locations 1, 3 and 4, belonging to the hypertrophic category according to Harper (1992).

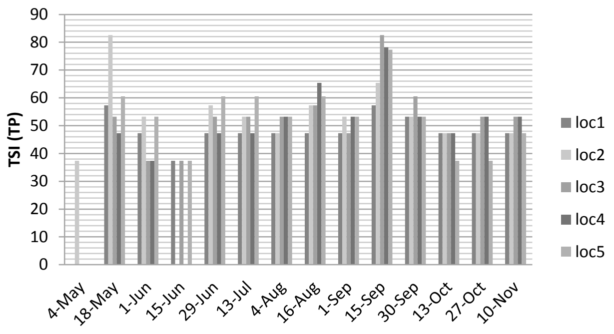

Fig. 4. Trophic state index – total phosphorus (filtered).

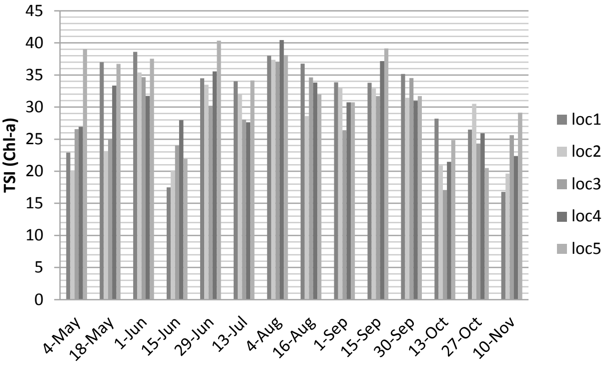

Fig. 5. Trophic state index – chlorophyll a.

According to the concentrations of total phosphorus, Blidinje lake is mesotrophic/eutrophic. This is probably due to the large amount of reactive phosphorus that binds solid particles of suspended matter and is thus inhibited.

According to Table 1, the obtained total phosphorus concentrations mostly correspond to TSI (TP) values from 40 to 70, which belong to a mesotrophic/eutrophic category (Table 2). The TSI (TP) calculated using Carlson‘s equation ranges from 37.35 to 82.56. The average value is 52.35. According to Figures 3 and 4, there is enough oscillation of the TSI (TP) depending on the location and time of sampling. However, most of the calculated TSI (TP) values at all lake locations are in the mesotrophic category (TSI) (TP) is 40–50) or eutrophic (TSI (TP) is 50–60) of lakes (Table 2). Only a couple of times during the year, high TSI (TP) (>70) was calculated at certain locations according to which the lake would fall into the category of hypertrophic lakes (Table 2).

Carlson's Trophic State Index – Chlorophyll a

The chlorophyll concentration ranges from 0.24 to 2.73 mg/m^3. The mean value is 1.15 mg/m^3. Although there is a large difference between concentrations measured at the same time

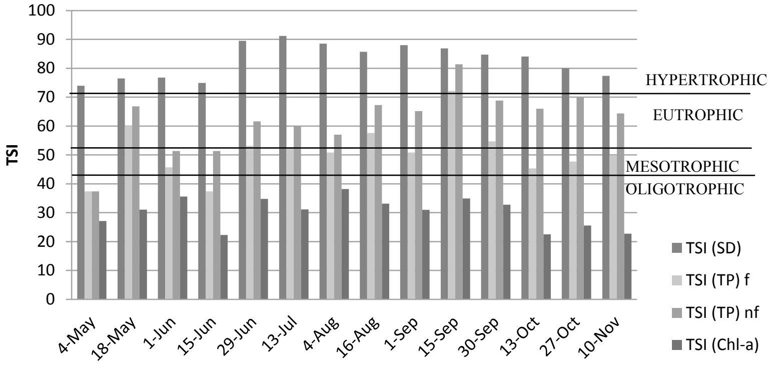

Fig. 6. Average TSI values for trophic indicators over the observed period.

at different locations, most of the measured values point to the oligotrophic status of lakes by many authors (Ivanković et al., 2011).

TSI (Chl-a) ranges from 16.78 to 40.44. The average value is 30.19 (Fig. 5). All calculated values at all locations rank Blidinje lake in the category of oligotrophic lakes with extremely clear water (Table 2).

Comparison and interpretation of results (relation between individual variables)

By comparing the mean values of the calculated TSI for transparency, total phosphorus and chlorophyll, it is apparent that each determines the trophic status of the lake and is not about the same or dependent on each other. Low TSI (Chl), High TSI (TP) and Highly High TSI (SD) (Fig. 6) cannot blend the Blidinje lake into the same category of trophy.

Considering that this is a shallow lake with a large amount of suspended matter (Ivanković, 2010) located on a limestone base, it cannot be classified into a given category for the value comparison shown in Table 3. However, most of the values are closest to the TSI (TP) = TSI (SD)> TSI (CHL) → no algae or colouring particles and light reduced to what has been the case with previous research (Ivanković, Hafner, 2012).

The TSI (Chl-a) is in the oligotrophic range throughout the test period, whilst TSI (TP) is mostly in mesotrophic and eutrophic waters and TSI (SD) is in hypertrophic waters (Table 2), Very low TSI (Chl-a) corresponds to a very high transparency up to a few metres of depth, and a low concentration of total phosphorus, but in the case of Blidinje lake, there are no such conglomerates.

According to Scheffer (2004), total phosphorus as a traditionally applied indicator for describing the level of trophy is not the best choice for shallow lakes where the water–sediment interaction is more pronounced. The dissolved reactive phosphor binds itself to solid suspended particles and is thus inhibited, algae cannot be used for the photosynthesis process, and this is the reason for its high concentration in samples, that is, high TSI (TP). And this also contributes to the reduction of transparency.

TSI (Chl-a) is low all the time during the test and indicates the oligotrophic state of the lake. This state of affairs should correspond to a clear transparency, that is, a low TSI (SD). In Blidinje lake, this is not the case, and the reason for the high TSI (SD) is a large amount of suspended matter rather than the phytoplankton density. Such a relationship between transparency and chlorophyll is common in limnology of shallow lakes of varying volumes, where the surface is subject to strong wind influences which constantly leads to resuspension of the precipitate (Engel, Nichols, 1994). Although there is no data on the macrophytes of Blidinje lake, it is evident that the lake is not rich in macrophytes (Ivanković, Hafner, 2012). According to Scheffer (2004), the lack of macrophytes contributes to blurredness, reduced transparency and increased resuspending of the sediment. Long, cold winter with ice cover also contribute to the lack of macrophytes in Blidinje lake (Ivanković et al., 2011).

The TSI (Chl-a) is low and fairly uniform all the time (Fig. 6) and at all locations (Fig. 5), indicating that Blidinje lake has no trend of increase in primary production and does not affect other trophies.

Conclusion

Wind, geological base and mountain character are the factors that affect the physical and chemical properties of the lake. The lake is shallow and exposed to the wind making it suitable not only for fishing but also for recreation and sport.

Blidinje lake has very low transparency, respectively, high TSI (SD), which would place it in hypertrophic lakes, whilst other relevant parameters do not show it. The reason for such a high TSI (SD) is the large amount of suspended particles, not the high TSI (Chl). TSI (Chl-a) and TSI (TP) (soluble phosphorus is the only phosphorus form that can be used by photosynthesis algae) in the lake indicate that it is oligotrophic/mesotrophic. Such a relationship between transparency and chlorophyll is common in limnology of shallow lakes of varying volumes, where the surface is subject to strong wind influences that constantly leads to resuspension of the precipitate. TSI (Chl-a) is all the time consistent, indicating that the lake has no trend of increasing primary production.

TSI (TP) is high, indicating eutrophic status, the reason being the separation of phosphorus that is absorbed from particles of suspended matter. Because the lake is shallow, this condition can be quickly changed. TSI (Chl) can be taken as the most relevant indicator for determining the status of Blidinje lake trophy. The slowdown of the eutrophication process is influenced by factors such as altitude, mountain climate and very rainy area. The trophic status of the lake may be useful for warning, monitoring and prediction. Data and information obtained by monitoring have a particular importance for the formation of different prognostic models and ecological modelling.

References

APHA (American Public Health Association) (1998*). Standard methods for the examination of water and wastewater.* Washington.

Brancelj, A. (Ed.). (2002). *High-mountain lakes in the eastern part of the Julian Alps*. Ljubljana: Založba ZRC.

Carlson, R.E. (1977). A trophic state index for lakes. *Limnol. Oceanogr.*, 22(2), 361–369. DOI: 10.4319/lo.1977.22.2.0361.

Carlson, R.E. (1981). Using trophic state indices to examine the dynamics of eutrophication. In *Proceedings of the International Symposium on Inland Waters and Lake Restoration* (pp. 218–221). U.S. Environmental Protection Agency.

Carlson, R.E. (1983). Discussion on "Using differences among Carlson's trophic state index values in regional water quality assessment". *Water Resour. Bull.*, 19, 307–309. DOI: 10.1111/j.1752-1688.1983.tb05335.x.

Carlson, R.E. & Simpson J.A. (1996). *Coordinator's guide to Volunteer lake monitoring methods*. North American Lake Management Society.

Engel, S. & Nichols S.A. (1994). Aquatic macrophyte growth in a turbid windswept lake. *J. Freshw. Ecol.*, 9, 97–109. DOI: 10.1080/02705060.1994.9664436.

Harper, D. (1992). *Eutrophication of freshwaters.* London: Chapman & Hall.

Ivanković, A. (2010). *Physico-chemical and biological characteristics of Blidinje lake.* Faculty of Agronomy and Food Technology, University of Mostar, Doctoral Thesis.

Ivanković, A., Velagić Habul, E. & Knezović Z. (2011). Physico-chemical characteristics of shallow, high mountain lake Blidinje (in a karst area of Bosnia and Herzegovina) with emphasis on its trophic status. *Oceanological and Hydrobiological Studies*, 40(3), 19–27. DOI: 10.2478/s13545-011-0025-4.

Ivanković, A. & Hafner D. (2012). Water quality assessment of the Blidinje shallow high mountain lake in Bosnia and Herzegovina, based on its biological properties. *Ekológia (Bratislava)*, 31(3), 295–308. DOI: 10.4149/ekol_2012_03_295.

Jeffrey, S.W., Mantoura, R.F.C. & Wright S.W. (1997). *Phytoplankton pigments in oceanography: guidelines to modern methods*. Paris: UNESCO Publishing.

Musa, S. (2005). Geomorphological specifications of Blidinje Nature Park (in Croatian). In *International Scientific Symposium Blidinje* (pp. 93–120). Mostar: Faculty of Civil Engineering.

Naumann, E. (1929). The scope and chief problems of regional limnology. *International Revue der Gesamten Hydrobiologie und Hydrographie*, 22(1), 423–444. DOI: 10.1002/iroh.19290220128.

OECD (1982). *Eutrophication of waters. Monitoring, assessment and control*. Paris.

Rocha, R.R.A., Thomaz, S.M., Carvalho, P. & Gomes L.C. (2009). Modeling chlorophyll-α and dissolved oxygen concentration in tropical floodplain lakes (Paraná River, Brazil). *Braz. J. Biol.*, 69(Suppl. 2), 491–500. DOI: 10.1590/S1519-69842009000300005.

Scheffer, M. (2004). *Ecology of Shallow lakes*. Dordrecht, Boston, London: Kluwer Academic Publishers.

Soldo, V. (2005). Research on habitat conditions and establishment of economic capacity for the cultivation of wild game species in Blidinje Nature Park (in Croatian). In *International Scientific Symposium Blidinje* (pp. 211–260). Mostar: Faculty of Civil Engineering.

Spahić, M.I. (2001). *Natural lakes of Bosnia and Herzegovina (in Bosnian).* Limnological monograph. Tuzla: Harfo-Graf.

Šimunović, V. & Bognar A. (2005). Geomorphological features of Blidinje Nature Park (in Croatian). In *International Scientific Symposium Blidinje* (pp. 25–40). Mostar: Faculty of Civil Engineering.

US EPA (2000). *Nutrient criteria technical guidance manual lakes and reservoirs*. EPA-822-B00-001. Washington: U.S. Environmental Protection Agency, Office of Water, Office of Science and Technology.

Welch, B.E. (1980). *Ecological effects of waste water*. Cambridge: Cambridge University Press.

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WATER SURFACE OVERGROWING OF THE TATRA'S LAKES

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Abstract

Kapusta J., Hreško J., Petrovič F., Tomko-Králo D., Gallik J.: Water surface overgrowing of the Tatra's lakes. Ekológia (Bratislava), Vol. 37, No. 1, p. 11–23, 2018.

Tatra's lakes are vulnerable ecosystems and an important element of the alpine landscape. Mainly some shallow lake basins succumb to intense detritus sedimentation, fine fractions of material from the catchment area or to the overgrowing of water level by vegetation. In this paper, changes and dynamics of the 12 Tatra's lake shorelines that were selected based on the detailed mapping of their extent are pointed out. Changes were assessed by accurate comparisons of historical and current orthophoto maps from the years 1949, 1955 and 2015 – and therefore, based on the oldest and the latest relevant materials. Due to the overgrowing of lakes caused by vegetation, their water surface decreased from -0.9% up to -47.9%, during the examined period. Losses were caused by the overgrowing of open water surface by the communities of sedges and peat bogs. The most significant dynamics of the shorelines during the last decades were reached by those lakes, into which fine sediments were simultaneously deposited by means of mountain water coarse. These sediments made the marginal parts of the lake basins shallower and accelerated rapid expansion of vegetation to the detriment of the open water surface. The overgrowing of shallow moraine lakes lying in the vegetation zone is a significant phenomenon of the High Tatras alpine landscape. It leads to their gradual extinction, turn into peat bogs and wet alpine meadows.

Key words: lakes, lake overgrowing, landscape changes, the High Tatra Mts.

Introduction

The glacial lakes of the High Tatra Mts. represent one of the most important and the most interesting natural elements of the alpine landscape. They are definitely complete and do not pose a natural threat to people today, as is the case in other alpine regions. They greatly increase the diversity of the alpine landscape and the whole landscape scenery as well.

According to Hreško et al. (2012), alpine lakes are vulnerable, integrated geomorphological-hydrological ecosystems. Their lake basins were finished, especially 27,000 to 19,000 years ago, during the climax of the last Ice Age peaked in the Tatra's. The last Würm glaciation lasted about 60,000 years and ended 10,000 to 8,500 years ago (Lukniš, 1973; Baumgart-Kotarba, Kotarba, 2001; Zasadni, Kłapyta, 2014; Engel et al., 2015). Most lakes were created in the last stages or immediately after the end of the Ice Age. According to Kłapyta et al.

(2015), deposition of sediments in lake basins of the Tatra's lakes began 9,000 to 8,000 years ago. Some shallow moraine lakes, after the appearance of their lake basins, were for certain time probably dry depressions. Their filling with water was only due to the onset of humid climate and consequent positive hydrological balance. It attests to the fact that there are no sediments older than those from the Holocene period in the shallow lakes.

The period after the final geomorphological completion of lake basins and the beginning of the sedimentation are considered as the beginning of the Holocene development of the Tatra's lakes and their basins. Since that time, exogenous geomorphological processes have begun. In many cases, it has strongly influenced the geomorphology of the lake basins and the shape of the shore lines. This development is caused mainly by the transport of sediments into lakes from the surrounding debris cones, rock gutters and walls. As a consequence, some lake basins become shallower by the gradual accumulation of the produced material, so that they gradually lose their open water and finally become extinct. The formation of lake sediments, according to Hreško et al. (2012), is a typical morphodynamic phenomenon of the alpine landscape in the High Tatra Mts. Debris flows and deposition of fine fractions in shallow parts of lake basins are the main causes of the creation of the sediments in lakes. Fine fractions are flooded out from adjacent debris cones, moraines and gutters, especially during their high water saturation due to extreme precipitation (Lukniš, 1973; Gregor, Pacl, 2005; Kapusta et al., 2010; Hreško et al., 2012; Długosz, Kapusta, 2015; Gallik, Bolešová, 2016). Intensive accumulation of weathered material is mainly due to the fact that the lake basins and their surroundings at highly rugged relief by elevation present a local erosive base for geomorphological processes. Debris flows and sediments that are washed out of debris cones and catchment loose movement energy, and deposit intensively as a result of a decrease in their positional energy. Especially in the shallower marginal parts of the lake basins, the sediments accumulate quickly to such extent that they gradually reach the water level and form a typical alluvial plains. These are characteristic geomorphological forms of the relief in the surroundings of many Tatra's lakes. Their extent might indicate the intensity of sedimentation processes. The abundant extension of these geomorphological forms on the bottom of the valleys indicates the distinctiveness of this phenomenon within the many lakes of the High Tatra Mts.

The exogenous geomorphological processes also include organogenic processes, which significantly represent the overgrowing of lakes by vegetation in the alpine region of the Tatra's. Based on the analyses of the peat sediment thickness (Dyakowska, 1932; Krippel, 1963; Obidowicz, 1996; Łajczak, 2014), it is evident that this process has long-term influence on the development of some lake basins in the alpine environment. From the view-point of the capturing the surface changes of glacial lakes caused by the growth of organic matter or by the overgrowing of open water by vegetation, attention has not been paid to this process. In the case of some lakes, the overgrowing of the lakes' marginal parts is a significant and typical phenomenon. According to our own measurements and field research, it is an exogenous geomorphological process, which — together with the draining of fine fractions and debris flows — causes the greatest changes and dynamics of the lake shorelines in the conditions of the High Tatra Mts. The shape of the shoreline and the rate of change over time can be an important indicator of the postglacial development of the Tatra's lakes.

In this paper, surface changes and changes in the shorelines of 12 selected shallow moraine lakes of the Tatra's are assessed. Analysis of changes is based on the detailed comparison of the oldest aerial photos (1949, 1955) and the latest orthophoto maps (2015). Changes in the shoreline due to the overgrowing of their open water by vegetation for the last 60–65 years are depicted. These particular lakes have been chosen according to the apparent overgrowing of the water surface by vegetation. This choice represents Tatra's lakes where the biggest parts of the water surface have been overgrown.

Study area

The High Tatras are one of the smallest high mountains in the world with a typically developed alpine relief – with glacial valleys and peaks. It was formed by multiple actions of mountain glaciers during the Pleistocene. The mountains were glaciated at least three times, being the most glaciated mass of the entire Carpathians. The High Tatras lie in an area of only 341 km^2, at the borderline of Slovakia and Poland (more than three quarters of their area spread out in the Slovak part). The whole mountain range of the Carpathian Mountains reaches the highest elevation by the peak Gerlachovský štít (2,654.4 m a.s.l.), with another 24 peaks exceeding the 2,500 m a.s.l. The research area is part of the Tatra National Park.

Nowadays, there are about 150–230 lakes of various sizes and depths (depending on the size and periodicity criteria) in the High Tatras. Approximately half of them are periodic. In addition, there is a significant number of already extinct, fully overgrown lakes and peat bogs. Mostly, there are small lakes with a surface of less than 1 ha and depth up to 2 m (Gregor, Pacl, 2005). The biggest lakes are Morskie Oko and Wielki Staw Polski (both with a surface of more than 0.3 km^2), the deepest being Wielki Staw Polski with a depth of 79.3 m.

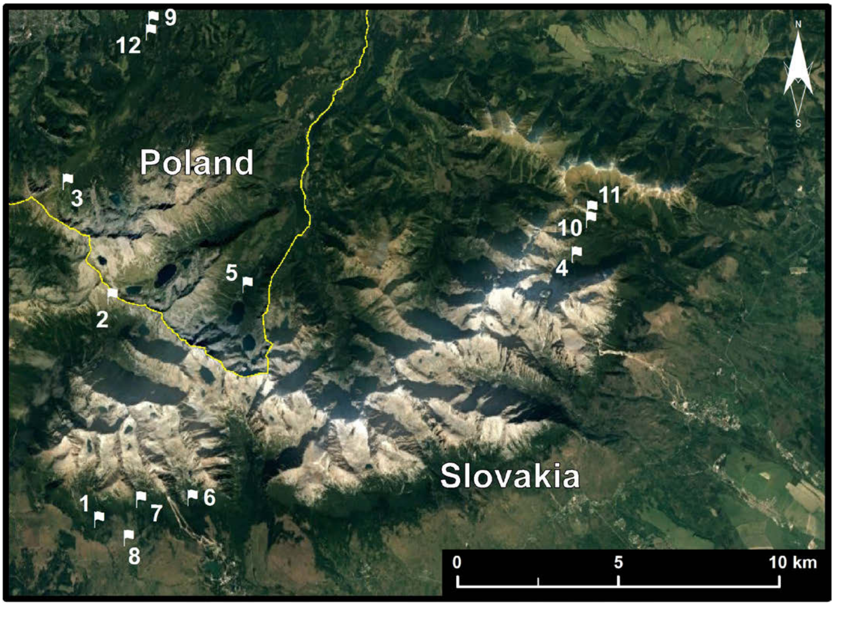

Fig. 1. The High Tatra Mountains – location of representative lakes: (1) Jamské pleso, (2) Kobylie pleso, (3) Litworowy Staw Gąsienicowy, (4) Malé Čierne pleso, (5) Małe Morskie Oko, (6) Mlynické pliesko, (7) Nižné Furkotské pleso, (8) Nižné Rakytovské pliesko, (9) Niżni Toporowy Staw, (10) Trojrohé pleso, (11) Veľké Biele pleso, (12) Wyżni Toporowy Staw.

The selected 12 lakes are located predominantly in the lower parts of different valleys, mostly in the forest zone or dwarf pine zone (Fig. 1). They lie at different elevations (from 1,089.0 to 1,734.3 m a.s.l.), as well as the northern and southern side of the mountain. All these lakes are relatively shallow (maximum depth ranges from 0.8 to 5.9 m). The average depth of the water in some lakes is only about 0.5 m. Selected lakes were created either as depression – melted hollows – by melting the dead ice floes buried under the detritus or by the rising of the water level behind waterproof moraine dams. The depth of water in some lakes of the interest ranges significantly during the year.

Material and methods

For each lake, the surface changes of the selected lake shorelines were identified based on a detailed comparison of the oldest historical aerial photo and the current orthophoto map. Historical aerial photos and current high-resolution orthophoto maps represent the ideal basis for the assessment of the landscape changes in the hardly accessible terrain in connection with the GIS tools. Such combination can be considered as a unique tool for mapping spatial changes in the glacial lake shorelines and assessing their dynamics over a longer time horizon. Today, remote sensing of the Earth (RSE) materials are the most accurate basis for detailed mapping and assessment of long-term changes of lake shorelines and alpine landscapes, generally. Twelve lakes, where changes in their shorelines are most clearly visible, were selected based on the preliminary analysis of the orthophoto maps of the entire High Tatra Mts. territory and on the terrain research.

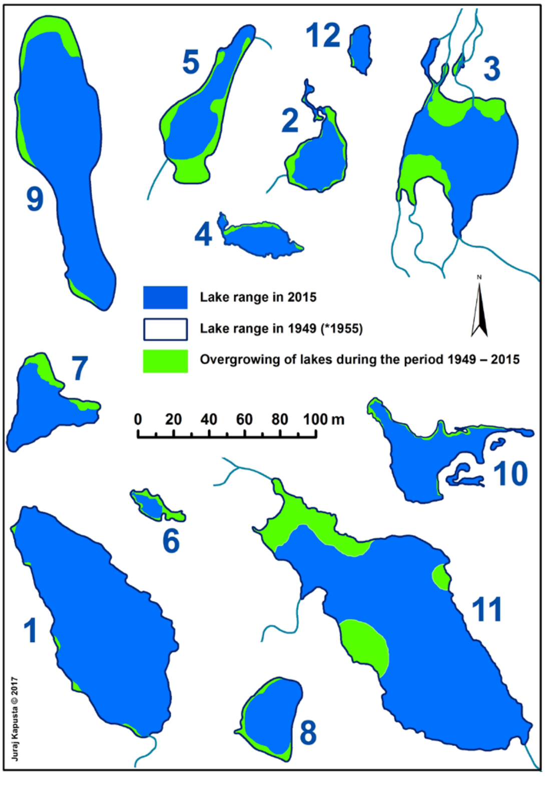

Fig. 2. Overgrowing of the Tatra´s lakes during the period 1949–2015.

For the identification of the current state, colour orthophoto maps from 2015 were used. They were created by Eurosense s.r.o. Bratislava and are of high resolution – up to 20 cm/pixel. Based on these orthophoto maps, current state of the lake shorelines of selected lakes were mapped in detail in the ArcView GIS — ArcMap 9.3 program. Digitisation was performed via the method of visual interpretation ('on screen') at a scale of 1:300 – 1:500. The punctuality of digitalization depends on the source's resolution, whereby the scale 1:500 provides precise results and for this type of the problem is above standard.

With the use of GIS tools, the oldest historical aerial photos were precisely matched with current orthophoto maps, based on a larger number of pass points. Based on this operation, historical photos were transformed into historical orthophoto maps, while positional distortion was minimised. In most of the lakes, the oldest material from which their historical range could be conquered, were aerial photographs from the year 1949. Boundaries of the historical shorelines on the Polish side were marked based on the aerial photos from 1955. These were panchromatic aerial photos scanned at a high resolution of 1,200 dpi and converted to digital format. Panchromatic aerial photos were provided by the Topographical Institute in Banská Bystrica.

The digitization of shorelines from historical orthophoto maps followed the rules of 'backdating' (Feranec et al., 2005). A current orthophoto map was always compared with a historical orthophoto map. Boundaries of the lake shorelines were based on the current orthophoto maps. Changes occurred only at the point of evident changes in their overlap with the historical orthophoto map. This method is useful particularly for identification of changes in landscape cover and structure (Boltižiar, 2007; Falťan, Bánovský, 2008; Solár, 2013; Solár, Janiga, 2013; Haladová, Petrovič, 2015; Kaczka et al., 2015; Kubinský et al., 2015), but it seems to be applicable for the assessment of geomorphological processes (Kapusta et al., 2010; Hreško et al., 2012; Długosz, Kapusta, 2015) and dynamics of lake shorelines (Kapusta, 2016). When digitizing the data in GIS (shorelines in this case), only a copy of the original layer, not the completely new layer is created. Then the original layer is modified so that the common boundaries of logically unchanged areas are left unchanged. Only boundaries of the shorelines are modified in places where there has been a definite change. The purpose of this method is to minimise the inaccuracies that result from the simple uncorrected overlapping of the digital map layers on each other.

Results

Based on the analysis of the oldest and the latest orthophoto maps of the selected Tatra's lakes in the years 1949, 1955 and 2015, changes of their shorelines mapped in detail are presented (Fig. 2). Determined dynamics of the shorelines is projected into surface losses calculated in the GIS environment. These surface losses represent losses of the open water level of individual lakes, due to the overgrowing of their water levels by vegetation. Some of the monitored lakes underwent significant changes in the above mentioned period (Figs 2, 3, 4, Table 1).

In the case of absolute changes, due to overgrowing, the losses of the open water level ranged from 15.4 m² (lake Wyżni Toporowy Staw) to 1,595.8 m² (lake Veľké Biele pleso – Fig. 5) during the monitored period. For some lakes, the ab-

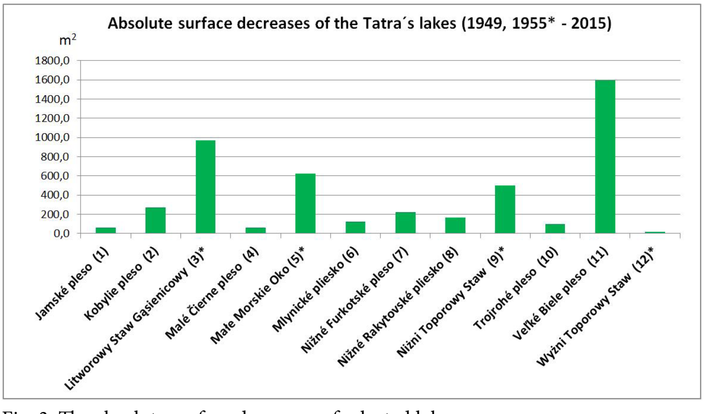

Fig. 3. The absolute surface decreases of selected lakes.

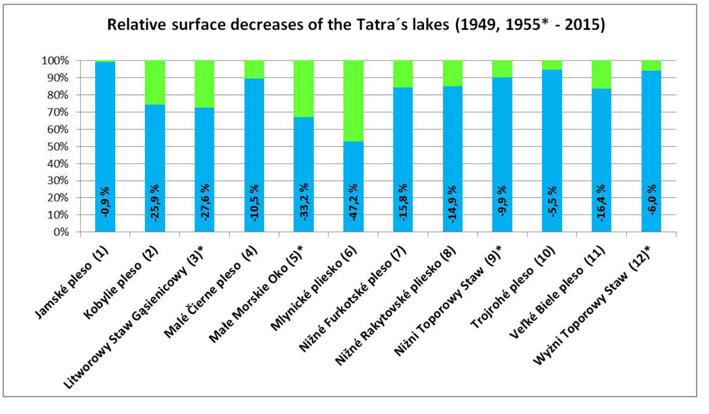

Fig. 4. The relative surface decreases of selected lakes.

T a b l e 1. Surface decreases of selected Tatra lakes.

The lake (Country)		Elevation (a.s.l.)	Max. depth (m)	Surface (m^2)			Absolute decrease of the surface (m^2) 1949, 1955*-2015	Relative decrease of the surface (%) 1949, 1955*-2015
				1949	1955*	2015		
1	Jamské pleso (SK)	1,447.5	4.3	6,383.7	-	6,323.9	59.8	-0.9
2	Kobylie pleso (SK)	1,734.3	1.0	1045.2	-	774.6	270.6	-25.9
3	Litworowy Staw Gąsienicowy (PL) *	1,618.0	1.1	-	3,517.1	2,546.5	970.6	-27.6
4	Malé Čierne pleso (SK)	1,565.9	2.0	608.3	-	544.4	63.9	-10.5
5	Małe Morskie Oko (PL) *	1,391.7	3.3	-	1,881.5	1,256.8	624.7	-33.2
6	Mlynické pliesko (SK)	1,552.0	-	262.9	-	138.8	124.1	-47.2
7	Nižné Furkotské pleso (SK)	1,626.0	1.2	1,406.4	-	1,184.3	222.1	-15.8
8	Nižné Rakytovské pliesko (SK)	1,307.0	2.1	1,136.0	-	966.2	169.8	-14.9
9	Niżni Toporowy Staw (PL) *	1,089.0	5.9	-	5,067.5	4,565.9	501.6	-9.9
10	Trojrohé pleso (SK)	1,610.8	1.4	1,811.5	-	1,711.4	100.0	-5.5
11	Veľké Biele pleso (SK)	1,615.4	0.8	9,726.8	-	8,131.0	1,595.8	-16.4
12	Wyżni Toporowy Staw (PL) *	1,120.0	1.1	-	256.7	241.3	15.4	-6.0
	Note: * - the oldest historical aerial images of the given lake.							

Fig. 5. The lake Veľké Biele pleso, view from the peak Jahňací štít – 2,229 m a.s.l. (Photo: J. Kapusta, 11.10.2014).

solute losses of the surface are inconsiderable (lake Wyżni Toporowy Staw – 15.4 m^2, lake Jamské pleso – 59.8 m^2, lake Malé Čierne pleso – 63.9 m^2, lake Trojrohé pleso – 100 m^2 (Fig. 6), lake Mlynické pliesko (Fig. 7) – 124.1 m^2). However, in the case of the last one, due to its small size, the above stated absolute loss represents up to 47.2%

of its loss compared to the year 2015 (Table 1). However, for some lakes, losses of the open water levels over the past 60 – 65 years, in consideration of the Tatra lake sizes, are significant (lake Veľké Biele pleso – 1595.8 m^2, lake Litworowy Staw Gąsienicowy – 970.6 m^2 (Fig. 8), lake Małe Morskie Oko – 624.7 m^2 (Fig. 9), lake Niżni Toporowy Staw – 501.6 m^2). In the case of relative changes, losses of the open water level ranged from -0.9% (lake Jamské pleso) up to -47.2% (lake Mlynické pliesko) compared to their original surfaces in 1949 and 1955 (Fig. 2, Table 1).

Fig. 6. The lake Trojrohé pleso, view from the peak Jahňací štít – 2229 m a.s.l. (Photo: J. Kapusta, 11.10.2014).

Fig. 7. The lake Mlynické pliesko. (Photo: J. Kapusta, 04.08.2017).

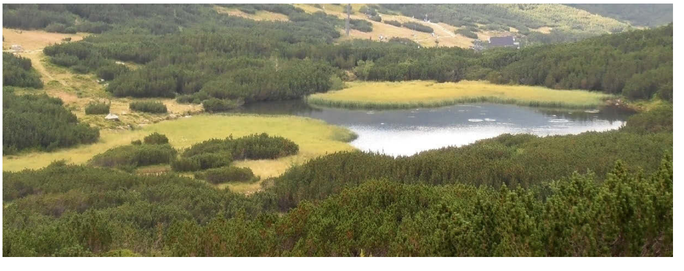

Fig. 8. The lake Litworowy Staw Gąsienicowy. (Photo: J. Kapusta, 06.08.2015).

All surface losses of monitored lakes were caused by overgrowing of their open water level by vegetation. Based on a field survey, two dominant ways of overgrowing with different intensity can be distinguished within the selected set of lakes: peat bog communities (Fig. 6, Fig. 10) and sedge communities (Fig. 5, Fig. 8). While the decrease in surface extent of lakes due to the accumulation of organic material is relatively slow nowadays, the overgrowing of water levels by sedges is in the case of some lakes considerably fast.

Typical examples of decrease of open water level due to the accumulation of organic material are lakes Wyżni Toporowy Staw and Trojrohé pleso (Fig. 6). Both lakes are a part of typical Tatra's peat bogs. The extent of peat bogs is several times larger than the surfaces

Fig. 9. The lake Małe Morskie Oko. (Photo: J. Kapusta, 03.09.2016).

Fig. 10. The lake Malé Čierne pleso. (Photo: J. Kapusta, 03.08.2017).

of the current lake open water levels. In the monitored period, their absolute decrease of the open water levels was relatively small (Fig. 2, Table 1). In the case of the lake Trojrohé pleso, the extremely complicated shape of the shoreline evokes the rapid and intense overgrowing of water level. In the 65 years, approximately 100 m^2 of its water surface was overgrown, which represents a decrease of only -5.5% of its surface compared to the year 1949. Parallel with other lakes (Fig. 3, Table 1), it is a low surface decrease. However, part of this decrease was caused by the overgrowing of the northwest shallow lake bay by sedge communities. The maximum depth of the lake is 1.4 m, but the edges of its lake basin are probably still too deep to become more overgrown with sedges also in other parts.

However, overgrowing of the open water level of the lakes by sedge communities shows to be much faster and more dynamic. This type of overgrowing mainly affects shallow lake basins. Their marginal parts have a depth of only a few centimetres in some places, resp. several tens of centimetres. Aggressive vegetation is spreading extremely fast here at the expense of open water level. Typical examples of water level decrease due to overgrowing by sedges are the lake Veľké Biele pleso (Fig. 5) and the lake Litworowy Staw Gąsienicowy (Fig. 8). In the case of these lakes, significant changes of shorelines in a relatively short time are multiplied by the input of fine sediments from their catchments. The surface of the lake Veľké Biele pleso decreased between 1949 and 2015 by 1,595.8 m^2, which represents a decrease of the open water level by 16.4% compared to its surface in 1949 (Fig. 11). Similarly, the lake Litworowy Staw Gąsienicowy showed a very large surface decrease (-970.6 m^2) in the period 1955–2015. This surface area represents loss of the open water level by up to -27.6% compared to the year 1955 (Fig. 11).

Watercourses flow through the both of lakes. In particular, during extreme precipitation and high-water conditions, watercourses bring larger quantities of suspended loads and sand from the higher lying catchment areas. Sediments accumulate in the lake basins and gradually make the lake shallower. At the same time, in the marginal shallow parts of lakes, suitable substrate conditions are created for sedge root systems, which can then rapidly expand at the expense of open water. Their old dying parts, which accumulate under the water, also contribute to the expansion. This creates a layer of organic remains mixed with fine sediments and flooded sand. Both basins are very shallow, the average depth of the lake Veľké Biele pleso is only about 0.5 m. The combination of an acceptable substrate for the root systems of the sedges and of lake shallowness creates suitable conditions for the rapid overgrowing of the open water level of the lake.

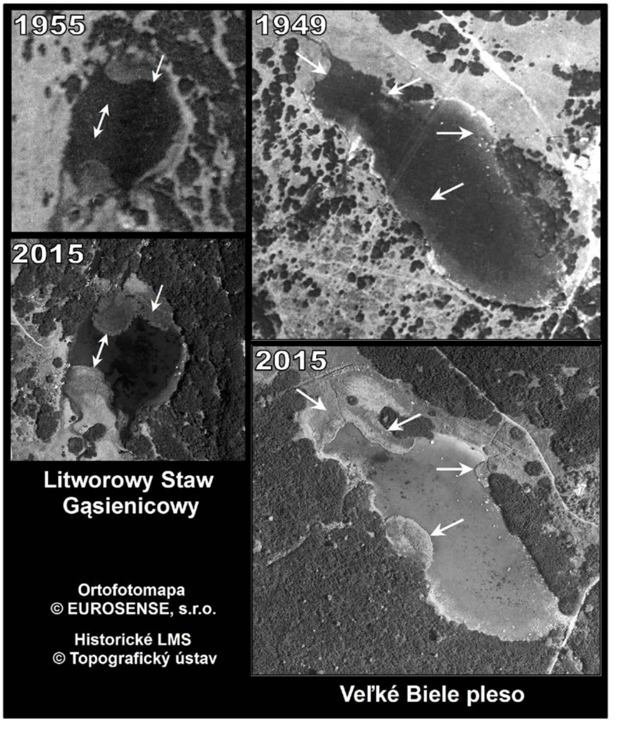

Fig. 11. Comparison of the oldest (1949, 1955) and the newest (2015) orthophoto maps - Litworowy Staw Gąsienicowy and Veľké Biele pleso. (Author: J. Kapusta, 2017).

The third highest decrease in open water level during the monitored period was recorded in the lake Małe Morskie Oko (624.7 m^2, -33.2%). It is also a flow-through lake. Although this lake reaches a relatively large maximum depth (3.3 m), it intensively overgrows along the edges (Fig. 9). In the case of other explored lakes, open water level decreased mostly by sedge communities as well as peat bogs.

The highest relative surface decrease was reached in the case of the lake Mlynické pliesko during the period of years 1949–2015; it was up to -47.2%. The area that was covered by vegetation is relatively small (124.1 m^2). However, this lake is generally of very small size, therefore, even the low absolute decrease of the surface was manifested by a high relative decrease. It is a periodic lake, relatively less known, and it is marked in more detailed maps only. From an ecological point of view, however, in case of such small dimensions, it is also the lake ecosystem.

The overgrowing of individual lakes has a different dynamic. While some of the representative lakes show a relatively negligible speed of the overgrowing, the dynamics and changes of other lake shorelines are literally extreme (Fig. 2, Fig. 11, Table 1). Moreover, the degree of the overgrowing also significantly affects the periodicity and the changing hydrological balance of the lake basins. The most significant changes show lakes that have their bottoms with distinct shallow areas, into which fine sediments are brought by the water stream and that are significantly overgrown with sedge communities (Litworowy Staw Gąsienicowy, Veľké Biele pleso). During the low water levels, sedges can expand rapidly; while during the high-water levels, occasionally, some overgrown areas are temporarily below the water surface. These lakes probably become overgrown much faster than those with typical overgrowing by peat bog communities (e.g.the lake Trojrohé pleso). In 1949, the lake Veľké Biele pleso had surface of open water level about 5.3 times larger than the lake Trojrohé pleso. However, the overgrown area between the years 1949–2015 at the lake Veľké Biele pleso is up to almost 16 times bigger than the overgrown area at the lake Trojrohé pleso (Table 1). It follows that the overgrowing intensity of the water level in the lake Veľké Biele pleso is thus more than 3 times higher than in the lake Trojrohé pleso.

The strong spatial dynamics of some lake shorelines suggests the intense interaction of mountain glacial lakes and organogenic deposition processes. Overgrowing of the open water levels of Tatra's lakes by vegetation leads to their gradual extinction and it is the result of this interaction. At the bottom of the valley, it is possible to identify a large number of already overgrown lakes or lakes filled with peat (e.g. the lake Slepé pleso near to the lake Štrbské pleso) on the basis of orthophoto map and field survey. These are the lake basins of the former lakes, which nowadays represent the final stage of organogenic processes – the full coverage of open water by vegetation. Nowadays, they are valuable biotopes in the form of peat bogs and alpine wetlands with rare plant species.

Discussion

The display of the negative changes of the Tatra's lakes' shorelines, based on the historical and current orthophoto maps, is still rare in scientific works published so far. It has been used in researches where it emerged secondary from a study of other landscape phenomena (Kapusta et al., 2010; Hreško et al., 2012; Długosz, Kapusta, 2015; Gallik, Bolešová, 2016; Kapusta, 2016).

Displayed dynamics of the shorelines (Fig. 2, Fig. 11) shows their real-time development as a result of the overgrowing of the open water levels with vegetation. Based on the presented changes, many alpine lakes are not stable and unchangeable, but they are an open and dynamic system. The most intense overgrowing by vegetation is associated with the deposition of fine-grained fractions and the accumulation of organic material in their shallow coastal parts. The biggest changes occurred in shallow moraine lakes, where the process of extinction is much more distinct than in deeper and in bigger lakes by volume.

According to Mason et al. (1994), lakes are a sensitive indicator of changing geomorphology in catchment areas and of the climate change. This fact is confirmed in particular by the analyses of lake sediments (Owens, Slaymaker, 1994; Kotarba, 1996; Šporka et al., 2002; Irmler et al., 2006; Rybníčková, Rybníček, 2005; Hamerlík, Bitušík, 2009; Mîndrescu et al., 2010; Hutchinson et al., 2016; Kłapyta et al., 2015), and also by the results of other studies (Kotarba, 2004; Kapusta et al., 2010; Hreško et al., 2012; Necsoiu et al., 2013; Kohler et al., 2014; Gądek et al., 2015; Kubinský et al., 2015; Gallik, Bolešová, 2016). In particular, smaller and shallower lakes react much more sensitively to changes than larger lakes because of their small volume of the water. They can thus be a strong indicator of the ongoing changes in their catchment areas (Mason et al., 1994; Gerten, Adrian, 2000; Adrian et al., 2009).

Glacial lakes in high mountains, where glaciers are currently melting as a cause of climatic change, react via significant surface expansion of water levels due to the positive hydrological balance (Ives et al., 2010; Strozzi et al., 2012; Emmer et al., 2015). According to Adrian et al. (2009), shallow lakes have a strong potential to become natural indicators of ongoing climate change. It seems that some shallow moraine Tatra's lakes respond to the ongoing changes by rapid overgrowing of the open water level and consequent surface decrease. In many cases, the spatial losses of Tatra's lakes, according to our own depth measures of the lake bottom, undoubtedly indicate a significant reduction in the retention capacity of the lake basins too.

Overgrowing of the shallow moraine lakes on the bottom of the valleys is one of the typical phenomena of the alpine landscape in the Tatras. Together with the accumulation of sediments from debris flows and fine fractions of material flooded from debris flow fans, it can be classified as an exogenous process that over the last decades affected the development of lakes and their shorelines to a high extent. Surrounding shallow moraine glacial lakes is a powerful indicator of the changes taking place in the Tatra's alpine landscape. The above stated glacial lakes are strong indicators of changes that occur in the high mountainous landscape of the High Tatra Mts.

Conclusion

The mapping of negative changes in the shorelines of Tatra's lakes, based on historical and current orthophoto maps, represents one of the ways to capture, analyse and quantitatively visualize their dynamics and development. Between 1949 and 2015, the number of significant decreases in the open water level of the moraine lakes was identified as a result of their overgrowing with sedge and peat bog communities. Lakes that succumbed to the accumulation of fine sediments from water streams during this period generally show much more intense extent of overgrowing than others. Systematic overgrowing of water levels of the shallow moraine lakes is one of the typical phenomena of the Tatra's alpine landscape. Since the beginning of the Holocene, many lakes, due to the action of these organogenic processes, have been changed to a typical alluvial plains, alpine wetlands and peat bogs. These are the final stages of the development of shallow moraine lakes located at the lower elevations in the vegetation zone. Accumulation of fine sediments and organogenic material in the lake basins of such lakes is undoubtedly reflected in a systematic reduction of their original volume. Overgrown lakes thus become the sensitive indicator of long-term changes taking place in the alpine landscape of the High Tatra Mts.

A following research would be focused on further systematic monitoring of the development of representative lake shorelines and their interactions with organogenic processes. The attention is currently focused on the identification of not only the original range of some of the existing basins of Tatra's lakes, but also extinct.

Historical and current orthophoto maps in combination with GIS tools and field research have a great potential for providing valuable information on changes of the alpine glacial lakes.

Confirmed dynamics of the lake shorelines presented in this paper can serve as a clear example of the future development of shallow lakes also in other high mountains of the temperate climatic zone.

Acknowledgements

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References

Adrian, R., O'Reilly, C.M., Zagarese, H., Baines, S.B., Hessen, D.O., Keller, W., Livingstone, D.M., Sommaruga, R., Straile, D., Van Donk, E., Weyhenmeyer, G.A. & Winder M. (2009). Lakes as sentinels of climate change. *Limnol. Oceanogr.*, 54(6), 2283–2297. DOI: 10.4319/lo.2009.54.6_part_2.2283.

Baumgart-Kotarba, M. & Kotarba A. (2001). Deglaciation in the Sucha Woda and Pańszczyca Valleys in the Polish High Tatras. *Studia Geomorphologica Carpatho-Balcanica*, 35, 7–38.

Boltižiar, M. (2007). *Štruktúra vysokohorskej krajiny Tatier*. Nitra: ÚKE SAV.

Dąbrowska, K. & Guzik M. (Eds.) (2015). *Atlas of the Tatra Mountains – Abiotic Nature*. Zakopane: Wydawnictwo Tatrzańskiego Parku Narodowego.

Długosz, M. & Kapusta J. (2015). Debris flows. (Plate V.2). In K. Dąbrowska & M. Guzik (Eds.), *Atlas of the Tatra Mountains – Abiotic Nature*. Zakopane: Wydawnictwo Tatrzańskiego Parku Narodowego.

Dyakowska, J. (1932). Analiza pyłkowa kilku torfowisk tatrzańskich. *Acta Botanica Polonica*, 9(3–4), 473–530.

Emmer, A., Merkl, S. & Mergili M. (2015). Spatiotemporal patterns of high-mountain lakes and related hazards in western Austria. *Geomorphology*, 246, 602–616. DOI: 10.1016/j.geomorph.2015.06.032.

Engel, Z., Mentlík, P., Braucher, R., Minár, J., Léanni, L. & Team A. (2015). Geomorphological evidence and ^{10}Be exposure ages for the Last Glacial Maximum and deglaciation of the Veľká and Malá Studená dolina valleys in the High Tatra Mountains, central Europe. *Quaternary Science Reviews*, 124, 106–123. DOI: 10.1016/j.quascirev.2015.07.015.

Falťan, V. & Bánovský M. (2008). Changes in land cover in the area of Vyšné Hágy-Starý Smokovec, impacted by the wind calamity in November 2004 (Slovakia). *Moravian Geographical Reports*, 16, 16–26.

Feranec, J., Cebecauer, J. & Oťaheľ J. (2005). *Photo-to-photo interpretation manual*. Bratislava: Institute of Geography, Slovak Academy of Sciences, BIOPRESS.

Gądek, B., Grabiec, M. & Kędzia S. (2015). Application of ground penetrating radar to identification of thickness and structure of sediments in postglacial lakes, illustrated with an example of the Mały Staw lake (the Karkonosze Mts.). *Studia Geomorphologica Carpatho-Balcanica*, 49, 5–13. DOI: 10.1515/sgcb-2015-0006.

Gallik, J. & Bolešová L. (2016). sUAS and their application in observing geomorphological processes. *Solid Earth*, 7, 1033–1042. DOI: 10.5194/se-7-1033-2016.

Gerten, D. & Adrian R. (2000). Climate-driven changes in spring plankton dynamics and the sensitivity of shallow polymictic lakes to the North Atlantic Oscillation. *Limnol. Oceanogr.*, 45, 1058–1066. DOI: 10.4319/lo.2000.45.5.1058.

Gregor, V. & Pacl J. (2005). Hydrológia tatranských jazier. *Acta Hydrologica Slovaca*, 6(1), 161–187.

Haladová, I. & Petrovič F. (2015). Classification of land use changes (model area: Nitra town). *Ekológia (Bratislava)*, 34(3), 249–259. DOI: 10.1515/eko-2015-0024.

Hamerlík, L. & Bitušík P. (2009). The distribution of littoral chironomids along an altitudinal gradient in High Tatra Mountain lakes: Could they be used as indicators of climate change? *Ann. Limnol. – International Journal of Limnology*, 45, 145–156. DOI: 10.1051/limn/2009021.

Hreško, J., Kanásová, D., Bugár, G., Petrovič, F. & Mačutek J. (2012). Morphodynamic effect on lacustrine deposits in the High Tatra Mts. *Ekológia (Bratislava)*, 31(4), 390–404. DOI: 10.4149/ekol_2012_03_390.

Hutchinson, S.M., Akinyemi, F.O., Mîndrescu, M., Begy, R. & Feurdean A. (2016). Recent sediment accumulation rates in contrasting lakes in the Carpathians (Romania): impacts of shifts in socio-economic regime. *Regional Environmental Change*, 16(2), 501–513. DOI: 10.1007/S10113-015-0764-7.

Irmler, R., Daut, G. & Mäusbacher R. (2006). A debris flow calendar derived from sediments of lake Lago di Braies (N. Italy). *Geomorphology*, 77, 69–78. DOI: 10.1016/j.geomorph.2006.01.013.

Ives, J.D., Shrestha, R.B. & Mool P.K. (2010). *Formation of Glacial Lakes in the Hindu Kush-Himalayas and GLOF Risk Assessment*. Kathmandu: ICIMOD.

Kaczka, R., Lempa, M., Czajka, B., Janecka, K., Rączkowska, Z., Hreško, J. & Bugár G. (2015). The recent timberline changes in the Tatra Mountains: a case study of the Mengusovská valley (Slovakia) and the Rybi potok valley (Poland). *Geographia Polonica*, 88(2), 71–83. DOI: 10.7163/GPol.0016.

Kapusta, J., Stankoviansky, M. & Boltižiar M. (2010). Changes in activity and geomorphic effectiveness of debris flows in the High Tatra Mts. within the last six decades (on the example of the Velická dolina and Dolina Zeleného plesa valleys). *Studia Geomorphologica Carpatho-Balcanica*, 44, 5–34.

Kapusta, J. (2016). *Metodika rekonštrukcie brehovej čiary tatranských plies*. Rigorózna práca, UKF Nitra.

Kłapyta, P., Zasadni, J., Pociask-Karteczka, J., Gajda, A. & Franczak P. (2015). Late Glacial and Holocene paleoenvironmental records in the Tatra Mountains, East-Central Europe, based on lake, peat bog and colluvial sedimentary data: A summary review. *Quaternary International*, 415, 126–144. DOI: 10.1016/j.quaint.2015.10.049.

Kohler, T., Wehrli, A. & Jurek M. (Eds.) (2014). *Mountains and climate change: A global concern. Sustainable Mountain Development Series*. Bern: Centre for Development and Environment (CDE), Swiss Agency for Development and Cooperation (SDC), Geographica Bernensia.

Kotarba, A. (1996). Osady jeziorne jako wskaźnik przemian środowiska naturalnego Tatr Wysokich. In A. Kotarba (Ed.), *Z badań fizycznogeograficznych w Tatrach, Dokumentacja Geograficzna*, 4 (pp. 33–47). Warszawa: IGiPZ PAN.

Kotarba, A. (2004). *Rola małej epoki lodowej w przekształcaniu śrdowiska Tatr*. Warszawa: IGiPZ PAN.

Krippel, E. (1963). Postglaciálny vývoj lesov Tatranského národného parku. *Biologické Práce*, 9(40), 5–44.

Kubinský, D., Weis, K., Fuska, J., Lehotský, M. & Petrovič F. (2015). Changes in retention characteristics of 9 historical artificial water reservoirs near Banská Štiavnica, Slovakia. *Open Geosciences*, 1, 1–8. DOI: 10.1515/geo-2015-0056.

Łajczak, A. (2014). Rola podłoża w rozwoju torfowisk w polskich Karpatach. *Studia Limnologica et Telmatologica*, 8(1), 19–36.

Lukniš, M. (1973). *Reliéf Vysokých Tatier a ich predpolia*. Bratislava: SAV.

Mason, I., Guzkowska, M., Rapley, C. & Street-Perrott F. (1994). The response of lake levels and areas to climatic change. *Clim. Change*, 27, 161–197. DOI: 10.1007/BF01093590.

Mîndrescu, M., Cristea, I. & Hutchinson S.M. (2010). Bathymetric and sedimentological changes of glacial Lake Ştiol, *Rodna Masiff. Carpathian Journal of Earth Environmental Sciences*, 5(1), 57–65.

Necsoiu, M., Dinwiddie, C.L., Walter, G.R., Larsen, A. & Stothoff S.A. (2013). Multitemporal image analysis of historical aerial photographs and recent satellite imagery reveals evolution of water body surface area and polygonal terrain morphology in Kobuk Valley National Park, Alaska. *Environmental Research Letters*, 8(2), 025007. DOI: 10.1088/1748–9326/8/2/025007.

Obidowicz, A. (1996). A Late glacial-Holocene history of the formation of vegetation belts in the Tatra Mts. *Acta Paleobot*., 36(2), 159–206.

Owens, P. & Slaymaker O. (1994). Post-glacial temporal variability of sediment accumulation in a small alpine lake. Variability in Stream Erosion and Sediment Transport. Proceedings of the Canberra Symposium, December 1994. *International Association of Hydrological Sciences*, 224, 187–195.

Rybníčková, E. & Rybníček K. (2005). Pollen and macroscopic analyses of sediments from two lakes in the High Tatra mountains, Slovakia. *Veg. Hist. Archaeobot.*, 15, 345–356. DOI: 10.1007/s00334-006-0050-1.

Solár, J. (2013). Effect of climate change on Mountain Pine distribution in Western Tatra Mountains. In B.R. Singh (Ed.), *Change - Realities, Impacts Over Ice Cap, Sea Level and Risks*, pp. 437–458. Rijeka: InTech. DOI: 10.5772/54724.

Solár, J. & Janiga M. (2013). Long-term changes in Dwarf Pine (Pinus mugo) cover in the High Tatra Mountains, Slovakia. *Mt. Res. Dev.*, 33(1), 51–62. DOI: 10.1659/MRD-JOURNAL-D-12-00079.1.

Strozzi, T., Wiesmann, A., Kääb, A., Joshi, S. & Mool P. (2012). Glacial lake mapping with very high resolution satellite SAR data. *Natural Hazards and Earth System Sciences*, 12, 2487–2498. DOI: 10.5194/nhess-12-2487-2012.

Šporka, F., Štefková, E., Bitušík, P., Thompson, A.R., Agustí-Panareda, A., Appleby, P.G., Grytnes, J.A., Kamenik, C., Krno, I., Lami, A., Rose, N. & Shilland N.E. (2002). The paleolimnological analysis of sediments from High Mountain Lake Nižné Terianske pleso in the High Tatras (Slovakia). *J. Paleolimnol.*, 28, 95–109. DOI: 10.1023/A:1020376003123.

Zasadni, J. & Kłapyta P. (2014). The Tatra Mountains during the Last Glacial Maximum. *Journal of Maps*, 10(3), 440–456. DOI: 10.1080/17445647.2014.885854.

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ALIEN PLANT SPECIES IN THE AGRICULTURAL HABITATS OF UKRAINE: DIVERSITY AND RISK ASSESSMENT

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Abstract

Burda R.: Alien plant species in the agricultural habitats of Ukraine: diversity and risk assessment. Ekológia (Bratislava), Vol. 37, No. 1, p. 24–31, 2018.

This paper is the first critical review of the diversity of the Ukrainian adventive flora, which has spread in agricultural habitats in the 21st century. The author's annotated checklist contains the data on 740 species, subspecies and hybrids from 362 genera and 79 families of non-native weeds. The floristic comparative method was used, and the information was generalised into some categories of five characteristic features: climamorphotype (life form), time and method of introduction, level of naturalisation, and distribution into 22 classes of three habitat types according to European Nature Information System (EUNIS). Two assessments of the ecological risk of alien plants were first conducted in Ukraine according to the European methods: the risk of overcoming natural migration barriers and the risk of their impact on the environment. The exposed impact of invasive alien plants on ecosystems has a convertible character; the obtained information confirms a high level of phytobiotic contamination of agricultural habitats in Ukraine. It is necessary to implement European and national documents regarding the legislative and regulative policy on invasive alien species as one of the threats to biotic diversity.

Key words: adventive flora, taxonomiccomposition, typologicalstructure.

Introduction

The threat of alien plant species, spreading in Ukrainian agricultural habitats, is indeed real. In recent decades, there has been an intensification of the invasions of aggressive neophytes in Ukraine. A checklist of alien plant species, containing 740 species, was compiled in the endangered areas of our country. Here, alien species are considered as species that formed secondary areas. In this article, in addition to alien species, all non-native species distributed in agricultural environments regardless of man are analysed. However, only the ecological risk of alien species is assessed.

The goals of the study were to research the composition of alien species and the taxonomic and typological structure of the adventive flora and assess the ecological risk of alien plants in agricultural habitats.

Material and methods

The checklist of alien plant species in the agricultural habitats of Ukraine is the result of authorial research since 2001 and the analysis of other scientific papers by the author. A pragmatic classification of the Angiosperm system (Mosyakin, 2013) and international databases on the nomenclature (The International Plant Name Index; ...; The Plant List. A working list of all plant species ...) were used. The typological analysis involved the application of floristic comparative methods.

The information was generalised using five descriptions of species *according to the* Ecoflora of Ukraine (Didukh et al., 2000): climamorphotype, 5 categories; time of introduction, 3 categories; method of introduction, 3 categories; level of naturalisation, 4 categories and distribution of species in 22 classis of three habitat types of European Nature Information System (EUNIS) (Davies et al., 2004). The habitat types are listed in accordance with the habitat classification version for Ukraine (Didukh et al., 2016). In Ukraine, 'agricultural habitats' is the term used to describe agricultural land (arable field, old field, fallow, haymaking, pasture meadow, cultivated pasture, intensive pasture and orchard) and land for non-agricultural use (rural settlement, field road, forest belt, protective forest tape). The graphic material was prepared using Statistical 6.0 package (Stat Soft. Inc.).

The risk assessment for overcoming natural migration barriers was made based on the known botanical classification of alien plant species (Richardson et al., 2000). The study of the ecological impact of alien plant species involved the use of the unified classification of the latter, depending on the magnitude of their environmental impact (Blackburn et al., 2014). The classes of impacts were distinguished as follows: 'massive', 'major', 'moderate', 'minor' and 'minimal'. The impact class was determined using mechanisms of the Global Invasive Species Database, The International Union for Conservation of Nature IUCN (Blackburn et al., 2014), such as competition, hybridisation, transmission of diseases, parasitism, toxicity, bioaccumulation, inflammability and interaction with other invasive species.

Results

The taxonomic diversity of the studied weed flora comprises 700 species, 23 subspecies and 17 hybrids from 362 genera and 79 families (about 65% of those recorded in the country). Amongst families, *Asteraceae* (55 genera, 117 species), *Poaceae* (29 and 85, respectively) and *Brassicaceae* (30 and 53) are the richest in the quantity of genera and species. Consequently, there is an evident prevalence of the families, common for our flora, whose richness in adventive fractions are predefined by their variety in Palaearctic on the whole. Exotic alien species belong to the families with one (22 families) or two (13 families) species, for example, *Martyniaceae* (*Proboscidea louisiana* (Mill.) Thell.), *Hippocastanaceae* (*Aesculus hippocastanum* L.), *Phytolaccaceae* (*Phytolacca americana* L.), *Simaroubaceae* (*Ailanthus altissima* (Mill.) Swingle) and *Commelinaceae* (*Commelina communis* L. and *Tradescantia virginiana* L.). Only *Amaranthaceae* includes 15 species and the only genus, *Amaranthus* L. A separate approach for inclusion in the checklist … is adopted for the genus *Solanum*. The 'List of regulated pests ...' of the Ministry of Agrarian Policy of Ukraine contains *Solanum rostratum* Dunal. It is known from scientific publications that this quarantine weed occupied about 1,688 ha in 2002, but the measures taken allowed reducing the area to 134 ha. Three species of the genus *Solanum* (*Solanum carolinense* L., *S. elaeagnifolium* Cav. and *S. triflorum* Nutt.) in the 'List of quarantine pests…' are considered to be absent in our country. However, a rare alien species *S. carolinense* was collected in Ukraine only once: in Kyiv, near a grain mill (Mosyakin, Fedoronchuk, 1999). Another species of *S. judaicum* Besser was noted by Besser from Podolia in 1809, but the fate of this species is not known. As the invasion of *Solanum* species continues in Ukraine, all the mentioned species of the genus are included in the checklist… with the corresponding assessments.

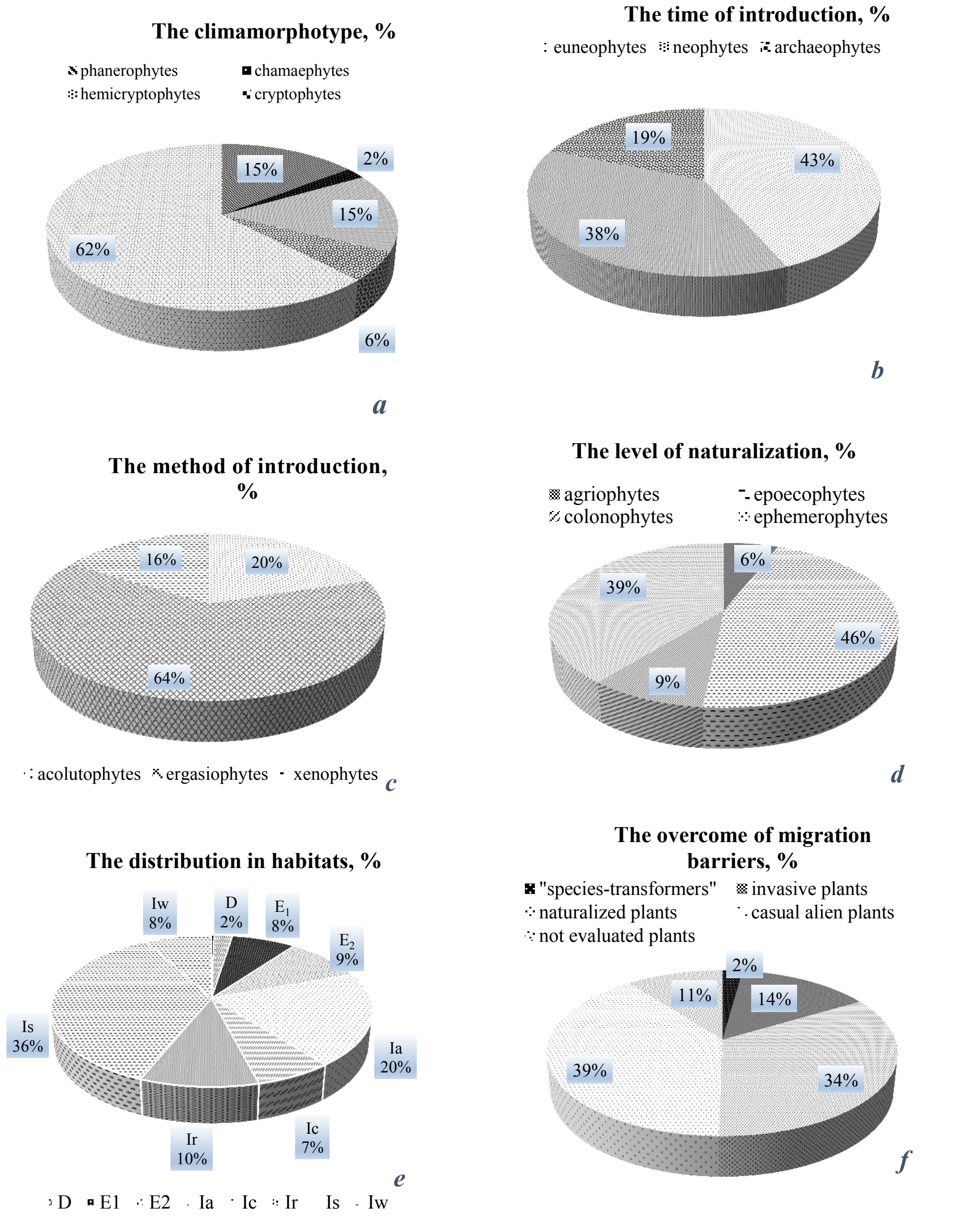

Fig. 1. Typological diversity of alien plant species in agricultural habitats.
(Marked in Fig. 1e: D, wet habitats of grass; E_1, meadow steppe habitats; E_2, true steppe habitats; Ia, cultivated agricultural; horticultural and domestic habitats;Ic, shrub habitats; Ir, ruderal habitats; Is, rural public habitats; and Iw, wayside habitats).

The typological diversity of the studied weed alien species predetermines their adaptive ability, related to their existence in conditions, changed due to human activity. The non-native species are classified into five categories of climamorphotype: therophytes, cryptophytes, hemicryptophytes, chamaephytes and phanerophytes. The prevailing number of therophytes is evident, but the share of phanerophytes is also high here, which is related to the rural public habitats (Fig. 1a).

The prevailing number of neophytes and euneophytes has been observed for chronic-elements (Fig. 1b). As for the method of introduction, the ergasiophytes prevail here also (Fig. 1c). An interesting picture of variability of alien plants is seen at the level of naturalisation, which shows dynamics and instability of the studied fraction of flora. Almost half of the fraction is made up of epoecophytes, which prefer habitats disturbed by human activity. The ephemerophytes have just begun to adapt to new conditions, thus local populations have not yet formed-but are quite diverse. Some species (colonophytes) formed colonies; and only some of them – agriophytes – have their own ecological niche in semi-natural habitats (Fig. 1d).

This review of typological structure of adventive flora with high participation of unstable migration elements is a prerequisite for the accumulation of hidden environmental risks. The majority of ephemerophytes, according to the type of life strategy, are explerents that are capable of producing mass diasporas and rapidly increasing the population, occupying and retaining an additional territory.

The distribution of alien species in agricultural habitats was taken to presence–absence in three types of habitats according to EUNIS: type *D*, wet habitats of grass; type *E*, grassland habitats: grasslands, steppes and wasteland, 2 habitats; and type *I,* habitats shaped by human activity, 19 habitats. (Fig. 1e shows only habitats in which more than 2% of all the alien species are present).

In terms of overcoming migration barriers of natural character (spatial, climatic, abiotic, biotic, reproductive and the like), the alien species in agricultural habitats are presented by all five categories (Fig. 1f).The casual alien plants dominate along with the naturalised but not common species. Yet virtually invasive plants on the list are more than a hundred, and they represent an environmental risk for local plant diversity. The invasive plant category includes hard-to-eradicate weeds, for example, *Amaranthus albus* L., *A. retroflexus* L., *Ambrosia artemisiifolia* L., *Apera spica-venti* (L.) P. Beauv., *Bromus squarrosus* L., *Centaurea diffusa* Lam., *Cyanussegetum* Hill., *Cynodon dactylon* (L.) Pers., *Descurainia sophia* (L.) Webb ex Prantl, *Diplotaxis tenuifolia* (L.) DC., *Echinochloa crus-galli* (L.) P. Beauv., *Iva xanthiifolia* Nutt., *Galinsoga parviflora* Cav., *Lactuca sativa* L., *L. serriola* L., *Setaria verticillata* (L.) P. Beauv., *S. viridis* (L.) P. Beauv., *Sinapis alba* L., *Sonchus arvensis* L.**,** *S. asper* (L.) Hill.**,** *S. oleraceus* L., *Tripleurospermum inodorum* (L.) Sch. Bip., *Rhaponticum repens* (L.) Hidalgo. These include alien species of plants with a heterotrophic type of food – obligate plant parasites – such as *Cuscuta campestris* Yunck., *C. cesatiana* Bertol., *C. epilinum* Weihe, *C. gronovii* Willd. ex Roem. & Schult., *C. suaveolens* Ser., *C. tinei* Insenga, *Orobanche brassicae* (Novopokr.) Novopokr., *O. cernua* Loefl., *O. gracilis* Sm., *O. minor* Sm. and *O. ramose* L. Some weeds occur exclusively in certain field habitats, for example, *Monochoria korsakowii* Regel & Maack litter rice fields or *Sorghum halepense* (L.) Pers. is noted from time to time in the vineyards.

But the most threatening group is that of 16 'species–transformers', including *Acer negundo* L., *Bromus tectorum* (L.) Nevski, *Elaeagnus angustifolia* L., *Grindelia squarrosa* (Pursh)

Dunal and *Salix* × *fragilis* L. They are capable of changing the nature of the ecosystem. 'A Unified Classification…' (Blackburn et al., 2014) involves the use of certain principles for assessing the response of an ecosystem to the impacts of alien plant species (Fig. 2).

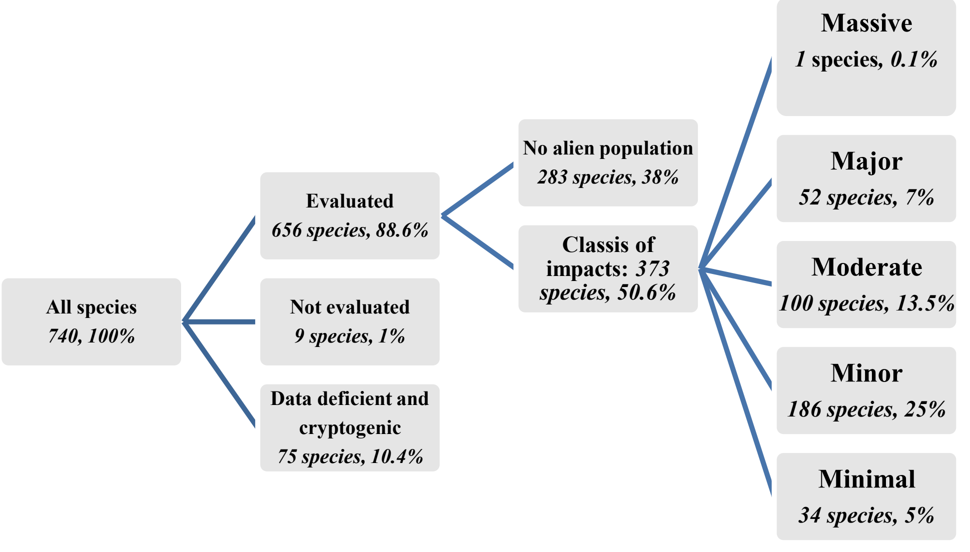

Fig. 2. Scheme of the relations between different categories of impact of alien species.

The presence of alien species of the class of greatest impacts (class of 'massive' impact) in the ecosystem has profound consequences: local extinction of native species and irreversible changes in the community composition. 'Even if the alien species is removed, the system does not recover its original state' (Blackburn et al., 2014). In our case, such an example may be found as a result of hybridisation of a native species *Salix alba* L. and an alien species *S. fragilis* sometimes noted in agricultural habitats.

The class of 'major' impact includes species that cause local extinction of at least one native species and lead to reversible changes in the community composition. Significant effects are caused by species recognised as 'transformers'. For example, *Erigeron annuus*, *E. canadensis*, *E. strigosus* var. *septentrionalis* and *Solidago canadensis* sometimes 'transform' meadow steppe habitats (E_1). The class of 'moderate' impact includes the alien species whose appearance causes a decrease in the density of populations of local species but does not change the composition of the community. These changes are reversible, for example, *Artemisia annua* L., *Bidens frondosa* L., *Conium maculatum* L. and *Neslia paniculata* (L.) Desv. The class of 'minor' impact includes the alien species whose presence slows down the individual state, viability and growth; the renewal is impaired; the sustainability of natural species is reduced, but the density of their populations does not change. The examples include *Abutilon theophrasti* Medik. and *Bryonia alba* L.

The alien species of the class of insignificant impacts (class of 'minimal' impact) cause minimal consequences, and it is unlikely that they have tangible impact on the environment: *Anagallis arvensis* subsp. × *foemina* (Mill.) Schinz & Thell., *Viciaervilia* (L.) Willd. For a 10th part of species, there is a lack of information (deficient data and cryptogenic species): the impact of alien species that has not been demonstrated because of the inadequate study of them under new conditions. A group of the so-called 'cryptogenic species' was discovered with an indefinite status of impact on local diversity (*Orobanche aegyptiaca* Pers., *Setaria pumila* (Poir.) Roem. & Schult., *Sorghum* × *almum* Parodi).

Discussion

On the basis of the above assessment and considering the qualitative and not quantitative characteristics, the impact of all the alien species in the studied habitats is defined as reversible, and after the removal of non-native species, the ecosystems are capable of self-revival. The exception is found in the alien species *Salix fragilis*, which is prone to genetic absorption of the native species *S. alba*.

The species composition of phytobiotic contamination of agricultural habitats in Ukraine coincides with the composition of alien plants in European countries. This demonstrates the European level of plant invasions. It is estimated that more than 13% (129 out of 740 species) are on the list of 150 non-native plant species that have spread to 49 European countries (Lambdon et al., 2008). There their status is estimated as 'naturalised alien plants' or 'casual alien plants'. The highest rank on distribution is occupied by species from agricultural habitats such as *Erigeron canadensis* in 47 countries, *Datura stramonium* L. agricultural in 45 countries, *Amaranthus retroflexus* L. and *Galinsoga parviflora* Cav. in 44 countries, *Helianthus tuberosus* L. and *Xanthium strumarium* L. in 43 countries, *Oenothera biennis* and *Robinia pseudoacacia* in 42 countries, *Galinsoga quadriradiata* Ruiz & Pav., *Matricaria matricarioides*, *Panicum miliaceum* L. and *Veronica persica* Poir. in 41 countries. Total in 40 or more countries there are 17 species, in 39–21 species, respectively, in 35–43 species, in 30 countries – 90 species.

According to our results, the share of invading species in the local flora of the Nature Reserve Fund of the Flatland Part of Ukraine varies from 8 to 24%; the background phytobiotic contamination is 8−10% (Burda et al., 2015). Obviously, the phytobiotic contamination of agricultural habitats is several times higher than the background (65% vs. 8−10%). This confirms our assumptions about the existence of a sufficiently large threat from agricultural habitats for local biotic diversity. Our results contain the applied aspect. They determine the priorities of construction of strategy for the prevention of invasive plants in the agrarian habitats.

Let us emphasise that the solution to the problem of phytobiotic pollution of agricultural habitats in our country depends not only on the application of appropriate technical measures for timely and quality carebut it also lies on the plane of Ukraine's compliance with the requirements of the Global strategy on invasive alien species (McNeely et al., 2001). The differentiation given in the paper on the classes of impacts of alien species on the environment is the basis for prioritising the national strategy on invasive alien plant species. To implement the European policy on invasive alien species, Ukraine should adopt the national strategy on invasive alien species, approve national and regional lists of them, introduce local codes of recommendations and so on.

Conclusion

The taxonomic richness of the adventive fraction of flora in agricultural habitats in Ukraine is 740 species from 362 genera and 79 families. The typological diversity of the fraction determines its adaptive capacity for specific habitats with regular human intervention. Amongst the climamorpho-types, the therophytes predominate and amongst the chronic-elements, the neophytes do; by the method of distribution, the ergasiophytes predominate and by the level of naturalisation, the ephemerophytes. This typological structure demonstrates the instability of the studied fraction of the flora, and a high percentage of the life strategy of the explerents in it indicates hidden environmental risks.

According to the composition of alien plants common for other European countries, phytobiotic pollution of agricultural habitats in Ukraine proves to be similar to some extent: 129 out of 740 identified weed species are mentioned amongst 150 non-native species common for 49 European countries. Unstable random species dominate or overcome natural migration barriers. There are more than a hundred of invasive species, and a group of 16 'species-transformers' is of specific threat.

Alien species in agricultural habitats affect the environment significantly: more than one-fifth of them have 'moderate' or higher classes of impacts, one-third haveclasses of 'minor' and 'minimal'impacts; about 40% have not yet created their local populations. In general, the impacts of all invasive alien species on these habitats are assessed as having a reversible character. An exception is *Salix fragilis*, which absorbs local *S. alba* sometimes.

The main reason for the intensification of expansions of invasive alien species in Ukraine in the past decades is anthropogenic. The main vectors of alien species penetration are well-known: from a culture. The main number of species occupiedthe agricultural habitats with the intensive annual care of cereal crop fields, infertile crops fields, perennial herb fields and habitats of rural settlements. In determining the priorities for building the national strategy on invasive alien species, it is important to consider the role played by the outlined classes of environmental impacts of invasive alien species living in agricultural habitats.

The solution to the problem of phytobiotic contamination of agricultural habitats in Ukraine lies in the implementation of the European policy on invasive alien species (adoption and introduction of national and regional documents on invasive alien species of legislative and recommendatory nature).

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References

Blackburn, T.M., Essl, F., Evans, T., Hulme, P.E., Jeschke, J.M., Kuhn, I., Kumschick, S., Markova, Z., Mrugała, A., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D.M., Sendek, A., Vila, M., Wilson,

J.R.U., Winter, M., Genovesi, P. & Bacher S. (2014). A Unified classification of alien species based on the magnitude of their environmental impacts. *PLOS Biology*, 12(5), 1–11. DOI: 10.1371/journal.pbio.1001850.
Burda, R.I., Golivets, M.A. & Petrovych O.Z. (2015). Alien species in the flora of the Nature Reserve Fund of the Flatland Part of Ukraine. *Russian Journal of Biological Invasions,* 6(1), 6–20. DOI: 10.1134/S2075111715010038.
Davies, C.E., Moss, D. & Hill M.O. (2004). *EUNIS habitat classification. Revised 2004.* Final Report to the European Topic Centre on Nature Protection and Biodiversity. European Environment Agency.
Didukh, Ya., Plyuta, P., Protopopova, V., Ermolenko, V., Korotchenko, I., Karkutsiev, G. & Burda R. (2000). *Ecoflora of Ukraine*. 1. Kyiv: Fitosotsiosentr.
Didukh, Ya.P., Mala, J.I., Pashkevich, N.A., Fitsailo, T.V. & Khodosovtsev O.E. (2016). *Biotopes of the Crimean Mountains.* Kyiv: Interservis.
Lambdon, P.W., Pyšek, P., Basnou, C., Hejda, M., Arianoutsou, M., Essl, F., Jarošík, V., Pergl, J., Winter, M., Anastasiu, P., Andriopoulos, P., Bazos, I., Brundu, G., Celesti-Grapow, L., Chassot, P., Delipetrou, P., Josefsson, M., Kark, S., Klotz, S., Kokkoris, Y., Kühn, I., Marchante, H., Perglová, I., Pino, J., Vila, M., Zikos, A., Roy, D. & Hulme P.E. (2008). Alien flora of Europe: Species diversity, temporal trends, geographical patterns and research needs. *Preslia,* 80, 101–149.
McNeely, J.A., Mooney, H.A., Neville, L.E., Schei, P.J. & Waage J.K. (Eds.) (2001). *Global strategy on invasive alien species.* Switzerland and Cambridge: IUCN, Gland.
Mosyakin, S.L. & Fedoronchuk M.M. (1999). *Vascular plants of Ukraine: A nomenclatural checklist*. Kiev: M.G. Kholodny Institute of Botany, Missouri Botanic Gardens.
Mosyakin, S.L. (2013). Rodyny i porjadky kvitkovyh roslyn flory Ukrainy: pragmatychna klasyfikaciia ta polozhennia u philogenetychnyi systemi. *Ukrainian Botanical Journal,* 70(3), 289–307. DOI: 10.15407/ukrbotj70.03.289.
Richardson, D.M., Pyšek, P., Rejmanek, P.M., Barbour, M.G., Panetta, F.D. & West C.J. (2000). Naturalization and invasion of alien plants: Conception and definitions. *Divers. Distrib.,* 6, 93–107. DOI: 10.1046/j.1472-4642.2000.00083.x.
The International Plant Name Index (IPNI). Retrieved from http://www.ipni.org/index.html.
The Plant List. A working list of all plant species. Retrieved from http://www.theplantlist.org.

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QUANTIFYING THE VEGETATION HEALTH BASED ON THE RESILIENCE IN AN ARID SYSTEM

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Abstract

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Proper management of natural ecosystems is not possible without the knowledge of the health status of its components. Vegetation, as the main component of the ecosystem, plays an important role in its health. One of the key determinants of vegetation health is its resilience in the face of environmental disorders. This research was conducted in parts of the Namakzar-e Khaf watershed in Northeast of South Khorasan Province with the aim of quantifying the vegetative resilience on behalf of the ecosystem health in response to long-term precipitation changes. First, the annual precipitation standardization was performed during a thirty-year period by the SPI method. Then, the average variation in TNDVI index obtained from the Landsat satellite images was examined and the resilience was tested by calculating the four effective factors (amplitude, malleability, damping and hysteresis). According to the results, the amplitude in the survey period was 6.04% and the vegetation has had different values of damping over the years. The most prominent example of vegetation resilience occurred between 1986 and 1996, with malleability of 0.7 and damping of zero. Vegetation in this period, after the elimination of drought effects (1986), has not only returned to the amount of vegetation of reference year with severe precipitation (1996) but also increased by 0.25%. This increase, as the index of hysteresis, has been presented for the first time in the ecosystem health discussion quantitatively in the present study. A set of quantitative calculations showed that despite reduced annual precipitation and drought events, the vegetation has been able to maintain its resilience, which indicates the health of vegetation in the studied ecosystem.

Key words: ecosystem health, resilience, hysteresis, quantitative study, remote sensing.

Introduction

Vegetation resilience is an important factor in ecosystem health, and a variety of environmental disorders can disrupt it. The term resilience was first introduced in 1970 in the study of ecological systems (Holling, 1973) and demonstrates the ability of the ecosystem to maintain its performance in the face of environmental disorders (Elmqviste et al., 2003). A resilience-based system is not only equipped with a disorder adjustment mechanism but also has the potential to benefit from changes in a way that lead to creating an opportunity for development, innovation, and updating (Rockstrom, 2003). Therefore, when a change occurs, the resilience provides the needed conditions for restarting and reorganization (Gunderson, Holling, 2002; Friend, Moench, 2013). If this goes beyond the disturbing forces, the system will have the power to return to maximum vegetation density with least erosion effects, otherwise the system will be vulnerable to the change that was created and could be controlled (Kasperson R.E., Kasperson J.X., 2001). These predictions have been confirmed and proven in the recent field studies by Abrahams et al. (1995), as well as Puigdefábregas and Sánchez (1996). Natural calamities such as flood, earthquake and drought change the process of evolution and the sequence of nature. In contrast to these disorders, the nature has complex and adaptive methods to deal with such occurrences (Holling, 1986). But quantifying the effect of these occurrences can be difficult because the current state of the system depends on its previous state (Carpenter et al., 2001). However, examining the characteristics of the ecosystem on large spatial and temporal scales is one way that can be used to track the effects of disorders in the landscapes. Ecosystem indicators such as vegetation status can be analysed using information, software, and spatial methods (such as remote sensing techniques) to measure the resilience (Washington-Allen et al., 2008). Vegetation is one of the main components of natural ecosystems, where degradation can be observed as loss of diversity and change in vegetation type compared to the initial state of the landscape density (Ringrose et al., 1990; Goheen et al., 2007). Normalized difference vegetation index (NDVI) is useful particularly for measuring the amount of photosynthesis of biomass in semi-arid and savannah areas (Martiny et al., 2006), where vegetation is not high (Richard, Poccard, 1998). At the same time, there are potential limitations in using the NDVI index in these areas, depending on the type and reflection of the soil (Farrar et al., 1994). The provision of additional facilities (MSS and TM Landsat multidimensional sensors) have provided the appropriate spatial scale as the ideal point for comparative regional research for the analysis of long-term changes (Griffiths, Philippot, 2013). These limitations are now minimized in Landsat 7 and 8 satellites with ETM +, OLI, and TRIS sensors and image quality enhancement. Griffiths and Philippot (2013) examined the soil health by examining the status of microbes available in it under environmental and managerial disorders, and soil contamination with heavy metals qualitatively, and concluded that the soil stability and resilience do not depend on the percentage of microbial species, but on the functional characteristics in the soil structure. Westman and O'Leary (1986) have developed 4 criteria of environmental malleability, including the time to return to the reference, amplitude, malleability and damping. In this study, the response of coastal plants in southern California was used against fire disorder. This study was carried out using the data obtained from field observations and a simulation model. The values of the criteria were taken using graphic charts. Washington-Allen et al. (2008) and Cui et al. (2013) examined the vegetative resilience qualitatively by plotting the mean-variance plot in Bolivia, in America and South Africa, respectively. Also, Pricope et al. (2015) examined the changes in the vegetation of Savannah's landscape due to the flood and fire in South Africa qualitatively, but it is only a history of quantifying the ecological resilience using satellite images and exact mathematical equations related to the study by Washington-Allen et al. (2008) on the vegetation in South Africa. This study was conducted on the total vegetation and various cover classes using TNDVI Vegetation Index and Landsat satellite images from 1972 to 1987. Three factors of amplitude, malleability and damping were examined and compared in this study on the vegetation. Among the existing vegetation classes, the meadow showed the highest amount of resilience and malleability compared to the other vegetation and was introduced as a key source and indigenous vegetation survived from the climate change in the region.

Material and methods

The study region with an area of 2193 km^2 is located at 33° 19'–33° 22' N, 60° 23'–60° 40'E in the Northeast of South Khorasan Province (Fig. 1). The climate of the region based on the Amberge categorization is cold and arid, and the average annual precipitation is 160 mm. Within the scope of the study, three uses of pasture, forest and woodland can be distinguished; in each of them, according to the soil and altitude conditions, herbaceous, bush, tree and shrub species are established. For example, you can see herbaceous species of *Artemisia* sp., *Ephedra strobilacea, Astragalus* sp., *Stipagrostis plumosa* and some saline soil like *Seidlitzia rosmarin, Salsola tomentosa* in pastures. Forest species include *Haloxylon ammodendron, Tamarix* sp.*, Amygdalus scoparia, Pistacia atlantica*, *Ammodendron persicum*, which have created the thin forests of the study area.

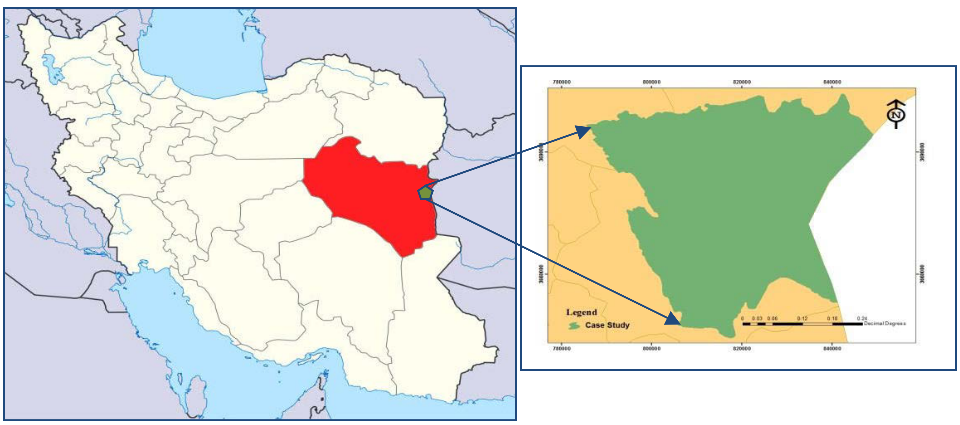

Fig. 1. Location of study area.

Using the daily precipitation data from 15 meteorological stations around the study area (Fig. 2), annual precipitation was extracted over the course of thirty years (1986–2015). Then, the data were interpolated in software ArcGIS for the study area using IDW method and by standard precipitation index (SPI), dry, wet and normal years were determined. In order to complete the years without any precipitation data of each station, correlation with neighbouring stations and software SPSS were used.

From the Landsat 5 and 7 satellite images' archives of the study area, for each year of the thirty-year study period, an image was taken for the period starting from June 15 to July 15, with permanent vegetation under the best growing condition. Finally, 24 images were available (Table 1).

Using the software ENVI and ArcGIS, after making the required satellite images' corrections, NDVI and TNDVI vegetation indices' maps were provided for each image. NDVI is a normalized vegetation difference index used in analyses, remote sensing measurements and vegetation assessment of an area, and its numerical value varies from 1 to 1+ in satellite images per pixel (equation 1).

This index is calculated using equation 1, where R is the red band of the electromagnetic spectrum and NIR is the infrared band near the electromagnetic spectrum.

$$NDVI = NIR - R / NIR + R \quad (1)$$

Removal of negative values and increasing the numerical range of this index was done with the aim of facilitating quantitative examination and analysis in the process of long-term vegetation changes, by converting NDVI index to TNDVI index with a range from zero to 100. In all the subsequent studies, TNDVI vegetation index was used. TNDVI vegetation index is calculated using equation 2.

$$TNDVI = 50\ (NDVI + 1) \quad (2)$$

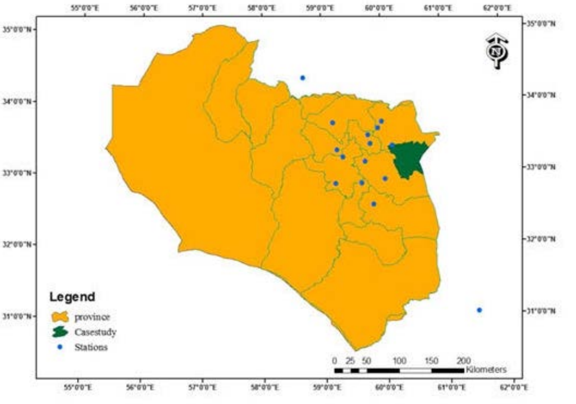

Fig. 2. Location of meteorological stations in study region.

T a b l e 1. Characterize of satellite images.

Scanner	Satellite	Study Term
TM	Landsat 5	1986–1999
ETM+	Landsat 7	2000–2015
NO Image	1988–1994–1995–1997–1999–2006	

In order to examine and quantify ecologically resilience-related changes, it is necessary to recognize factors such as amplitude, malleability, damping and hysteresis. These concepts and methods of calculating them are presented by Washington-Allen et al. (2008), which will be mentioned later in the article. After preparing the TNDVI vegetation index maps for each year using the software ArcGIS, the average values of this index were extracted in each image and all calculations were manually performed using this parameter. The unit of measurement of the average vegetation index of TNDVI and all factors of vegetation's resilience is %.

1. The factor amplitude = $|\Delta TNDVIA|$

The amount of change after the disorder stage, or simultaneously that the system may return to the reference stage or not, and is calculated according to equation 3, where D is the year of occurrence of the disorder and R is the reference year.

$$|\Delta TNDVI\ A| = TNDVI\ D - TNDVI\ R \quad (3)$$

2. The factor malleability = $(\Delta TNDVI\ M)$

The amount of vegetation compensation (recovery) in one or a few years' average after the removal of disorder and calculated according to equation 4.

$$\Delta TNDVI\ M = |\Delta TNDVI\ A| - (\{TNDVI\ (\text{year of removal of the disorder effect}), \ldots TNDVI\ (\text{date}+1)\} - TNDVI\ R) \quad (4)$$

Washington-Allen et al. (2008) in their study stated that the return to the reference state occurs, if the factor of malleability is equal to or smaller than zero.

$$(\Delta TNDVI\ M \leq 0)$$

3. The factor hysteresis

Increasing the vegetation beyond the range of the reference state, which is created in the opposite direction of the path of destruction.

4. The factor damping
A part of the change in vegetation that has not returned to the reference state after a disorder.

Our investigations showed that equation 4 does not provide the expected result in the calculation of the factor of malleability. So, we proposed new equations and added additional descriptions. In the research method, modified equations were used.

When a disorder occurs, particularly in the vegetation, the destructive effect of the disorder according to the conditions may be observed in the year of occurrence or delayed in the subsequent year(s). So, the term 'the year of the effect of a disorder' was used instead of the term 'the year of occurrence' in the equations.

To determine how a vegetation returns to the reference after the removal of the disorder effect, equation 5 was used instead of equation 4, where R' represents a recovery. The result of this equation proportional to being zero, positive or negative, represents different conditions of vegetation recovery. It should be kept in mind that this equation does not provide the amount of vegetative malleability.

$$|\Delta TNDVIA| - (\{TNDVIR' \text{ (year of removal of the disorder effect)} - TNDVID \text{ (year of the disorder effect)} \{= (0, +, -) \quad (5)$$

1. If the result of this equation is zero, it indicates the recovery of the vegetation exactly to the reference state.
2. If the result is a negative number, it indicates that not only the vegetation returned to the reference state but also increased in comparison to the reference. This amount of increase is called hysteresis.

In the states of 1 and 2, the numerical value of the factor of malleability is equal to the amplitude.

$$TNDVI\ M = |\Delta TNDVI\ A|$$

3. If the result of the equation is a positive number, it indicates that the vegetation has moved towards the reference state but has not reached it, and the numerical value obtained from equation 5 shows the amount of vegetation damping compared to the reference. In this case, Equation 6 should be used to obtain the malleability value.

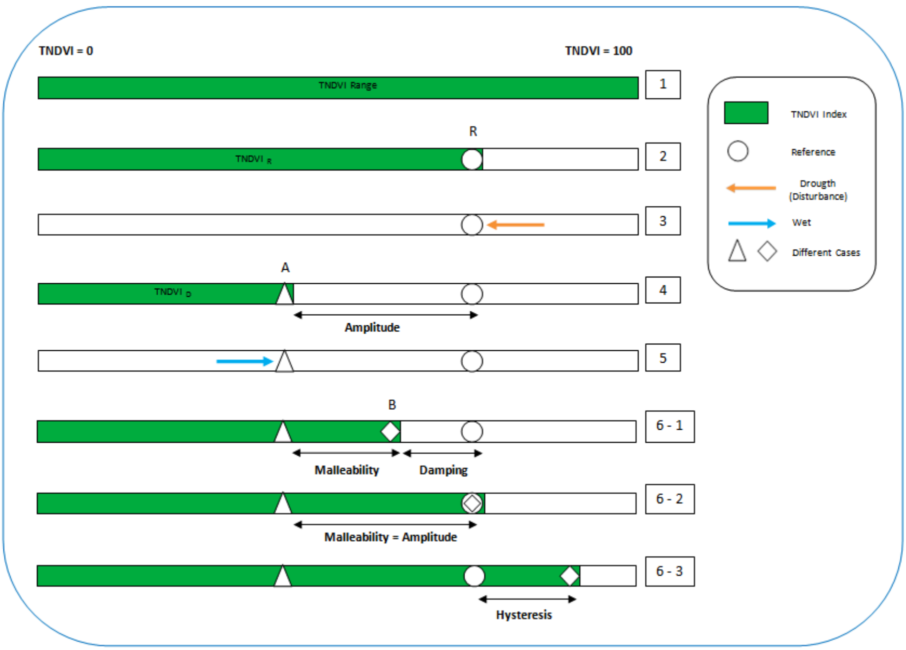

Fig. 3. Conceptual chart for resilience parameters.

TNDVI M = |ΔTNDVI A| - (Damping)   (6)

To calculate the factors of malleability, damping and hysteresis in each disorder occurrence, three years are required for comparison, including reference year, year of the occurrence effect, and average of one year or several years after the occurrence is resolved. The concept of the factors studied and the corrective comments are given in Fig. 3 in the form of a conceptual chart.

Results and discussion

Comparison of annual precipitation variations in the thirty-year series (1986–2015) indicates two approximate wet and dry periods. The wet period is related to the first fourteen years of the time series (1986–1999) and the dry period is related to the last sixteen years (2000–2015) (Fig. 4).

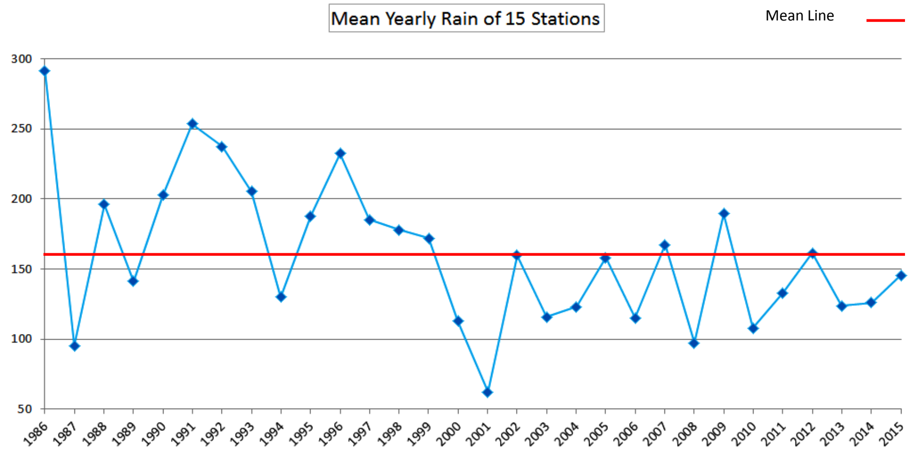

Fig. 4. Mean annual precipitation changes.

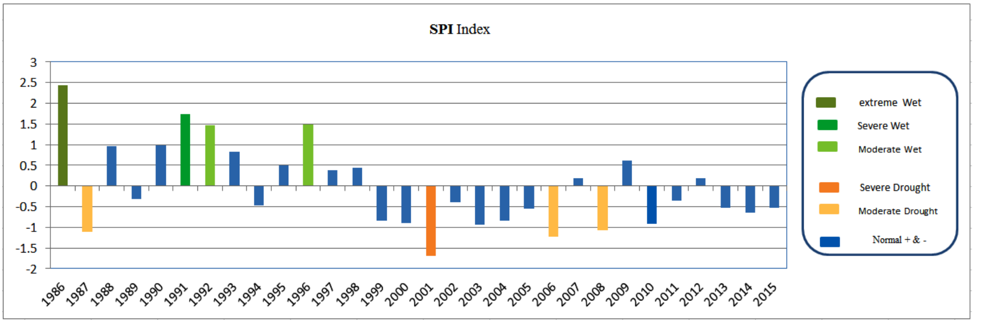

Fig. 5. Comparison of Standard Precipitation Index (SPI) and the severity of events within the study area based on the Interpolation (IDW).

Figure 5 shows the annual precipitation levels in the thirty-year series of the study area based on Standard Precipitation Index (SPI). Precipitation and drought with different intensities as well as the annual precipitation changes in the normal years are occurrences that are given in this Figure.

This table shows 4 precipitation occurrences with different intensities including 1 case of very intense precipitation (1986), 1 case of intense precipitation (1991) and 2 cases of moderate precipitation (1996 and 1992). Also, 4 drought occurrences including 1 case of intense drought (2001) and 3 cases of moderate drought (1987, 2006 and 2008) also occurred. All precipitation occurrences are related to the first half and most drought occurrences are related to the second half of the studied period. According to the distribution of SPI index,

T a b l e 2. TNDVI index values, annual precipitation and SPI index classes of the study area.

Row	Year	Mean TNDVI	annual precipitation	SPI index classes
1	1986	49.37	291.22	Extreme Wet
2	1987	48.96	95.26	Moderate Wet
3	1988	No Image	196.24	Normal +
4	1989	48.96	141.2	Normal -
5	1990	48.85	202.94	Normal +
6	1991	48.73	253.67	Sever Wet
7	1992	48.92	237.51	Moderate Wet
8	1993	49.03	205.36	Normal +
9	1994	No Image	129.97	Normal -
10	1995	No Image	188.02	Normal +
11	1996	49.62	232.48	Moderate Wet
12	1997	No Image	185.2	Normal +
13	1998	49.2	177.88	Normal +
14	1999	No Image	171.9	Normal -
15	2000	47.71	112.97	Normal -
16	2001	43.77	62.38	Severe Drought
17	2002	43.94	159.91	Normal -
18	2003	44.47	115.79	Normal -
19	2004	43.49	123.07	Normal -
20	2005	44.26	158.22	Normal -
21	2006	No Image	115.08	Moderate Drought
22	2007	44.41	167.41	Normal +
23	2008	43.81	97.23	Moderate Drought
24	2009	44.52	189.53	Normal +
25	2010	43.58	108.07	Normal -
26	2011	43.84	132.93	Normal -
27	2012	43.91	161.37	Normal +
28	2013	43.84	123.72	Normal -
29	2014	43.76	126.1	Normal -
30	2015	44.28	145.56	Normal -

precipitation has occurred in years when precipitation has exceeded 232 mm, and drought has occurred with an annual precipitation of 95 mm. The interval between these two values represents the normal state. The number zero in SPI index is equivalent to the average precipitation of 160.3. The value between the average and precipitation is wet normal (normal +) and less than the value to drought occurrence is dry normal (normal -). Precipitation variations, especially in the dry period and drought occurrence in the year of occurrence or delayed, have had undesirable effects on the average vegetation of the study area. Table 2 shows the average amount of vegetation in the study area in 24 years of the thirty years with good satellite images.

In the first case, the vegetation resilience was examined over the entire thirty-year period. The year 1986 was the first year of study reference of the vegetation changes; the year 2010 was the year of decline in vegetation to the lowest amount during the thirty years of study period; and the year 2015 was the final year of the studied period. The amount of vegetation

T a b l e 3. Quantitative examination of the resilience factors in the selected years.

TNDVI resilience factors in 1986–2015 (Whole time series)						
Case 1	Year	Mean TNDVI (%)				
		Amount	Amplitude	Malleability	Damping	Hysteresis
	1986	49.37	-	-	-	-
	2010	43.58	5.79	-	-	-
	2015	44.28	-	0.7	5.09	0
	Legend	Reference = 1986		Reduce in vegetation index = 2010		End of time series = 2015
TNDVI resilience factors in 1986–1996 (Wet time series).						
Case 2	Year	Mean TNDVI (%)				
		Amount	Amplitude	malleability	Damping	Hysteresis
	1986	49.37	-	-	-	-
	1991	48.73	0.64	-	-	-
	1996	49.62	-	0.64	0	0.25
	Legend	Reference = 1986		Reduce in vegetation index = 1999		Returned to the reference = 1996
TNDVI resilience factors in 2000–2003 (Drought time series).						
Case 3	Year	Mean TNDVI (%)				
		Amount	Amplitude	malleability	Damping	Hysteresis
	2000	47.71	-	-	-	-
	2001	43.77	3.94	-	-	-
	2003	44.47	-	0.7	3.24	0
	Legend	Reference = 2000		Reduce in vegetation index = 2001		Returned to the reference = 2003
TNDVI resilience factors in 2003–2009 (Drought time series).						
Case 4	Year	Mean TNDVI (%)				
		Amount	Amplitude	malleability	Damping	Hysteresis
	2003	44.47	-	-	-	-
	2008	43.81	0.33	-	-	-
	2009	44.52	-	0.33	0	0.05
	Legend	Reference = 2003		Reduce in vegetation index = 2008		Returned to the reference = 2009

in 1986 was 49.37%, which, by reduced precipitation, reached the lowest amount of 43.58% in 2010. By increasing the moisture in 2015, it returned to 0.7% of the reference vegetation but did not reach it. Therefore, the amount of vegetation damping in this recovery was 5.09%. Despite this, the vegetation also experienced resilience in this case.

The second case is related to the wet period between 1986 and 1996, in which, the year 1986 is the reference year for the comparison of vegetation changes. In 1991, the vegetation was reduced due to the reduced precipitation and drought. The interval between the value of the reference vegetation and the year of the disorder effect (1991) was the amplitude of the vegetation changes between these two years, and its value during this period was 0.64 percent. In 1996, the conditions had shifted to rising precipitation, resulting in increased vegetation. Vegetation in this year, in addition to being returned to the reference state (0.64%), had increased by 0.25% compared to the reference (the factor hysteresis). In this period, because the vegetation completely returned to the reference, the damping index, the difference between the amplitude and malleability indices, was zero. The third and fourth cases are related to the vegetation recovery to the reference in the dry period, and despite the dominance of dry conditions, the vegetation's recovery to the reference state is also observed. In case 3, the resilience reproved in the dry period between 2000 and 2003, and 2001 was faced with intense drought.

In the fourth case, the vegetation, in addition to returning to the reference state, had 0.05% hysteresis.

It is reminded, among the cases presented, only 4 cases of vegetation changes were in the study period, which were presented in the study area to calculate the quadruple factors and prove the presence of the vegetation resilience in the study area. In the studied period, we can examine more similar cases and calculate the vegetation resilience.

Conclusion

Awareness of the health status of the vegetation and its response to long-term precipitation changes and environmental disorders, such as drought occurrence, ensure the success of the managerial plans for renewable natural resources. A quantitative study of the vegetation resilience was first performed as the control of the ecosystem health in 2008 by Washington-Allen et al. in South Africa with an average precipitation of 402 mm, and was able to show the natural vegetation resilience in the study area by calculating the three factors of amplitude, malleability and damping. Our study is the second study on quantifying the vegetation resilience and the first study under dry climatic conditions in Asia (an average annual precipitation of 160 mm) conducted in Iran by calculating four factors related to the resilience, and is the first study that has presented the factor hysteresis in the calculations. Given the large difference in the average annual precipitation in the two studied regions, both of which are presented as arid climates, the present study can be considered as a unique example in the ecosystem health discussion in regions with similar conditions. Despite the difficult conditions and the consistency, the native vegetation of the study area has been able to return the reference state not only by resolving the disorder relatively, but also it has experienced hysteresis stage. With the presence of such amazing protective and consistent mechanisms in the vegetation of arid regions, it is possible to maintain and restore these regions by proper managerial plans.

References

Abrahams, A.D., Parsons, A.J. & Wainwright J. (1995). Effects of vegetation change on interrill runoff and erosion, Walnut Gulch, southern Arizona. *Geomorphology*, 13, 37–48. DOI: 10.1016/0169-555X(95)00027-3.

Carpenter, S.R., Walker, B.H., Anderies, J.M. & Abel N. (2001). From metaphor to measurement: Resilience of what to what? *Ecosystems*, 4, 765–781. DOI: 10.1007/s10021-001-0044-9.

Cui, X., Gibbes, C., Southworth, J. & Waylen P. (2013). Using remote sensing to quantify vegetation change and ecological resilience in a semi-arid system. *Journal of Land*, 2, 108–130. DOI: 10.3390/land22020108.

Elmqvist, T., Folke, C., Nystrom M., Peterson, G., Bengtsson, J., Walker, B. & Norberg J. (2003). Response diversity, ecosystem change, and resilience. *Frontiers in Ecology and the Environment*, 1, 488–494. DOI: 10.1890/15409295(2003)001[0488:RDECAR]3.0.CO;2.

Farrar, T.J., Nicholson, S.E. & Lare A.R. (1994). The influence of soil type on the relationships between NDVI, rainfall, and soil moisture in semiarid Botswana. II. NDVI response to soil moisture. *Remote Sens. Environ.*, 50, 121–133. DOI: 10.1016/0034-4257(94)90039-6.

Friend, R. & Moench M. (2013). What is the purpose of urban climate resilience? Implications for addressing poverty and vulnerability. *Urban Climate*, 6, 98–113. DOI: 10.1016/j.uclim.2013.09.002.

Goheen, J., Young, T., Keesing, F. & Palmer T. (2007). Consequences of herbivory by native ungulates for the reproduction of a savanna tree. *J. Ecol.*, 95, 129–138. DOI: 10.1111/j.1365-2745.2006.01196.x.

Griffiths, B.S. & Philippot L. (2013). Insights into the resistance and resilience of the soil microbial community. *FEMS Microbiol. Rev.*, 37, 112–129. DOI: 10.1111/j.1574 6976.2012.00343.x.

Gunderson, L.H. & Holling C.S. (2002). *Panarchy: Understanding transformations in human and natural systems*. Washington: Island Press.

Holling, C.S. (1973). Resilience and stability of ecological systems. *Annu. Rev. Ecol. Syst.*, 4, 1–23. DOI: 10.1146/annurev.es.04.110173.000245.

Holling, C.S. (1986). The resilience of terrestrial ecosystems: Local surprise and global change. In W.C. Clark & R.E. Munn (Eds.), *Sustainable development of the biosphere* (pp. 292–320). Cambridge: Cambridge University Press.

Kasperson, R.E. & Kasperson J.X. (2001). *Climate change, vulnerability and social justice*. Stockholm: Stockholm Environment Institute.

Martiny, N., Camberlin, P., Richard, Y. & Philippon N. (2006). Compared regimes of NDVI and rainfall in semi-arid regions of Africa. *Int. J. Remote Sens.*, 27, 5201–5223. DOI: 10.1080/01431160600567787.

Puigdefábregas, J. & Sánchez G. (1996). Geomorphological implications of vegetation patchiness on semi-arid slopes. In M.G. Anderson & S.M. Brooks (Eds.), *Advances in hillslope processes* (pp. 1027–1060). Wiley.

Pricope, N.G., Gaughan, A.E., All, J.D., Binford, M.W. & Rutina L.P. (2015). Spatio-temporal analysis of vegetation dynamics in relation to shifting inundation and fire regimes: Disentangling environmental variability from Land management decisions in a Southern African transboundary watershed. *Land*, 4, 627–655. DOI: 10.3390/land4030627.

Richard, Y. & Poccard I. (1998). A statistical study of NDVI sensitivity to seasonal and interannual rainfall variations in Southern Africa. *Int. J. Remote Sens.*, 19, 2907–2920. DOI: 10.1080/014311698214343.

Ringrose, S., Matheson, W., Tempest, F. & Boyle T. (1990). The development and causes of range degradation features in southeast Botswana using multi-temporal Landsat MSS imagery. *Photogrammetric Engineering and Remote Sensing*, 56, 1253–1262.

Rockström, J. (2003). Resilience building and water demand management for drought mitigation. *Physics and Chemistry of the Earth*, 28, 869–877. DOI: 10.1016j.pce.2003. 08.009.

Washington-Allen, R.A., Ramsey, R., West, N.E. & Norton B.E. (2008). Quantification of the ecological resilience of drylands using digital remote sensing. *Ecology and Society*, 13, 33–37. http://www.ecologyandsociety.org/vol13/iss1/art33/

Westman, W.E. & O'Leary J.F. (1986). Measures of resilience: the response of coastal sage scrub to fire. *Vegetatio*, 65(3), 179–189. DOI: 10.1007/BF000444818.

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INCLUSION OF THE PUBLIC IN THE NATURAL CAPITAL, ECOSYSTEM SERVICES AND GREEN INFRASTRUCTURE ASSESSMENTS (RESULTS OF STRUCTURED INTERVIEWS WITH STAKEHOLDERS OF COMMUNE LIPTOVSKÁ TEPLIČKA)

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Abstract

Moyzeová M.: Inclusion of the public in the natural capital, ecosystem services and green infrastructure assessments (results of structured interviews with stakeholders of commune Liptovská Teplička). Ekológia (Bratislava), Vol. 37, No. 1, p. 42–56, 2018.

Nowadays, topics like natural capital assessment, ecosystem services and green infrastructure have become frequent subjects of a number of national and international projects accomplished on local, regional, national and cross-frontier levels. These projects respond to the deterioration of biotopes due to their fragmentation and degradation as a result of constructions and tourism/recreation. This situation requires an economic assessment of ecosystems from the view point of their capacities to satisfy human necessities with simultaneous conservation of the environmental quality, and the optimal status of landscape diversity both in rural and urban areas. The aim of the Green Infrastructure initiative is to stop the loss of land as an irreplaceable natural resource and to contribute to the inclusion of ecological and sustainability aspects into the spatial planning and regional development in rural and urban areas. Green Infrastructure is the tool that may reduce the loss of ecosystem services connected with future occupation of land and improve functions of land. It may support ecological measures aimed at conservation of agricultural landscape and adoption of measures in the sphere of forest and water economies. Important role in the assessment of ecosystems is played not only by the scientists but also by experts and the public at large. This is the reason why ever more stakeholders possessing knowledge of local territory and personal life experience participate in these projects. Their judgments and views, often bearing information important for the above-mentioned assessment, are applied to proposed measures aimed at the improvement of environmental quality and quality of life in terms of sustainability. This article brings the possible example of how to include a selected sample of stakeholders into the assessment of natural capital and ecosystem services on local level in the frame of Green Infrastructure. The aim of this paper is to analyse attitudes of the involved for the evaluation of natural capital and ecosystem services at a local level by means of structured interviews. Obtained views will be applied for the assessment of ecosystem services and proposals aimed at protection and conservation of natural capital and building of green infrastructure. The research was carried out in the model territory of the rural commune Liptovská Teplička.

Key words: rural settlements, natural capital, green infrastructure, sociological research, stakeholders.

Introduction

Humans are not only part of nature their activities continuously influence components of nature. Effects of human activities on landscape are dual. Human activities connected with nature conservation and rational exploitation of natural, cultural and historical sources are positive, while the ever more expanding and continuous economic growth and prosperity are negative. Disruption, even devastation of landscape, threaten the very human existence as the benefits that nature brings to humans in the form of healthy food, clean air and water and raw materials diminish. It is the reason why solutions and measures bound to mitigate negative effects of human activities on landscape are sought in order to stop the exhaustion of the capacity of ecosystems to fulfil their services indispensable for human life and existence. According to the Resource Efficient Europe plan (COM, 2011a), the deficient protection of natural capital and the underestimation of the value of ecosystem services will have to be tackled by measures supporting intelligent, sustainable and inclusive growth as a priority of the EU in the frame of the document Strategy Europe 2020 (COM, 2010). Investment into green infrastructure is one of the important steps for the protection of natural capital. In the field of biodiversity, the Strategy Europe 2020 (COM, 2011b) is the commitment of the Commission to create the strategy for green infrastructure as a tool providing for ecological, economic, and social benefits by natural solutions. Green infrastructure is based on the principle that conservation and improvement of nature and natural resources should be included into spatial planning and territorial development. Among numerous definitions available, the one by the European Commission (COM, 2013) defines green infrastructure as a strategically planned network of natural and semi-natural areas with varied environmental properties that are created and managed in a way providing a wide scale of ecosystem services. It includes green places (or blue ones in cases of water ecosystems) and additional physical characteristics in dry land (including coastal areas) and the sea areas. Green infrastructure on the mainland is in rural and urban areas.

Natural capital, as defined by OECD (2011) and WB (2012), represents natural resources providing a flow of rare commodities and services. Among the principal components of natural capital are farmland, mineral deposits (crude, gas, coal and minerals), forest, water, fishing areas, and atmosphere. Commodities and services of natural capital are indispensable for economic growth, they provide inputs for agriculture, industry and services, and they increase productivity of agriculture and reliability of services of infrastructure by control of climate. Ecosystem services (services of natural capital) ensue from functions of ecosystems. Ecosystem functions are productive (P) involving production of food, fibres and wood mass, regulative (R), that is, regulation of erosion, sequestration of carbon, and sanitary functions, and cultural, informative, spiritual, and supportive (C). In case these functions (e.g., productive, that is, production of biomass and crops) are financially assessable or if they influence human health (sanitary functions) or economic prosperity, they are referred to as ecosystem services. Regarding the fact that many of these services have still not been financially assessed, they are not sufficiently taken into account by those who prepare economic models (RF, 2010).

Natural and cultural heritage is an integral part of the territorial capital and identity, not only of the EU but also of individual member states, their cities and communes. Excessive use of resources is considered a threat to the territorial development. According to the European Commission (COM, 2013), the aim of green infrastructure is to harmonise human activities with natural environment and to boost the possibilities of socio-economic development in local communities when the provision of basic commodities and services must be accompanied by conservation of physical properties of ecosystems and landscape identity. What is the role of the public in this task? Apart from the Aarhus Convention (1998), which talks about the participation of the public in matters concerning the environment, there are also many examples of addressed and accomplished projects. The Institute of Landscape Ecology was involved with the national and international projects such as *The Assessment of Ecosystem Services in Historical Structures of Farmland* and the *OpenNESS* Project. According to the European Economic and Social Committee (COM, 2013), inclusion of citizens and organisations into the green infrastructure projects, application of the bottom-up approach and support to partnerships with the representatives of communes, industries, agriculture, forest and water economies, nature conservation, and the NGOs focused on nature conservation and the environment is crucial. This aim, the same as raising of awareness of green infrastructure amongst the main groups of stakeholders was also the one task of our research focused on the ecosystem services and natural capital assessment. The research was conducted in the model territory of rural settlement Liptovská Teplička.

Material and methods

This assessment of ecosystems and ecosystem services drew on the experience obtained from work on the international project of the EU 7th Framework Programme OpenNESS: *Operationalisation of Natural Capital and Ecosystem Services: from Concept to Application*. The principal objective of this research was to generate operational frame for the application of the concept *natural capital* (sum of natural elements and their utility properties for human) and that of *ecosystem services* (benefits for humans obtained from ecosystems, that is, the live and inanimate nature) in the process of landscape and urban planning, territorial management, and natural resources in the European countries. In the assessment of the natural capital (NC) and ecosystem services (ES), the cascade model (Potschin, Haines-Young, 2013) and the most recent classification of ES by CICES (2013), which discerns three types of ES in landscape: producing (P), regulating (R) and cultural (C) ecosystem services, were applied. Close cooperation of project workers with prominent representatives of relevant public and private institutions and businesses played an important role in this process.

Research aims agreed with those of the European Commission (COM, 2013), that is, improvement of knowledge about ecosystems, estimation of their status, and provision of ecosystem services. Local stakeholders were invited to participate in the assessment. They analysed feasibility and application of NC and ES in the management of the territory with an emphasis on conservation of biodiversity, rational use of natural resources, improvement of water quality, and environmental quality.

The assessment of NC in commune Liptovská Teplička has been carried out by means of structure interviews. Sociological survey was accomplished in 2014 and 2015. Conceptual analysis (Prokša et al., 2008; Chrenščová, 2011) was used for summarization, categorization, and interpretation of viewpoints. The purpose was to find out to what extent the addressed sample of respondents realizes the existence and significance of ecosystem services, natural capital and green infrastructure for the development of the territory from the point of view of their individual professions. Selection of respondents followed the rule that they should represent the basic groups of local stakeholders. They were owners, users, and administrators of plots, local self-administration officials, private farmers, forest managers, water managers, state conservationists, and members of local amateur organisations.

They were asked the following questions:

Which ecosystems are typical for the cadastral territory of Liptovská Teplička? What functions are fulfilled by these ecosystems? What are their benefits for humans and their existence? Did respondents notice any changes, if there were any, and what positive or negative did they bring?

Study area

Administratively, the commune Liptovská Teplička (central part of village – GPS coordinates 48^0 58´ 02.6" N 20^0 05´21.2" E) is part of the administrative Region Prešov, district Poprad (Fig. 1). It is located at 846 to 1429 m above sea level. The area of its cadastre is 9,845.87 ha. Share of males and females in its total population of 2,394 (as of 31 December 2016) is 1,174 and 1,220 respectively.

The arable land-meadow-pasture pattern of local landscape is typical for well-developed agriculture and forest economy. Among the land use elements, forest plots with total area of 7,996.61 ha (81.22%) dominate, followed by farmland (1,682.95 ha, 17.09%), water bodies (54.88 ha 0.56%), built-up areas (96.12 ha 0.98%), and others (15.32 ha 0.16%) (Fig. 2). Variegated character of natural conditions along with historical, cultural and socio-economic factors determines a number of ecosystem services.

Fig. 1. Localization of Liptovská Teplička village within the Slovak Republic.


Fig. 2. View of the commune Liptovská Teplička (M. Moyzeová, 2014).

Results

Agriculture (Shared Agricultural Cooperative members, private farmers)

Management of farm land or the use of forest greatly influences the status of NC in the cadastral territory of Liptovská Teplička. Tools and measures aimed at supporting of green infrastructure and expansion of areas with high natural value are provided in the frame of the Common Agricultural Policy and the Rural Development Policy (RDP). It concerns the extensive direct support for farmers in the frame of the first CAP pillar in order to prevent land abandonment and fragmentation, and some smaller measures via the RDP programs in the second pillar including unproductive investments, agro-environmental measures (such as conservation of farming landscape, conservation and improvement of hedges, buffer zones, terraces, dry walls, measures in the field of forest and pasturing management, etc.) payments encouraging the contact with the *Natura2000* network, cooperation in conservation of valuable field edges and conservation and restoration of rural heritage features (COM, 2013). The European Commission included some additional ecologizing aspects in its proposals of the common agricultural policy reform. Apart from other, there is the request that the farmers who accept payments in the frame of the first pillar should conserve the existing permanent grassland and that 7% of arable land and land containing permanent crops should be the area of ecological interest (COM, 2011c). If correctly applied, these measures may contribute to green infrastructure. How does farming develop in Liptovská Teplička?

Delimited soil types or subtypes and their complexes were classified by their production potential (Vadovičová, Džatko, 1992) to the top producing soils: typical Fluvisol, complex of typical Fluvisol and gleyic Fluvisol. Productive soils: Rendzina-Cambisol, Cambisol-Rendzina and their complexes, typical Cambisol, Rendzina-Cambisol, Rendzina complex Rendzina and Cambisol-Rendzina. Less productive soils: typical Cambisol, Dystric Cambisol, complex typical Cambisol and Dystric Cambisol. Least productive soils: complex typical Cambisol, Dystric Cambisol and Cambisol-Ranker, complex typical Cambisol and Cambisol-Ranker. Granularity of these soils is sand-loamy and loamy (Čurlík, Šály, 2002). In spite of the fact that the farmland here does not belong among the most fertile ones, farming in this territory enjoys a long tradition. Agricultural activities are carried out by the Shared Agricultural Cooperative of Liptovská Teplička and private farmers. Three-field crop rotation is still pursued. Cereals grown here are first of all oats and wheat. Recently, ecological spelt and buckwheat were also introduced.

Analysis of stakeholders' responses showed that:

- *Production functions (P) in the cadastral territory of the commune derive from fields, meadows and pastures. Agricultural potential concentrates on sheep and cow keeping and growing of fodder plants. Arable land in this territory is smaller and its productive potential is exploited first of all for production of cereals, legumes, and oil seeds. Ecological husbandry prevails. No artificial fertilizers are used so as to protect waters and soil, and it is the reason why we receive subsidies (R). The Cooperative also dedicates itself to yard sale of assortment of cheese, milk, and meat.*
- *Tangible assets are the plots around us, that is, fields and pastures used by the sharers. Likewise, mosaics of fields, meadows and pastures with dispersed vegetation represent the natural capital exploited by local husbandmen to grow potatoes.*
- *The Cooperative respects important landscape elements in the sense of legislation, the Law about nature and landscape protection (No. 543/2002). They are elements located on arable land. The Cooperative work fields located in 'disadvantaged area' are subsidized by the state. But the subsidies are each year lesser. While in 2013 we were eligible to €150/ha, this year it was only €120/ha. The difference of €30 per 1 ha is quite an amount that is missing. Apart from fields, mown pastures subsidized by the state also fulfil productive function. They are pastures located around the Cooperative and near the River Čierny Váh. Meadows and pastures with sheep farms are situated in elevated positions. Sheep farms also fulfil other than tangible functions, as they attract tourists (C).*

- *In the past, when there was no machinery available, the land was worked by hand with the aid of horses or oxen. The grass was mown by hand* (Fig. 3). *There were hay-barns in each meadow and each family had as many as five hay-barns. The mowers worked until dark, they passed the night in the hay-barn and the next day, they started at dawn; mowing of a 1–2 ha meadow took 4 to 5 weeks. Five husbandmen owned 8 to 14 ha of land. Everybody, even the poor farmers, kept at least one cow. Once, this area focused on potato growing and production of milk. There were about 300–400 cows and the surplus milk was passed down to the collection centre (P). The foundation of the Cooperative meant an end to it.*

Fig. 3. Local farmers are hand-mowing even today (M. Moyzeová, 2013).

- *Farming technology is on an advanced level nowadays. But production is low. Only a quarter of what was grown in 1989 is produced now. It is we ourselves who have destroyed the nature, and now we import products. When the meadows were mowed, we had here abundance of bilberries, raspberries, cranberries and mushrooms too (P). Apart from bilberry fruit, also their leaves were edible, and the tea made of bilberry leaves was rosy and tasty even without adding sugar. Now that the meadows are not mown, and the soil is not worked, everything is gone.*
- *Great problem now is the shortage of fodder. Wetlands cannot be mown. Everything depends on weather and the number sunny days, so that the hay would dry. It is important to harvest enough fodder for cattle during winter (P).*
- *Flocks of sheep and cattle were grazing on pastures in the past (P). Nothing is worked today, and the land is overgrown by shrubs and dwarf pine. Once it was maintained by hand mowing, but today it is not maintained and cleaned.*

In the eyes of the local farmers, the agricultural production has distinctly dropped in recent years. This critical stance was expressed by several local farmers who work the land in an old-fashioned way, following the local usage and traditions. This method is also favourable from the point of view of biodiversity because it is precisely the terraced fields with balks which maintain the high plant and animal biodiversity. The aim of green infrastructure in the sphere of agriculture is the introduction of thoughtful technologies and procedures and the support of bio-husbandry, which aim at improvement of the associated ecosystem services. The analysis of responses suggests that these aims are met in the farming sector in the cadastral territory of Liptovská Teplička. Despite this, the farmers are critical about the drop of production and loss of interest to work the land on the side of younger generation.

Forest economy (forest husbandmen, private farmers)

The objective of green infrastructure in the field of forest economy is to improve conservation of species and biotopes that depend on or are influenced by forest economy. Measures in this area must be then carried out with the aim to reduce fragmentation and degradation of forest in order to maintain their function and capacity to provide the associated ecosystem services.

According to the Environmental Regionalisation of the Slovak Republic (2010), the rate of forestation in the cadastre of Liptovská Teplička is high (45.01 to 60%, map of forestation by districts). Forest land of this territory is managed by the forest economic unit of Liptovská Teplička and the overall area of forest plots is 5,086.66 ha. The most abundant wood species are spruce and fir trees. Categories of protecting forests and those of forest with special assignment are also represented here. The area of stands of the protecting forest is 2,824.81 ha, that is, 57% of overall area of growths. Special assignment forest grows on an area of 47.45 ha (1% of overall area of growths).

Comments of an employee of forest administration and a local husbandman:

- *Forest ecosystems located around the commune fulfil a productive function, i.e., the production of wood providing a good quality spruce, fir, beech, ash and maple wood (P) on the one side, and regulative function (R), as air regulators and purifiers on the other. Protective function (Pr) is fulfilled by forest in localities with greater sloping threatened by landslide. Our activities are managed by the forest economic unit, and in case of damage, we report the level of damage.*
- *In the past, forest was first of all the place where the locals could gather forest fruit such as mushrooms, bilberries, raspberries and cranberries (P). Intangible functions are fulfilled by the locality Smrečiny that attracts tourists (C). From the point of view of development of tourism and the tours offering views of the environment, the most promising ones are those located in the direction of Kráľova hoľa Mt. (C).*
- *The State Forest comp. also owned forest and meadows. They mowed them and collected the hay (P).*

Forest growths also fulfil important function, that of mitigation of climate change and providing amounts of biological material; they also capture water and reduce the flood risk, and moreover, they reduce the natural hazards in areas prone to landslides, avalanches and flash floods. What changes have been observed by the stakeholders in the cadastre of Liptovská Teplička?

- *The greatest changes in our forests took place in the past. There used to exist beautiful forests around the commune. The branch of forest administration in Liptovský Hrádok* *(state owned forest) yearly harvested 100 000 m^3 of wood (P). The forest also became thinner because of forest aisles. Huge belts of forest were felled in order to allow rejuvenation, but what followed were strong winds and a bark beetle calamity. Clear cutting capped the process. Our forests are owned by the state, associations of owners or private persons. Everything has been devastated and 40 to 50 years old growths is all that is left.*

Responses of employees of forest administration show that they are aware of not only the productive function of forests (wood), but also its regulative and protective functions (protection of soil against landslides, regulation of microclimatic conditions, etc.). Cultural service of forest, which includes recreation opportunities and supply of forest fruit and medicinal plants, are also interpreted as useful and important. The agent that threatens the forest ecosystem is bark beetle that has annihilated hectares of forest growths, especially in the area of the Kráľova hoľa Mt., a fact very critically perceived and commented by locals. Potential risk of landslides, because of the absence of stabilizing forest growths, increases on steep slopes. This situation has to be borne in mind in terms of ecosystem measures.

Forest ecosystems in cadastre of Liptovská Teplička are important gene fund sources regarding hunting as well. The territory is part of the hunting ground of the deer game belonging to the *Poľovnícka oblasť Nízke Tatry – Sever II* with an area of 2,650 ha and it is cared for by the Hunting Association of Čierny Váh. Game in this territory is kept and the status and condition of animals is yearly assessed (processed data provided by the Hunting Association of Čierny Váh).

Amateur organisations (members of Hunting Association)

- *Population estimate of deer is 33, that of roe buck and boar is 20 and 12 respectively (P). These numbers are crucial for planning of keeping and hunting. Harvesting oscillates around 10 deer, 5 roe bucks and 9 boars. Apart from these, we also have red grouses, western capercaillies, hazel grouses, wolves, lynxes, bears and wild cats in our hunting ground. Pair of lesser spotted eagle, which occasionally visits the territory, is considered rare. Our main obligation, apart from care for the game, is to maintain the cadastral territory clean.*

Analysis of responses of interviewed members of hunting association suggests that hunters are first of all aware of production aspects and functions regarding the numbers of game that live in the territory. Care for the game and regulation of numbers also influence biodiversity in this territory.

Local authority (mayor, workers of the municipal corporation)

The origins of the commune Liptovská Teplička are associated with colonisation of Gorals who came here from the region of upper Orava. The commune was first mentioned around 1634, when it was owned by the estate of Liptovský Hrádok. People lived off farming, sheep keeping, forest works, logging and tanning. The commune is an important ethnic locality (Encyclopedia of towns and villages of Slovakia, 2005). Apart from the Roman Catholic Baroque church of 1759, there is the National Cultural Monument registered in the Central

Inventory of Monuments situated on the edge of the commune. It is a railway engine depot with its adjacent area (potential of cultural services). How do both the mayor and the local self-administration officials perceive the commune and its surroundings in terms of natural capital and ecosystem services?

- *Quality of landscape where we focus on conservation of traditional approaches also applying the modern management methods using advanced technologies is important for our village, as our final aim is to promote tourism (C). Local self-administration contributes to the enhancement of the environment yearly by about €2,500. The money is used for out planting of, for instance, hedges, for earth works, etc. Activation works financed by the state engage the Roma population. Some citizens are given jobs in the area of minor communal services, which is also partially financed by the commune. In this way, which encourages the working habit, we employ about 605 persons.*
- *The whole landscape around Liptovská Teplička fulfils productive, regulative, cultural and ecological functions. Everything relates to everything else. Tourism as a cultural service though, has been rather declining in the last years. While in the past, more than 100 local families offered accommodation to tourists, today it is only 3.*
- *Surroundings of the commune offer cultural services (C). Potato cellars and log barns, 500 of them in this territory, are much sought out landscape elements attracting tourists. Today the barns contain exhibitions of photographs of the past way of life of the locals.*
- *Educational trail running through the surroundings of Liptovská Teplička and the bike route Benkovo – Čierny Váh, through the valley of Čiernovážska dolina (C) is another tourist attraction. In future, we want to restore the narrow-gauge forest railway. Natural conditions as important natural capital offer the opportunity to develop winter sports. Ski lifts and boarding hotels providing board and lodging serve to winter recreation.*

The mayor attributes important natural potential to the slopes above the commune with ski lifts not only for the development of winter sports, but also for summer tourism. According to his opinion, cultural functions (services) stem from the history of the commune and cultural monuments both in the commune and in its environs. They attract not only the local visitors but also tourists as they offer cultural, spiritual, and aesthetic experience. Analysis of responses reveals that the mayor and the local self-administration officials encourage development of the territory respecting nature conservation and protection of the noteworthy localities and landmarks in this territory, favourable for the development of eco-tourism. As far as the green infrastructure is concerned, they stress on care for the vegetation and expansion of tourist trails as well as increased quality of life in the village.

Nature and landscape protection (workers of the state nature conservation)

The basis of green infrastructure in cadastre of Liptovská Teplička consists of NATURA2000 network elements of ecological stability and the elements of territorial nature protection pursuant the Law No. 543/2002 about nature and landscape protection in the wordings of later issued provisions (Fig. 4).

Cadastral territory of commune Liptovská Teplička is situated in the National Park Nízke Tatry and its buffer zone. The cadastre of the village includes, according to the State Inventory of Especially Protected Parts of Nature SR (as of 31.12.2015), one small protected area, that is, Nature Reserve (NR) of Martalúzka, which is protected under the fifth conservation degree. Two localities of NATURA2000, that is, the protected bird area of Nízke Tatry and the Site of Community Importance Kráľovohoľské Nízke Tatry are also in this territory. At the moment, there is proved occurrence of 174 taxa of protected plants in the NP Nízke Tatry, while 138 are vascular plants, 7 mosses, 9 lichens, and 20 mushroom species. Varied types of biotopes and relief dissection as well as inaccessibility of some parts of the territory of the Nízke Tatry Mts. are the characteristics that determine the animal species diversity. From the zoo-geographical point of view, this territory is part of the Western Carpathian Province of the Carpathian mountain ranges, where almost the complete West Carpathian, mountain and alpine species live. Many of them are endemic and relic or rare species, such as the rare insects, small but also big mammals, rare beasts and numerous birds (www.napant.sk/druhy/druhy.htm).

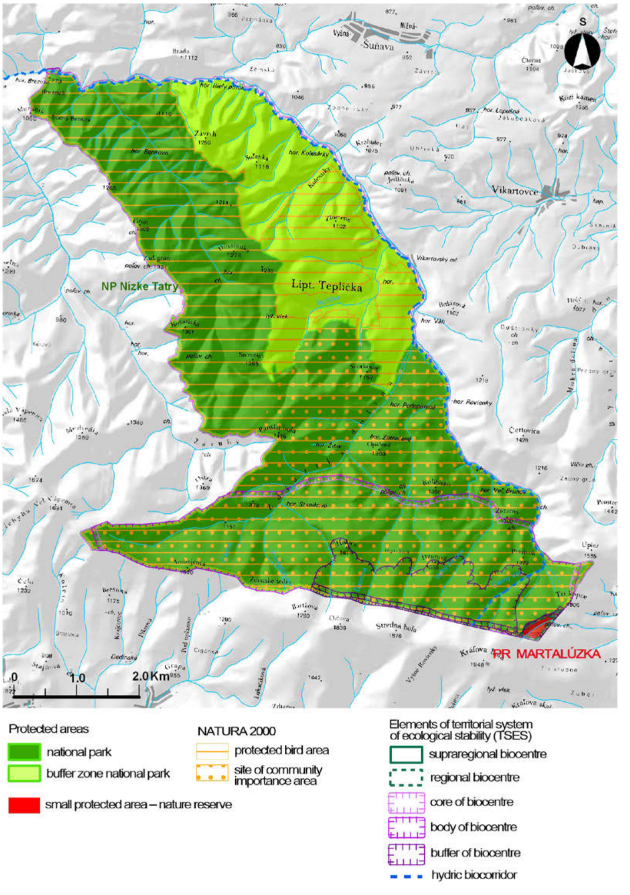

Fig. 4. Elements of nature and landscape conservation.

The landscape-ecological value of the cadastre is also enhanced by the elements of the territorial system of ecological stability (TSES) and ecologically important landscape segments, and cultural and historical landscape elements. The TSES elements include the Superregional Biocentre Nízke Tatry and the Regional Biocorridor River Čierny Váh. It is a hydric biocorridor stretching along the eastern part of the cadastre boundary, which is a natural migration route of animals in the basin. Ecologically important landscape segments, although they are not protected yet by legislation, require adequate management, conservation and enhancement as

Fig. 5. Terraced fields (M. Moyzeová, 2014).

Fig. 6. Potato cellars located on the slope beyond the commune (M. Moyzeová, 2014).

future parts of green infrastructure. Typical ecologically important landscape segments are terraces of grassland with tall and shrub vegetation groups (Fig. 5). There are also two wetlands of local significance situated next to the River Čierny Váh and the brook Ždiarsky potok. Both localities fulfil the eco-stabilizing function. They are locally important in terms of hydrology and as hatching sites of amphibians (regulating and supporting services). Cultural/historical landscape elements in the cadastre are the mosaics of small fields and permanent grassland around the built-up part of the inner space of the commune Liptovská Teplička. Potato cellars located on the slope beyond the commune (Fig. 6) and log barns are also part of this historical landscape structure.

Standpoint of state nature protection official to the assessment of natural capital and ecosystem services:

- *We do not have any conflicts between nature protection and economically active subjects because they respect limitations given by legislation. But there are different sentiments concerning husbandry in the village. Conservationists believe that the top productive function in our territory is fulfilled by forests and meadows (P). Meadows, abundant in the cadastre, should be regularly mown for the sake of maintaining productivity and biodiversity. As far as the regulative function is concerned, meadows and the River Čierny Váh are important. The productive function of the stream is only partial because fishing here is restricted. The whole cadastre of the village is important for circulation of nutrients and for intangible functions ensuing from the number of natural, cultural, and historical points of interest (C).*

Analysis of responses of the state nature conservation official reveal that the greatest significance is attributed to forest ecosystems in the cadastre of Liptovská Teplička that are protected in the category of National Park and Nature Reserve. They are reservoirs of biodiversity in this territory. The fact that the conflicts between conservation and land use are not serious and the economic subjects cooperate is positive from the point of view of the green infrastructure initiative.

Water economy (water economist, private farmers)

Green infrastructure helps to maintain good water quality thus fulfilling the guidelines of the Drinking Water Directive (EU) 98/83/ES.

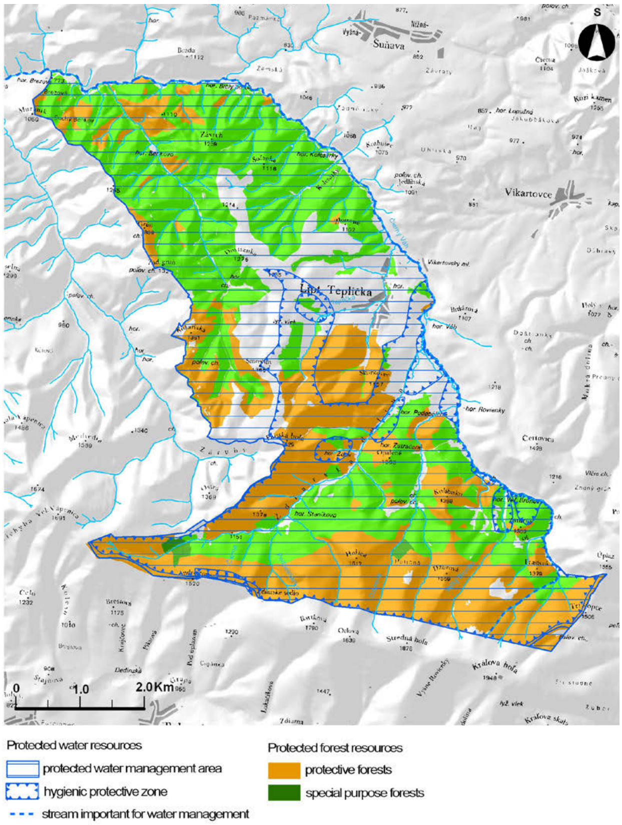

Fig. 7. Elements of natural resources protection.

The territory of Liptovská Teplička is remarkable for having important natural resources (Fig. 7) including water sources, which are mostly used for supplies. This territory is classified into two main hydrological regions (Malík, Švasta, 2002): Palaeozoic and Mesozoic. It is located in the protected water economy area, that is, Protected Water Area of Nízke Tatry-east. The overall area of the PWA is 805 km^2, with water source available for use at a volume of 4.76 m^3.s^{-1} (www.sazp.sk/slovak/periodika/sprava/psrsk/voda). The edge of the Protected Water Economic Area of upper basin of the River Hnilec extends to the south-eastern tip of the cadastre of Liptovská Teplička. The territory boasts several springs and wells that supply drinking water to the population. Water of these underground sources is of top quality and it satisfies the water quality indicators (Provision No. 496/2010 Ruling of the Government of the SR amending the Ruling of the Government SR No. 354/2006 setting the exigencies concerning water designated for human consumption and quality control of water designated for human consumption); 98% of the commune's population consume water of the public main (Report on the status of the environment in the Slovak Republic, 2014). The local water main is administered by the Podtatranská vodárenská spoločnosť, a.s water supply company, which supplies water from the above-mentioned water sources in Liptovská Teplička, and from other streams to several towns (Spišská Nová Ves, Kežmarok and Poprad) as well. According to Promulgation of the Ministry of Environment SR No. 211/2005, the river Čierny Váh is an important water economic stream.

Stakeholders evaluated water sources in the following manner:

- *Liptovská Teplička had its own water of very good quality, which was also supplied to Poprad, because there was not sic spring in the valley of Spiš (P). There are many bores and springs used for drinking. Among our water sources are springs Veľký Brunov and Malý Brunov, Macov and the bore of Rovienky. Water quality is regularly checked.*
- *We have a well beyond the commune (P). The stream of Teplica once passed through the village. In the 1970s, a water main was constructed. Today the village is supplied from the water main though people living above the village, in the area of ski lifts (Štefanov, Hálky), still have their wells drilled.*

Discussion and conclusion

Many international and Slovak authors are involved in the assessment of potentials. According to the studies of Neef (1966), Graf (1976), Haase (1978), Bierhals (1980), Buchwald (1996), Mazúr, Urbánek (1982), Oťaheľ, Poláčik (1987), Mannsfeld, (1979), Oťahel et al. (1991), Otaheľ (1994), Drdoš (1990), Izakovičová et al. (1997), Mazúr et al. (1985), and Hrnčiarová (1996), potential is defined as a complex of landscape properties interpreted as an offer, capacity or applicability to provide for various functions with the objective to satisfy society's needs in accordance with harmonious functioning of linkages in the landscape system. Forman and Godron (1986) interpret territorial potential as a possible use of certain area and its values. One of the principal types of territorial potential is the extractable production applying available technologies and expressible by a price. Another principal type of territorial potential difficult to express in economic terms is the basic role of landscape (its components and elements) for regulation, which is pursued with the intention to maintain balance in landscape. The third principal type of territorial potential, that is, the aesthetic, therapeutic and inspirational value of landscape for humans cannot be expressed by or reduced to simple economic terms (Forman, Godron, 1986).

Recently, the term of *natural potential* has been often replaced by the term *natural capital* in the context of ecosystem services.

The aim of this paper was to assess the natural capital and ecosystem serviced in cadastral territory of Liptovská Teplička in the view of a selected sample of local stakeholder. It must be also noted that the terms natural capital, ecosystem services and green infrastructure were rather too sophisticated for the addressed respondents. After a brief explanation of these terms, the stakeholders were able to work with them. They assessed the obtained information about ecosystems used in the form of producing, regulating and cultural services based on the thorough knowledge of the territory where they lived and worked. The attitude of the addressed stakeholders active in the field of agriculture, forest economy, water economy, private farming and also the representatives of the local self-government to natural resources, nature conservation and preservation of biotopes and biodiversity in their territory is positive. Regarding the provided ecosystem services, the territory of Liptovská Teplička was assessed as a farming landscape with high natural assets and high hygienic requests. As the territory in question is part of the National Park Nízke Tatry and its buffer zone, as well as of the Protected Water Economic Area of Nízke Tatry, several economic activities are limited or restricted by legal protection (Law about nature and landscape protection, Water Law).

The research showed that productive services of the ecosystem were assessed more frequently as they directly influence humans in terms of food, water, wood and fodder.

As all the natural resources represent important natural capital in the territory, it is necessary to preserve and further develop them and to apply the geo-ecosystem approach for the conservation of biodiversity, which is based on a full-area system of protection and optimal land use. Green infrastructure in Liptovská Teplička is based on conservation, maintenance, and protection of natural, semi-natural and artificial ecosystems that supply varied ecosystem services and benefits. In future, it is necessary to encourage the classical farming methods and processing of own harvest, to develop ecological management and production of bio products. These ecological methods applied in the territory increase the species diversity and contribute to increased ecological diversity of the territory. It is also necessary to continue with yard sale. Sheep keeping should be accomplished using the deep litter method in order to protect the soil and water quality against pollution. Green infrastructure in the cadastre of Liptovská Teplička already exists in form of protected territories, the TSES and NATURA2000 elements. Planning and decision-making procedures should respect the occurrence and value of natural capital and to avoid loss of ecosystem services. Stable economic, social and political situation in the commune Liptovská Teplička and drawing on the structural funds will support the traditional farming forms, development of agro-tourism and conservation of biodiversity and landscape diversity.

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References

Aarhus Convention (1998). *Convention on access to information*. Public Participation in Decision-Making and Access to Justice in Environmental Matters.

Bierhals, E. (1980). Ökologische raumgliederung für die landschaftsplanung. In K. Buchwald & W. Engelhardt (Eds.), *Handbuch für planung, gestaltung und schutz der umwelt* (pp. 80–104). München: BLV.

Buchwald, K. (1996). Landschaften als gegenstand nutzungs - und umweltbezogener planungen. In K. Buchwald & E. Engelhardt (Eds.), *Umweltschutz – grundlagen und praxis. II. Bewertung und planung im umweltschutz* (pp. 1–37). Bonn: Economica Verlag.

CICES (2013). Common International Classification of Ecosystem Services (WWW Document). Biodiversity Information System for Europe. URL http://biodiversity.europa.eu/maes/common-international-classification-of-ecosystem-services-cices-classification-version-4.3 (accessed 11.18.15).

COM (2010). 2020 final, OJ C 88 of 19.3.2011.

COM (2011a). 571 final, OJ C 37 of 10.2.2012.

COM (2011b). 244 final, OJ C 264 of 8.9.2011.

COM (2011c). 625 final/2.
COM (2013). 249 final.
Čurlík, J. & Šály R. (2002). Zrnitosť pôdy. In *Atlas krajiny Slovenskej republiky* (p. 110–111). Bratislava: MŽP SR, Banská Bystrica: SAŽP.
Drdoš, J. (1990). Contribution to the landscape carrying capacity issue (on example of the Tatra National Park) (in Slovak). *Geografický Časopis*, 42(1), 3–22.
Encyclopedia of towns and villages of Slovakia (in Slovak) (2005). Lučenec: Vydavateľstvo PS-LINE.
Environmental regionalisation of the SR (in Slovak) (2010). Bratislava: MŽP SR, Banská Bystrica: SAŽP.
Forman, R.T.T. & Godron M. (1986). *Landscape ecology*. New York: Wiley.
Graf, D. (1976). *Ökonomische bewertung von naturpotentialen und naturressourcen*. Geographische und Ökologische Grundlagen der Landschaftsplanung, Mitteilungsblatt.
Haase, G. (1978). Zur ableitung and kennzeichnung von naturpotentialen. *Petermanns Geographische Mitteilungen*, 122, 113–125.
Hrnčiarová, T. (1996). Evaluation of the ecological carrying capacity of the landscape. *Ekológia (Bratislava)*, 15(4), 441–447.
Chrenščová, V. (2011). Environmental quality in the perspective of local population in the territory of protected area Horná Orava (in Slovak). *Geografický Časopis*, 63(1), 69–85.
Izakovičová, Z., Miklós, L. & Drdoš J. (1997). *Landscape-ecological conditions of sustainable development (in Slovak)*. Bratislava: Veda, vydavateľstvo SAV.
Landscape atlas of the Slovak Republic (in Slovak) (2002). Bratislava: MŽP SR, Banská Bystrica: SAŽP SR.
Law No. 543/2002 on nature and landscape protection in wording of later issued provisions (in Slovak).
Malík, P. & Švasta J. (2002). Hlavné hydrogeologické regióny. In *Atlas krajiny Slovenskej republiky* (p. 104). Bratislava: MŽP SR, Banská Bystrica: SAŽP.
Mannsfeld, K. (1979). *Die beurteilung von naturraumpotentialen als aufgabe der angewandten physischen geographie*. Materialien zur Raumordnung.
Mazúr, E. & Urbánek J. (1982). Category of space in geography (in Slovak). *Geografický Časopis*, 34(3), 309–325.
Mazúr, E., Drdoš, J., Bučko, Š., Huba, M., Oťahel, J., Očovský, Š. & Tarábek K. (1985). *Landscape synthesis of the area of Tatranská Lomnica (in Slovak)*. Bratislava: Veda, vydavateľstvo SAV.
Neef, E. (1966). Zur frage des gebietswirtschaftlichen potentials. *Forschungen und Fortschritte*, 40, 65–96.
OECD (2011). Towards green growth. http://www.oecd.org/greengrowth/48224539.pdf
Oťahel, J. & Poláčik Š. (1987). *Landscape synthesis of the Liptovská kotlina Basin (in Slovak)*. Bratislava: Veda, vydavateľstvo SAV.
Oťahel, J., Lehotský, M. & Ira V. (1991). Environmental planning: Principles and procedures. In *Landscape synthesis research*. Conference papers. Guelph: University of Guelph.
Oťahel, J. (1994). Problems of environmental planning (regional solution) (in Slovak). In *Trvalo udržateľný rozvoj a krajinnoekologické plánovanie v európskych horských regiónoch* (pp. 167–174). Zvolen: Technická univerzita.
Potschin, M. & Haines-Young R. (2013). Landscapes, sustainability and the place-based analysis of ecosystem services. *Landsc. Ecol.*, 28, 1053–1065. DOI: 10.1007/s10980-012-9756-x.
Prokša, M., Held, Ľ., Haláková, Z., Tóthová, A., Orolínová, M., Urbanová, A. & Žoldošová K. (2008). *Methodology of pedagogical research and its application to didactics in sciences (in Slovak)*. Bratislava: PríF UK.
Promulgation No. 211/2005 of the Ministry of Environment SR which ordains the inventory of streams important for water economy and water works (in Slovak).
Provision No. 496/2010 Ruling of the Government of the SR amending the Ruling of the Government SR No. 354/2006 setting the exigencies concerning water designated for human consumption and quality control of water designated for human consumption (in Slovak).
Report on the status of the environment in the Slovak Republic (in Slovak) (2014). Bratislava: MŽP SR.
Republic Frances (2010). National sustainable development strategy 2010–2012 – Towards a green and fair economy. http://www.developpementdurable.gouv.fr/spip.php?page=article&id_article=21743
Vadovičová, E. & Džatko M. (1992). *Assessment of productive soil potential and soil-ecological units of model enterprises (in Slovak)*. Research study. Bratislava: VÚPÚ.
WB (2012). World Bank. Inclusive green growth: The pathway to sustainable development. Washington: World Bank. https://openknowledge.worldbank.org/handle/10986/6058
www.napant.sk/druhy/druhy.htm
www.sazp.sk/slovak/periodika/sprava/psrsk/voda

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RETHINKING *Eucalyptus globulus* Labill. BASED LAND USE SYSTEMS IN SMALLHOLDER FARMERS LIVELIHOODS: A CASE OF KOLOBO WATERSHED, WEST SHEWA, ETHIOPIA

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Abstract

Feyisa D., Kissi E., Kebebew Z.: Rethinking *Eucalyptus globules* Labill. based land use systems in smallholder farmers livelihoods: a case of Kolobo Watershed, West Shewa, Ethiopia. Ekológia (Bratislava), Vol. 37, No. 1, p. 57–68, 2018.

Despite their restriction, smallholder farmers have been continuing growing *Eucalyptus globulus* in the cultivated land in the central highland of Ethiopia. Literature has shown controversial issues against *E. globulus*. Therefore, the objective of the study was to investigate the compatibility of *E. globulus* in the smallholder farmers' land use system. Soil samples were collected from five different land uses and analysed for selected physical and chemical properties. The socioeconomic contribution of *E. globulus* was collected through household surveys from 110 households. Analysis of soil showed that organic carbon (OC), total nitrogen (TN) and cation exchange capacity (CEC) were significantly higher ($P<0.05$) under *E. globulus* compared to the cultivated land. The survey results also showed that the largest proportion (58%) of households was interested in growing *E. globulus* because of its multiple uses. About 83% of households responded that *E. globulus* help them to attain food security through increasing the purchasing power of smallholder farmers to buy agricultural inputs and food. This study has substantiated the role of *E. globulus* in the land use system of smallholder farmers. Most of the soil fertility indicators were better under *E. globulus*. The present finding reveals that *E. globulus* degrade the soil seemingly difficult to generalise. Growing *E. globulus* must be promoted under appealing land use to enhance smallholder farmers' livelihoods. Removing *E. globulus* from the land use system may jeopardise the food security situation of many households.

Key words: land use systems, farm forestry, soil fertility, rural livelihoods, socioeconomics.

Introduction

Tree-based land use is mainly the pathway of farming practices change that smallholder farmers undertake to improve their livelihoods (Newby et al., 2014). Many studies indicate that tree is a vital component of land use system that sustains rural livelihoods (Pretzsch, 2005). To this effect, smallholder farmers widely plant *Eucalyptus globulus* in various spatial patterns as component of land use in central highlands of Ethiopia (Mekonnen, 2000). An experience from Arsi Negelle in Ethiopia shows that about 11% of cropland has been converted to growing *Eucalyptus* woodlots (Jenbere et al., 2012).

Land is the fundamental component of production in Ethiopia (Teshome et al., 2014) that supports millions of the rural population particularly through agriculture (Mekuria, Aynekulu, 2011). The fear of displacing agricultural crops and the detrimental environmental effect has raised debate over growing *E. globulus* (Yitaferu et al., 2013). *Eucalyptus* has been alleged that it affects the physical and chemical properties of soil that leads the Oromia regional state to enact rural land use and administration (proclamation number 56/2002) that discourage growing *Eucalyptus* on cropland as agriculture is the means of livelihood (Kebebew, Ayele, 2010).

Smallholder farmers' agriculture is believed to tackle food security in the country (Negra et al., 2014). Nevertheless, land degradation coupled with climate change constraints agricultural production at subsistence (Karltun et al., 2013). This bring about millions of the population have failed to attain food security and suffered from severe shortage of food (Karltun et al., 2013). Smallholder farmers undertake farming practices change as an option to keep their livelihood sustenance (Kristjanson et al., 2012; Negra et al., 2014). In Ethiopia, rehabilitation of degraded land has taken attention to increase agricultural production, better off the household well-being. The farmers' experience depicts growing *Eucalyptus* as an innovative land use approach to normalise the land benefits (Negra et al., 2014).

Despite the controversy over *Eucalyptus*, there are no concrete empirical evidence that validates the detrimental effect of *Eucalyptus* on food security and livelihoods (Kidanu et al., 2005; Duguma et al., 2010; Yitaferu et al., 2013; Haile et al., 2014). In Ethiopia, food security is supposed to be addressed through increased agricultural production (Diao, Pratt, 2007; Van der Veen, Tagel, 2011). In contrast, this type of farming requires purchasing power of inputs. However, because of the nature of farming system (Abebaw et al., 2010) smallholder farmers have the limitation of purchasing power. The apparent experience is to undertake land use activities that complement agricultural production (Teshome et al., 2014). Growing *Eucalyptus* is one form of the land use that helps smallholder to sustain the livelihoods (Jenbere et al.; 2012, Duguma, Hager, 2011). For the rural people, *Eucalyptus* is a means of safety net (Kebebew, Ayele, 2010). Even if the policy makers discourage growing *E. globulus* on farm, smallholder farmers in the central highlands of Ethiopia have continued growing *E. globulus* as one component of land use (Kebebew, Ayele, 2010). The objective of the study was to investigate compatibility of *E. globulus* under smallholder farmers. This paper tries to answer the following research questions:

- Does the rate of change of soil go beyond the optimum soil fertility?
- How does *E. globulus* fit into smallholder farmers land use option in context of food security?

Material and methods

The study was conducted in Kolobo watershed, Adea Berga district of the west Shewa zone, Oromia, Ethiopia. The area where the study is conducted is about 250 ha. The district is located at a distance of 74 km from Addis Ababa on the way to Mogor Cement Enterprise. Geographically, it is located between 9°12′ to 9°37′N and 38°17′ to 38°36′E. The altitude of the area ranges from 1,500 to 3,180 m a.s.l. with an average annual temperature ranges from 15 to 25.5 °C. The annual rainfall of the study area goes up to 1,400 mm. The main rainy season is from June to September and August is the peak rainy month (Kidanu et al., 2005).

Land use complementary effect of *E. globulus* was investigated through assessing soil properties and socioeconomic benefits. Data was collected from September 2013 to April 2014. A reconnaissance survey was carried out and five land use types were chosen. Under farmers' experience, *Eucalyptus* is mostly harvested starting from year four (Kidanu et al., 2005). Hence, age of *E. globulus* was fixed at year four. To minimise soil properties variation because of the slope of the land, first, the area was divided into similar slope category (bottom 2–5%, middle slope 6–9%, top slope >9%) and five different homogeneous land uses were selected from middle slope category. The elevation is about 2,700 m a.s.l. Vertisols dominates the area. Land use history shows the land is used for growing barely, wheat, teff, beans and peas at different time. Next plot of size 20×20 m with three replicates from each land use types was identified. Within a plot, five different points (2×2 m) were selected, dug to the depth of 30 cm and finally collected soil sample per plot were composited to get one sample. A total of 15 soil samples were backed in plastic bags and transported to laboratory for analysis. Similar study approach was used by Duguma et al. (2010), Abbasi et al. (2007) and Jenbere et al. (2012).

Collected soil samples were air dried, mixed and passed through a 2-mm sieve for the analysis of selected soil physical and chemical properties at the Jimma University College of Agriculture soil laboratory. Soil texture, bulk density and moisture content were analysed for soil physical properties, whereas soil pH, electrical conductivity (EC), organic carbon (OC), organic matter (OM), total nitrogen (TN), availability of phosphorous (Av.P), cation exchange capacity (CEC) and exchangeable bases were analysed for soil chemical properties. Laboratory analysis was conducted following standard procedures of soil physical and chemical analysis.

Soil texture was analysed following the Boycouos hydrometric method (Abbasi et al., 2007). Hydrogen peroxide (H_2O_2) was first added to destroy OM and then sodium hexametaphosphate $(NaPO_3)_6$ was added to disperse the soil. Soil textural classes were determined using the USDA triangular guideline (Brady, Weil, 2002). Soil bulk density was determined by core method after drying the soil samples in an oven at 105 °C following Abbasi et al. (2007) and Sahlemedhin and Taye (2000). Soil moisture content was measured using gravimeter method as described by Sahlemedhin and Taye (2000).

Soil pH-H_2O and EC were estimated from soil-to-water ratio of 1:2.5 and 1:5, respectively (Sahlemedhin, Taye, 2000). The reading of pH was taken by pH meter, whereas EC was measured by electrical conductivity meter. The soil OC was measured by the Walkley–Black oxidation method with potassium dichromate ($K_2Cr_2O_7$) in a sulphuric acid and converted to soil OM by multiplying it by the factor of 1.724 as described by Nelson and Sommers (1982). TN was determined by the Kjeldahl methods and available phosphorus was determined using the Bray II extraction method (Van Reeuwijk, 1992).

Total exchangeable bases were determined after leaching the soils with ammonium acetate. Amounts of Ca^{2+} and Mg^{2+} in the leachate were analysed by atomic absorption spectrophotometer (AAS) and K^+ and Na^+ was analysed using flame photo metrically (Chapman, 1965). CEC was determined at soil pH level of 7 after the displacement by using 1N ammonium acetate method and then estimated titrimetrically by distillation of ammonium that was displaced by sodium (Chapman, 1965). Percent base saturation (PBS) was calculated by dividing the sum of the base-forming cations (Ca, Mg, Na and K) by the CEC of the soil and multiplying it by 100 (Fageria, 2009).

Structured and semi-structured questionnaires were prepared and pretested to collect the socioeconomic information. The household survey focused on *E. globulus* and land use under farmers' circumstances. A list of all households living (N = 561) in the watershed was obtained with the help of local administrative bodies (development agents). A sample size was determined using the Cochran (1977) formula. Based on the formula, a total of 110 households were randomly selected for the interview. The information was collected through face-to-face interview with all households (n = 110) at their convenient time. Each household was categorised as rich, medium and poor in the *kebele*. The local administrative body has the record of the wealth status of every household for any development activities. Hence, wealth status of the household was recorded based on the local administrative record. Cost of the fertiliser and income from the *Eucalyptus* took into account the actual farmer experience. Annual production and productivity of agricultural crops were estimated in quintal {metric} (1quintal {metric}= 100 kg). Farm gate price, cost of fertilisers and income from *Eucalyptus* were estimated in local currency (1 ETB= $ 0.049 USD). It must be known that income from *Eucalyptus* was collected only on actual income from *Eucalyptus* sales of farmers' experience. The information obtained through household survey were triangulated through focus group discussion (with development agent and expert from district agricultural bureau) and key informant interview (with local elders). Collected information was organised, coded and finally analysed.

Soil physico-chemical properties were analysed using one way analysis of variance (ANOVA). Before the analysis, the data was checked for the assumption of ANOVA. ANOVA was run using SAS software (SAS, 9.2). For variables showing statistically significant difference between treatments ($p < 0.05$), analysis of mean separation was carried out using least significant difference (LSD) at 5% probability. The socioeconomic data was analysed using descriptive and inferential statistics using SPSS version 20.

Results and discussion

Soil physical properties

Soil texture, bulk density and moisture were analysed for soil physical properties. The result showed significant difference ($P < 0.005$) amongst land use types (Tables 1 and 2). Soil under forest, *E. globulus*, cultivated and grazing land had clay texture, whereas degraded land had clay loam texture. The proportion of clay was highest (56%) under forest and lowest (27%) under degraded land. Soil under *E. globulus* had significantly higher proportion of clay particles (48%) than grazing (41%) and degraded land (27%). Although the proportion of clay under *E. globulus* and culti-

T a b l e 1. Mean ± SEM values for soil textural properties under five land use types.

Land use	Clay (%)	Sand (%)	Silt (%)	Textural class
Forest	55.67 ± 1.76^{a}	21.00 ± 3.33^{cd}	21.33 ± 4.67^{b}	Clay
E. globulus	47.67 ± 1.76^{b}	29.00 ± 3.06^{bc}	23.33 ± 4.06^{b}	Clay
Cultivated	41.67 ± 2.40^{bc}	35.33 ± 4.06^{ab}	24.00 ± 4.00^{b}	Clay
Grazing	41.00 ± 1.15^{c}	15.33 ± 2.40^{d}	43.67 ± 3.53^{a}	Clay
Degraded	27 ± 2.31^{d}	42.67 ± 2.40^{a}	30.33 ± 4.67^{ab}	Clay loam
PV	0.0001	0.0027	0.0417	-
LSD (0.05)	6.58	10.83	15.17	-
CV (%)	8.20	20.15	22.18	-

Notes: PV – p value; CV – coefficient of variance; LSD – least significant difference; SEM – standard error mean.

T a b l e 2. Mean ± SEM values for SMC and BD under different land use types.

Land use	SMC (%)	BD (g cm^{-3})
Forest	34.37 ± 1.78^{a}	0.97 ± 0.02^{b}
E. globulus	17.64 ± 0.55^{c}	0.87 ± 0.05^{c}
Cultivated	18.97 ± 0.87^{c}	1.14 ± 0.01^{a}
Grazing	26.67 ± 0.85^{b}	1.01 ± 0.01^{b}
Degraded	12.89 ± 0.57^{d}	1.10 ± 0.02^{a}
PV	<0.0001	0.0002
LSD (0.05)	3.56	0.07
CV (%)	8.56	3.75

Notes: SMC – soil moistures content; BD – bulk density; PV – p value; CV – coefficient of variance; LSD – least significant difference; SEM – standard error mean.

vated land was not significantly different, in terms of magnitude, there was a higher proportion of clay under *E. globulus* (48%) than cultivated land. The clay fraction under *E. globulus* was higher when compared to cultivated land. This might be due to high vegetation cover that reduces the clay fractions likely to be lost by selective erosion processes. Clay particles are removed and transported easily than sand particle (Selassie et al., 2015). Those agree with the findings of Lemenih et al. (2005) who had reported high clay content under *E. globulus* plantation as compared to cultivated and grazing land. This means that soils under *E. globulus* are more fertile than those of cultivated and degraded land.

Soil under *E. globulus* had the lowest bulk density (0.87 g cm^{-3}) as compared to other land use types. Cultivated land had the highest bulk density (1.14 g cm^{-3}). The lower bulk density under *E. globulus* implies that the soil is less compacted. The optimum range of bulk density of agricultural soil is between 0.9 and 1.2 g cm^{-3} (Frank, 1990). The result shows that soil under *E. globulus* is within the range of the optimum condition for agriculture (0.87–-1.14 g cm^{-3}) (Table 2). According to Miller and Donahue (1997), bulk densities need to be below 1.4 g cm^{-3} for good plant growth. Yitaferu et al. (2013) had reported lower bulk density (1.07 g cm^{-3}) under *Eucalyptus* compared to cultivated land (1.11 g cm^{-3}) in west Gojam Amhara regional state, Ethiopia. Similarly, Selassie and Ayanna (2013), Getachew et al. (2012) and Haile et al. (2014) had reported lower soil bulk density under *Eucalyptus* compared to cultivated land. The present results agree with the previous findings on the same.

Soil moisture content results showed significant difference under different land use types ($P < 0.05$). Soil under forest had the highest (34%) and degraded land had the lowest soil moisture (13%). Soil moisture under *E. globulus* (18%) was lower compared to that under the forest (34%), under grazing (27%), of cultivated land (18%) but greater than that under degraded land (12%). Although the absolute value of soil moisture under *E. globulus* was lower than the cultivated land, the results were not significantly different at $P > 0.05$ (Table 2). This is probably attributed to short rotation age of *E. globulus* (4 years). These findings are in agreement with similar study report by Selassie and Ayanna (2013), Getachew et al. (2012) and Haile et al. (2014) who had reported no significant difference on the same between *E. globulus* and cultivated land.

Soil chemical properties

Soil under *E. globulus* had significantly ($P < 0.05$) lower pH value (5.55) compared to other land use types. Owing to high cation uptake and low accumulation of exchangeable bases, soil under *E. globulus* had lower EC (0.025ds/m) compared to others (Table 3). Table 4 shows positive and significant correlation between EC and soil pH ($r = 0.85$), Mg^{2+} ($r = 0.63$) and Ca^{2+} ($r = 0.63$). The finding agrees with Haile et al. (2014) who had reported lower pH value under *Eucalyptus* woodlots compared to other land use types. However, the pH value (5.55) of this study is greater than that previous report by Selassie and Ayanna (2013) who reported pH values of 5.06 and 5.01 under the *E. globulus* at Abechikeli Mariam and Aferfida Georgis sites, respectively. Duguma et al. (2010) also reported a lower pH value (5.06) under *E. globulus* compared to homestead, croplands and grazing lands at central highlands of Ethiopia. The higher pH (5.55) value of this finding compared to the previous findings is probably attributed to the age of the *Eucalyptus* as the sample soil was taken under 4 years older *Eucalyptus*. According to Frank (1990), the pH values of most

T a b l e 3. Mean ± SEM values for soil pH and EC under different land use systems.

Land use	**pH**	**EC (ds/m)**
Forest	6.52 ± 0.11^{a}	0.045 ± 0.0028^{a}
E. globulus	5.55 ± 0.14^{b}	0.025 ± 0.0028^{c}
Cultivated	6.29 ± 0.25^{a}	0.032 ± 0.004^{bc}
Grazing	6.28 ± 0.09^{a}	0.037 ± 0.0015^{ab}
Degraded	5.60 ± 0.16^{b}	0.029 ± 0.003^{bc}
PV	0.007	0.003
LSD (0.05)	0.52	0.01
CV (%)	4.58	13.16

Notes: pH – power of hydrogen; EC – electrical conductivity; PV – p value; CV – coefficient of variance; LSD – least significant difference.

T a b l e 4. Pearson correlation matrix for the selected soil properties.

	pH	**EC**	**OC**	**TN**	**AvP**	**Ca**	**Mg**	**Na**	**K**	**CEC**
pH	1									
Ec	0.86**	1								
OC	0.53*	0.54*	1							
TN	0.33	0.41	0.94**	1						
AvP	0.74**	0.66**	0.66**	0.63*	1					
Ca	0.64*	0.63*	0.84**	0.75**	0.75**	1				
Mg	0.63*	0.72**	0.80**	0.72**	0.77**	-0.74**	1			
Na	0.36	0.49	0.60*	0.61*	0.18	-0.38	-0.43	1		
K	0.75**	0.67**	0.66**	0.62*	0.99**	-0.75**	0.77**	0.19	1	
CEC	0.46	0.47	0.93**	0.94**	0.72**	0.91**	-0.75**	0.50	-0.72**	1

Notes: pH – power of hydrogen; EC – electrical conductivity; OC – soil organic carbon; TN – total nitrogen; AvP – available phosphorous; Ca – calcium; Mg – magnesium; Na – sodium; K – potassium; CEC – cation exchange capacity; ** Correlation is significant at the 0.01 level (two-tailed); *Correlation is significant at the 0.05 level (two-tailed).

T a b l e 5. Mean ± SEM values for OC, OM, TN and Av.P under different land use systems.

Land use	**OC (%)**	**OM (%)**	**TN (%)**	**Av.P (ppm)**
Forest	3.05 ±0.22^{a}	5.27 ± 0.37^{a}	0.31 ± 0.02^{a}	5.71 ± 0.48^{a}
E. globulus	2.02 ± 0.28^{b}	3.47 ± 0.49^{b}	0.25 ± 0.04^{b}	2.08 ± 0.41^{c}
Cultivated	1.24 ± 0.28^{c}	2.13 ± 0.49^{c}	0.12 ± 0.02^{c}	3.99 ± 0.37^{b}
Grazing	2.24 ± 0.15^{b}	3.86 ± 0.26^{b}	0.19 ± 0.01^{b}	2.93 ± 0.68^{bc}
Degraded	0.48 ± 0.11^{d}	0.82 ± 0.19^{d}	0.04 ± 0.01^{d}	0.61 ± 0.37^{d}
p-v	0.0001	0.0001	0.0001	0.003
LSD (0.05)	0.62	1.07	0.06	1.39
CV (%)	18.26	18.20	15.77	24.22

Notes: OC – soil organic carbon; OM – organic matter; TN – total nitrogen; Av.P – available phosphorous; PV – p value; CV – coefficient of variance; LSD – least significant difference.

agricultural soils are in the range of 5.5–7. The pH value (5.5) under *E. globulus* is under medium rating categories implying that the land is suitable for growing crops from soil pH point of views. The experiences of cropping on the land that has been under *Eucalyptus* plantation in west Gojam Amhara regional state, Ethiopia, verify the same (Yitaferu et al., 2013). This result argues against the detrimental effect of *Eucalyptus* on soil pH, rather soil under *Eucalyptus* plantation has a good potential for cropping.

Soil OC, OM, TN and available phosphorous are presented in Table 5. Significant difference ($P < 0.05$) was observed amongst the land use types. Soil under forest had the highest soil OC, OM, TN and available phosphorous, whereas soil under degraded land had the lowest soil OC, OM, TN and available phosphorous. Soil under *E. globulus* had significantly higher soil OC (2.02%), OM (3.47%) and TN (0.25%) compared to cultivated land. The soil OC and OM under forest and *E. globulus* were rated as medium categories; whereas low under cultivated land. The findings agree with similar studies (Abbasi et al., 2007; Duguma et al., 2010; Getachew et al., 2012; Haile et al., 2014). Abbasi et al. (2007) had reported differences in OM amongst land use types. Getachew et al. (2012) had reported more OC than farmland. Haile et al. (2014) had reported more OM under grassland and woodlot compared to cereal farms. Duguma et al. (2010) had more soil OC and TN under small-scale woodlot than pasturelands and cereal farms. Soil under *E. globulus* had significantly ($P < 0.05$) lower available phosphorous than cultivated land (Table 5). There is positive and significant ($r = 0.74$) relationship between available phosphorous and pH (Table 4). Soil pH influences the availability of phosphorus. Phosphorus fixation takes place at lower pH (Kebede, Raju, 2011). Tisdale et al. (2002) also noted that maximum availability of phosphorus generally occurs in a pH range of 6.0–7. This finding agrees with Getachew et al. (2012), Haile et al. (2014) and Yitaferu et al. (2013) findings on the available phosphorous under *Eucalyptus*. The TN is directly and significantly associated with OC ($r = 0.94$) and CEC ($r = 0.94$) (Table 4). The result is similar to Mengist (2011) who had reported higher soil nitrogen under *E. globulus* compared to cultivated land.

CEC, percentage of base saturation (PBS) and exchangeable Ca, Mg and K were significantly ($P < 0.05$) affected by the land use systems, whereas exchangeable Na was not statistically affected by the land use systems (Table 6). Soil under *E. globulus* had higher CEC value (30.78 Cmol(+)/

T a b l e 6. Mean ± SEM values for CEC, exchangeable bases and PBS under different land use types.

LU	CEC	Ca	Mg	Na	K	Total cation	PBS (%)
	Cmol(+)/kg						
F	37.40 ± 1.65^{a}	18.50 ± 1.21^{a}	8.00 ± 0.58^{a}	0.33 ± 0.01	1.14 ± 0.1^{a}	27.97	74.7 ± 0.79^{a}
E	30.78 ± 0.87^{b}	11.86 ± 1.95^{b}	2.30 ± 1.01^{c}	0.31 ± 0.02	0.42 ± 0.08^{cd}	14.89	48.4 ± 3.21^{c}
C	25.09 ± 0.70^{c}	11.21 ± 0.41^{b}	2.97 ± 0.25^{c}	0.29 ± 0.00	0.80 ± 0.08^{b}	15.27	61 ± 2.74^{b}
G	28.21 ± 1.09^{b}	13.50 ± 0.85^{b}	5.83 ± 0.44^{b}	0.31 ± 0.00	0.60 ± 0.14^{bc}	20.24	71.5 ± 2.00^{a}
D	16.96 ± 1.46^{d}	6.17 ± 0.60^{c}	1.30 ± 0.21^{c}	0.29 ± 0.02	0.14 ± 0.05^{d}	7.90	46.4 ± 2.32^{c}
PV	<0.0001	0.0012	0.0003	0.3512	0.0003		<0.0001
LSD	3.05	3.92	2.04		0.28		7.62
CV (%)	5.84	16.99	20.52	8.94	23.80		6.70

Notes: LU – land use; F – forest; E – *E. globulus*; C – cultivated; G – grazing; CEC – cation exchange capacity; BPS – percentage base saturation; PV– p value; CV – coefficient of variance.

kg) compared to grazing (28.21 Cmol (+)/kg), cultivated land (25.09 Cmol (+)/kg) and degraded land (16.96 Cmol (+)/kg). The CEC ratings under forest and *E. globulus* were high. This variation might be attributed to the high accumulation of OM and high clay percentage under *E. globules* and forest land use types. There is positive and strong significant association between CEC and Clay ($r = 0.97$) and CEC and OM ($r = 0.93$) (Table 4). The lowest values of exchangeable Mg and K were also observed under *E. globulus* compared to cultivated land. This study disagrees with Duguma et al. (2010) who had reported higher CEC under cereal farms compared to *Eucalyptus*.

Eucalyptus globulus in land use system of smallholder farmers

E. globulus is an integral element of smallholder farmers' land use system in the study area. Table 7 shows the summary of the household characteristics. The response rate to the questionnaires was 100%. The family size of the household ranges between 2 and 15 with an average of 7 persons, which is higher than of the west Shewa zone, 5.3 persons per household. About 45% of the households belong to medium wealth category. The landholding size per household ranges between 0.25 and 14 ha with an average of 3.4 ha. The average landholding size of the households is more than the west Shewa zone of average landholding size who posses on 1.93 ha. About 55% of the households had the landholding size of less than the average landholding size, whilst only about 22% of the households had the landholding size of more than 5 ha.

Assessment of household land allocation showed that about 46% of land was put under cultivation. Only 16% of the land was put under *E. globulus* showing less displacement of cultivated land for growing *E. globulus*. The rich households planted (1.1 ha) more *E. globulus* as compared to poor (0.66 ha) and medium (0.73 ha) households. The difference was statistically significant as determined by one way ANOVA ($F(2,107) = 5.724$, $P = 0.04$). The largest proportion of households (58%) preferred *E. globulus* because of its multiple uses. Only 22% and 20% of the households prefer *E. globulus* because of its fast growth and easy management, respectively. The need for *E. globulus* showed that households prefer to plant *Eucalyptus* on cultivated land (40%), grazing land (34%) and degraded land (26%). The finding agrees with Jenbere et al. (2012) who had reported the rich households plant *Eucalyptus* more than the poor and medium households.

Analysis of *Eucalyptus* in household food security endeavour shows smallholder farmers' strive to produce more food from what they produce. Table 8 shows crop production and pro-

T a b l e 7. Summary of household characteristics.

HH characteristics	N	Min	Max	Mean	Valid Percent
Age classes(no)	110	27	71	45	100
Family size (no)	110	2	15	7	100
Landholding size(ha)	110	0.25	14	3.4	100
Land under *E. globulus* (ha)	110	0.13	3	0.80	100
Wealth status(no)	110				100.0
Medium	49				44.5
Poor	32				29.1
Rich	29				26.4

ductivity of major crops at household level. The rich households produce, on an average, 18.48 quintal per year. The medium and poor households produce 18.35 and 15.16 quintal per year, respectively. The poor, medium and rich households need 15.22, 16.27 and 16.17 quintal per year, respectively, to support their families. Currently, most households support their families between 9 and 11 months from what they produce implying to buy food crops for the remaining months. Without using the inputs, the annual production was below the bottom line of what is required to support their families. To increase agricultural production, smallholder farmers' use input such as commercial fertilisers. An average poor, medium and rich households need 3247.92, 3545.19 and 3380.20 ETB, respectively, to buy fertilisers (Table 9). This is equivalent to selling, for instances, 4.39, 4.86 and 4.61 quintal of barley, respectively. Smallholder farmers need cash to buy inputs to increase productivity. *E. globulus* is assists to obtain cash for buying inputs and food. A land user can increase agricultural production when *E. globulus* became a component of the land use. Land under *E. globulus* reduced the quantity of fertilisers to be purchased as *E. globulus* does not need fertiliser application. Therefore, smallholder farmers saved some cash which could have been used to purchase commercial fertilisers. Growing *E. globulus* need less labour implying the labour can shift to agricultural production. The absence of *E. globulus* from the land use systems has shifted all food secure households to food insecure because of the fact that 4 quintal is deducted from average production (17.33 quintal) to purchase fertiliser and then 13.33 quintal is less than the average production needed to support family (15.89 quintal). Therefore, restricting smallholder farmers from planting *E. globulus* may negatively affect their livelihoods. *E. globulus*

T a b l e 8. Crops productivity per hectare with and without inputs at household level.

	Wealth categories			
	Poor	**Medium**	**Rich**	**Average**
Average annual production [quintal (metric)] Total production	15.16	18.35	18.48	17.33
Production needed to support family	15.22	16.27	16.17	15.89
Food self sufficiency (months)	9.88	10.65	10.8	
Average productivity per ha [quintal (metric)] Without inputs				
Barley	3.16	4.33	3.59	3.69
Wheat	4.44	4.82	4.5	4.59
Peas	1.19	2.04	2.43	1.89
Beans	2.17	3	3.43	2.87
Teff	3.52	3.47	3.26	3.42
Average	2.89	3.53	3.44	3.29
With inputs				
Barley	15.3	16.3	16.9	16.17
Wheat	19.9	20.2	21	20.37
Peas	3.7	6	6.8	5.5
Beans	6.5	8.9	9.2	8.2
Teff	13.5	14.6	14.2	14.1
Average	11.8	13.2	13.6	12.87

made a substantial contribution to the income of the households, even more than agricultural crops (Selassie, Ayanna, 2013, Kebebew, Ayele, 2010).

Table 10 showed that 83% of households were food secure because of *E. globulus* contribution to their income. Of the total households, 45.5% preferred to sale *E. globulus* in the case of emergency as compared to crops and livestock. The cash obtained from selling the *Eucalyptus* has filled the food shortage gaps of the families. *Eucalyptus* trees are regarded as insurance resource or life saviour, because they are cut and readily converted to cash during critical needs. The average annual income from *E. globulus* under poor, medium and rich households are 11219.27, 12207.48 and 15024.14 ETB/ha, respectively. In the study area,

T a b l e 9. Farm gate price of different crops at household level.

	Wealth categories			
	Poor	Medium	Rich	Average
Price of crops and inputs (ETB) Crops price				
Barley	739.84	729.08	732.76	733.89
Wheat	1,043.75	1,030.61	1,048.28	1,040.88
Peas	800	823.98	812.07	812.02
Beans	695.31	687.76	689.66	690.91
Teff	1,515.63	1,569.39	1,593.1	1,559.37
Cost of inputs per year	3,247.92	3,545.19	3,380.2	3391.1
Amount of crops sold to buy inputs [quintal(metric)]				
Barley	4.39	4.86	4.61	4.62
Wheat	3.11	3.44	3.22	3.26
Peas	4.06	4.3	4.16	4.17
Beans	4.67	5.15	4.9	4.91
Teff	2.14	2.26	2.12	2.17
Average	3.68	4	3.8	3.83

T a b l e 10. Household income contribution of *E. globulus*.

	Wealth categories					
	Poor	Medium	Rich	Average	Total	%
Rotation age (years)						
4 years	16	22	13		51	46.36
5 years	16	27	16		59	53.64
Rotation age (years) coppicing						
3 years	16	30	18		64	58.18
4 years	16	19	11		46	41.82
Average annual income from *E. globulus* (ETB)	11,219.2	12,207.48	15,024.14	12,816.96		
Income from *E. globulus* satisfy household food security						
Yes	20	43	28		91	82.7
No	12	6	1		19	17.3

E. globulus can start to provide income from the first and second rotation of 4 or 5 and 3 or 4 years, respectively, excluding in-between benefits. Kebebew and Ayele (2010) reported that *E. globulus* serves as a cash crop to smallholder farmers and contributed significantly to farmers livelihoods. Smallholder farmers harvest *E. globulus* starting from year 3 to 4 depending on the perceived products.

Conclusion

The results of the study showed higher soil bulk density, soil OC content and TN under *Eucalyptus* as compared to cultivated and degraded land, which implies less detrimental effect of *Eucalyptus* on soil fertility. The clay particles and CEC were significantly higher under the *E. globulus* compared with all land uses except forest. From livelihoods perspective, *E. globulus* plays an important role in addressing food security. The results also showed that 82.7% households were food secure from the income obtained from *E. globulus*. The average annual income from *E. globulus* under poor, medium and rich households are 11 219.27, 12 207.48 and 15 024.14 ETB/ha, respectively. If *E. globulus* can be restricted from the land use systems, the livelihoods of all households depending on the activity will be negatively affected. Therefore, the issue of discouraging *E. globulus* must take into account the current contribution of *E. globulus* to agricultural production.

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References

Abbasi, M.K., Zafar, M. & Khan S.R. (2007). Influence of different land-cover types on the changes of selected soil properties in the mountain region of Rawalakot Azad Jammu and Kashmir nutrient cycling. *Agroecosystem*, 78, 97–110. DOI: 10.1007/s10705-006-9077-z.

Abebaw, D., Fentie, Y. & Kassa B. (2010). The impact of a food security program on household food consumption in Northwestern Ethiopia: A matching estimator approach. *Food Policy*, 35, 286–293. DOI: 10.1016/j.foodpol.2010.01.002.

Brady, N.C. & Weil R.R. (2002). *The nature and properties of soils*. New Jersey: Prentice-Hall.

Chapman, H.D. (1965). Cation exchange capacity. In C.A. Black, L.E. Ensminger & F.E. Clark (Eds.), Methods of soil analysis. *Agronomy*, 9, 891–901.

Cochran, W.G. (1977). *Sampling techniques*. New York: John Wiley and Sons.

Diao, X. & Pratt A.N. (2007). Growth options and poverty reduction in Ethiopia – An economy-wide model analysis. *Food Policy*, 32, 205–228. DOI: 10.1016/j.foodpol.2006.05.005.

Duguma, L.A., Hager, H. & Sieghardt M. (2010). Effects of land use types on soil chemical properties in smallholder farmers of central highland Ethiopia. *Ekológia (Bratislava)*, 29, 1–14. DOI: 10.4149/ekol_2010_01_1.

Duguma, L.A. & Hager H. (2011). Farmers' assessment of the social and ecological values of land uses in central highland Ethiopia. *Environ. Manag.*, 47, 969–982. DOI: 10.1007/s00267-011-9657-9.

Fageria, N.K. (2009). *The use of nutrients in crop plants*. New York: CRC Press.

Frank, B. (1990). *Interpreting soil test results: What do all the numbers mean*? Collingwood: CSIRO Publishing.

Getachew, F., Abdulkadir, A., Lemenih, M. & Fetene A. (2012). Effects of different land uses on soil physical and chemical properties in Wondo Genet Area, Ethiopia. *New York Science Journal*, 5(11), 110–118. DOI: 10.7537/marsnys051112.16.

Haile, G., Lemenih, M., Itanna, F. & Senbeta F. (2014). Impacts of land uses changes on soil fertility, carbon and nitrogen stock under smallholder farmers in central highlands of Ethiopia: Implication for sustainable agricultural landscape management around Butajira area. *New York Science Journal*, 7(2), 27–44. DOI: 10.7537/marsnys070214.04.

Jenbere, D., Lemenih, M. & Kassa H. (2012). Expansion of eucalypt farm forestry and its determinants in Arsi Negelle district, South central Ethiopia. *Small-Scale Forestry*, 11, 389–405. DOI: 10.1007/s11842-011-9191-x.

Karltun, E., Lemenih, M. & Tolera M. (2013). Comparing farmers' perception of soil fertility change with soil properties and crop performance in Beseku, Ethiopia. *Land Degrad. Dev.*, 24, 228–235. DOI: 10.1002/ldr.1118.

Kebebew, Z. & Ayale G. (2010). Profitability and household income contribution of growing *Eucalyptus globulus* Labill. to smallholder farmers: the case of central Highland of Oromia, Ethiopia. *European Journal of Applied Sciences*, 2, 25–29.

Kebede, Y. & Raju S.A.J. (2011). Effect of land use/land cover change on soil properties in the Hare River Watershed, Ethiopia. *The Ecoscan*, *An International Quarterly Journal of Environmental Sciences*, 5, 69–74.

Kidanu, S., Mamo, T. & Stroosnijder L. (2005). Biomass production of Eucalyptus boundary plantations and their effect on crop productivity on Ethiopian highland Vertisols. *Agrofor. Syst.*, 63, 281–290. DOI: 10.1007/s10457-005-5169-z.

Kristjanson, P., Neufeldt, H., Gassner, A., Mango, J., Kyazze, F.B., Desta, S., Sayula, G., Thiede, B., Förch, W., Thornton, P.K. & Coe R. (2012). Are food insecure smallholder households making changes in their farming practices? Evidence from East Africa. *Food Security*, 4, 381–397. DOI: 10.1007/s12571-012-0194-z.

Lemenih, M., Karltun, E. & Olsson M. (2005). Assessing soil chemical and physical property responses to deforestation and subsequent cultivation in smallholders farming system in Ethiopia. *Agric. Ecosyst. Environ.*, 105, 373–386. DOI: 10.1016/j.agee.2004.01.046.

Mekonnen, A. (2000). Valuation of community forestry in Ethiopia: A contingent valuation study of rural households. *Environmental & Developmental Economics*, 5, 289–308.

Mekuria, W. & Aynekulu E. (2011). Exclosure land management for restoration of the soils in degraded communal grazing lands in northern Ethiopia. *Land Degrad. Dev.*, 24(6), 528–538. DOI: 10.1002/ldr.1146.

Mengist, M. (2011). *Eucalypts plantations in the highlands of Ethiopia revisited: A comparison of soil nutrient status after the first coppicing*. Master Thesis - Mountain Forestry Program.

Miller, R.W. & Donhue R.L. (1997). *Soil in our environment*. Prentice-Hall.

Negra, C., Vermeulen, S., Barioni, L.G., Mamo, T., Melville, P. & Tadesse M. (2014). Brazil, Ethiopia, and New Zealand lead the way on climate-smart agriculture. *Agriculture & Food Security*, 3, 19. DOI: 10.1186/s40066-014-0019-8.

Nelson, D.W. & Sommers I.E. (1982). Total carbon, organic carbon and organic matter. Chemical and microbiological properties. *American Society of Agronomy*, 9, 639–679.

Newby, J., Cramb, R. & Sakanphet S. (2014). Forest transitions and rural livelihoods: Multiple pathways of smallholder Teak expansion in Northern Laos. *Land*, 3, 482–503. DOI: 10.3390/land3020482.

Pretzsch, J. (2005). Forest related rural livelihood strategies in national and global development. *Forests, Trees & Livelihoods*, 15, 115–127. DOI: 10.1080/14728028.2005.9752515.

Sahlemedhin, S. & Taye B. (2000). *Procedure for soil and plant analysis*. Technical Paper No. 74. National Soil Research Center: Ethiopian Agricultural Research Organization.

Selassie, Y.G. & Ayanna G. (2013). Effects of different land use systems on selected physic-chemical properties of soils in northwestern Ethiopia. *J. Agric. Sci.*, 5, 112–120. DOI: 10.5539/jas.v5n4p112

Selassie, Y.G., Anemut, F. & Addisu S. (2015). The effects of land use types, management practices and slope classes on selected soil physico-chemical properties in Zikre watershed, North-Western Ethiopia. *Environmental Systems Research*, 4, 3. DOI: 10.1186/s40068-015-0027-0.

Teshome, A., Graaff, J., Ritsema, C. & Kassie M. (2014). Farmers' perceptions about the influence of land quality, land fragmentation and tenure systems on sustainable land management in the north western Ethiopian highlands. *Land Degrad. Dev.*, 27(4), 884–898. DOI: 10.1002/ldr.2298.

Tisdale, S., Nelson, W., Beaton, J. & Havlin J. (2002). *Soil fertility and fertilizers*. New Jersey: Prentice-Hall.

Van der Veen, A. & Tagel G. (2011). Effect of policy interventions on food security in Tigray, Northern Ethiopia. *Ecology & Society*, 16, 18. http://www.ecologyandsociety.org /vol16/iss1/art18/.

Van Reeuwijk, L.P. (1992). *Procedures for soil analysis*. Wageningen: International Soil Reference and Information Center (ISRIC).

Yitaferu, B., Abewa, A. & Amare T. (2013). Expansion of eucalypts woodlots in the fertile soils of the highlands of Ethiopia: Could it be a treat on future cropland use? *J. Agric. Sci.*, 58, 97–107. DOI: 10.5539/jas.v5n8p97.

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REMEDIATION POTENTIAL OF FOREST FORMING TREE SPECIES WITHIN NORTHERN STEPPE RECLAMATION STANDS

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Abstract

Zverkovskyy V.M., Sytnyk S.A., Lovynska V.M., Kharytonov M.M., Lakyda I.P., Mykolenko S.Yu., Pardini G., Margui E., Gispert M.: Remediation potential of forest forming tree species within northern steppe reclamation stands. Ekológia (Bratislava), Vol. 37, No. 1, p. 69–81, 2018.

The aim of the research was to study the features of accumulation of heavy metals by assimilation apparatus of coniferous and deciduous arboreous plants. The research identified excess of factual concentrations for Arsenic in mining rock in relation to values stated in IPC (indicative permissible concentrations). It is stated that the metals can be divided into three groups according to their absolute content in unit of foliage biomass. The element of excessive concentration is Mn, medium concentration is characteristic for Pb and Zn and low concentration is observed for Sb, Cr, As, Cu, Ni and Sn. Calculation of coefficient of biological accumulation of the metals under research has shown its high values for Crimean pine. The data presented for Black locust indicate low values of coefficient of biological accumulation, which is best noticeable for Chromium, Antimony and Tin. It is determined that a small amount of Sb and Sn are a subject to uptake by Black locust leaves, whilst for Crimean pine needles, Sb and As are characterised by the lowest inflow. The average content of lead is 209.11 kg·ha^{-1} for Crimean pine in all age groups of trees, whilst for Black locust, this index is only 15.52 kg·ha^{-1}, which is 13.5 times less. Zinc accumulation is better performed by Black locust leaves, and it gradually decreases with increasing age. No definite trend of redistribution and subsequent accumulation of copper depending on tree species and age was found.

Key words: Black locust (*Robinia pseudoacacia* L.), Crimean pine (*Pinus pallasiana* L.), northern steppe of Ukraine, mining rock, heavy metals, bioaccumulation coefficient.

Introduction

Rapid development of industry in all countries of the world leads to local pollution due to emissions from industrial enterprises that has significantly exceeded the maximum permissible sanitary norms during recent decades (Jarup, 2003). Large-scale coal mining activities result in substantial erosion and pollution of vast areas (Kuznetsova et al., 2010; Alekseenko et al., 2017). Technogenic influence leads to global disturbances of ecological systems; therefore, an important task is to forecast changes occurring in ecological systems under the influence of anthropogenic factors (Alexander, 2000; Risto et al., 2005; Shahid et al., 2014).

In the process of coal mining, particularly under conditions of operation of mines, significant disturbance and pollution of land occurs, being especially relevant for agricultural lands. Also, withdrawal of significant areas from use is being carried out (Allen et al., 1995; Kaar, 2002). As a result, poly-elemental man-made anomalies are formed, which can cover all components of the biosphere. Arboreous vegetation that grows in such conditions primarily serves as a mechanical barrier for aerogenic migration of metals and prevents involvement of elements in the process of small biological cycling of substances (Chodak, Niklińska, 2010; Khokhotva, 2010; Marmiroli et al., 2011; Fernández et al., 2017).

Soil and plant objects are involved in all processes of transformation and migration of substances occurring in the biosphere and associated with functioning of ecosystems and with metabolism of substances in living organisms (Hüttl 1998; Prasad, Hagemeyer, 1999; Hüttl, Weber, 2001; Marko-Worłowska et al., 2011; Thapa et al., 2012). Heavy metals coming from different sources are accumulated in the soil; their subsequent redistribution depends on chemical nature of the elements as well as on specific properties of soils and plants (Kabata-Pendias, 2011).

In modern conditions of anthropogenic pressure intensification, with the constant 'enrichment' of habitats of plants with compounds of heavy metals, the environmental factor often impedes implementation of a genetic programme for absorption of chemical elements by plants (Saarelaa et al., 2005; Verbruggen et al., 2009; Appenroth, 2010; Chudzińska et al., 2016).

Many authors believe that the state of assimilation apparatus of arboreous plants can be used as an object of environmental monitoring, which is associated with the assessment of their environment stabilising role, as a mediator of pollutants spreading into the environment (Dmuchowski, Bytnerowicz, 1995; Pöykiö et al., 2010; Kabata-Pendias, 2011; Pietrzykowski, Socha, 2011; Pietrzykowski et al., 2014).

The purpose of this research was studying the peculiarities of accumulation of elements of the group of heavy metals in assimilation apparatus of coniferous and broadleaved tree species that grow under conditions of mining rock.

Material and methods

Sample plots for the research were established on the forest reclamation site of mine 'Pavlohradska' in Pavlohrad city, Dnipropetrovsk region, Ukraine. Samples of vegetal material were taken only from living plants, without any signs of damage and diseases, that were growing on mining rock. The object of the study was represented by foliage biomass (leave and needle biomass) of Black locust (*Robinia pseudoacacia* L.) and Crimean pine (*Pinus pallasiana* L.) trees.

Mine rock was defined as heavy loam, light and middle clays. Mine rock was characterised by adverse water–physical properties. The sulphur content in mine rocks indicated that the amount of pyrite was changing from 1.8% to 3.3%. Acidity (pH) of mine rock was 4.8.

Determination of concentrations of chemicals in mine rock and vegetal material was carried out by the method of inductively coupled plasma-optical emission spectrometry (ICP-OES) using Technologies 5100 (Agilent) spectrometer with an inductively coupled plasma.

Content of the following inorganic contaminants was researched: Cu, Ni, Cd, Zn, Pb, Cr, Sb, Sn and Mn, amongst which Ni, Mn, Co and Cu represent the so-called transition metals, compounds of which have significant biological activity.

To estimate the processes of intake and accumulation of heavy metals in foliage biomass of the woody species under research, the coefficient of biological accumulation was applied, as a ratio of an average content of heavy metals in foliage to their average content in mine rock:

$K_{bac} = C_f / C_{sub}$,

where K_{bac} is the bioaccumulation coefficient, C_f is the metal content in foliage biomass expressed in mg/g and C_{sub} is the metal content in mining rock expressed in mg/g.

In order to calculate the gross content of elements belonging to the group of heavy metals in foliage live biomass of the tree species, on the first stage of research, their average values were determined in dry state using the approach and methodology described by Lakyda (2003). The research was conducted at different ages of the investigated species and indicated as young-age (1–20 years), middle-age (21–40 years), maturing (41–60 years), mature (61–80 years), overmature (81–100 years) age groups.

For quantification of assimilation component of the aboveground live biomass, 45 model trees of each investigated tree species were analysed and biometric parameters of 250 model trees were determined. In order to determine dependency from the main biometric indices (diameter, height) for foliage biomass of Black locust and Crimean pine trees, analytical search for adequate models by means of Statsoft STATISTICA 10 software was performed.

Results

At the first stage of this research, the content of chemicals in the substrate for growing arboreous plants was determined and their compliance with the state ecological and sanitary standards was assessed (Table 1).

The results of analysis of the actual concentrations of chemicals in mine rock with pH 4.8, in relation to the IPC values, has shown an excess for only one metal – arsenic (5.2 times). The comparative analysis of compliance with the normative values of maximum permissible concentrations (MPC) has demonstrated the absence of excess for only one amongst the nine substances under research – manganese. Indicators of content of other inorganic contaminants in mine rock have exceeded the MPC values for chemical sub-

T a b l e 1. Content of inorganic contaminants in mine rock on the area of forest reclamation.

Index	Chemical elements								
	As	Sb	Zn	Pb	Cr	Ni	Cu	Mn	Sn
Concentration, mg/kg	25.8 ± 2.70	40.5 ± 1.44	56.5 ± 1.57	40.6 ± 4.58	93.9 ± 2.21	43.1 ± 2.53	27.5 ± 0.19	164.5 ± 1.25	40.5 ± 1.40
Maximum permissible concentration*, mg/kg	2.0 g/q*****	4.5 g/q	23.0 m/f***	32.0 g/q	6.0 m/f	4.0 m/f	3.0 m/f	1500.0 g/q	n/r***
Indicative permissible concentration**, mg/kg	5.0	n/r	110.0	65.0	n/r	40.0	66.0	n/r	n/r

Notes: * – values of maximum permissible concentrations (MPC) of chemical substances in soils by indices of harmfulness; ** – values of indicative permissible concentrations of gross content of chemical substances in soils for different types of land use; n/r*** – content of chemical substance is not regulated; m/f*** – mobile form of chemical substance; g/q***** – gross quantity of chemical substance.

stances in soil to various extent: Pb exceeded 1.3 times, Zn 2.5 times, Sb 9.0 times, Cu 9.2 times, Ni 10.8 times, As 12.9 times, Cr 15.7 times and Sn 20.3 times.

As metals are present in soils in two forms – in solid form and in soil solution – their form of existence, transformation and, most importantly, availability to plants are determined by medium reaction, chemical composition of soil solution and content of organic substances (Wuana, Okieimen, 2011).

The phytotoxicity of substances depends on their chemical properties as well as their ability to form complexes and, above all, their concentration. In most cases, metals are ranked by degrees of toxicity as follows: Cu > Ni > Cd > Zn > Pb > Hg > Fe > Mo > Mn (Brown et al., 1990; Grishko et al., 2012). Changes may occur in the given series because of genetic, physiological and biochemical characteristics of plants and their growing conditions.

A complex of edaphic factors determines the transformation and direction of migration of chemicals into vegetal organs and tissues. The inorganic contaminants under research on acidic substrates (represented by the studied mine rock with pH 4.6–4.8) have the following degrees of mobility: Ni, Cr, Pb and As have low mobility and Mn, Cu and Zn are mobile (Wuana, Okieimen, 2011).

The next stage of the research was aimed at the determination of concentrations of inorganic contaminants in foliage biomass of Black locust and Crimean pine, the results of which are presented in Fig. 1.

The studied chemical substances in the assimilating fraction of aboveground live biomass of tree species were divided into three groups based on the concentration: (1) substances with excess concentration (113.7–510.6 mg/kg of dry mass), Mn; (2) substances

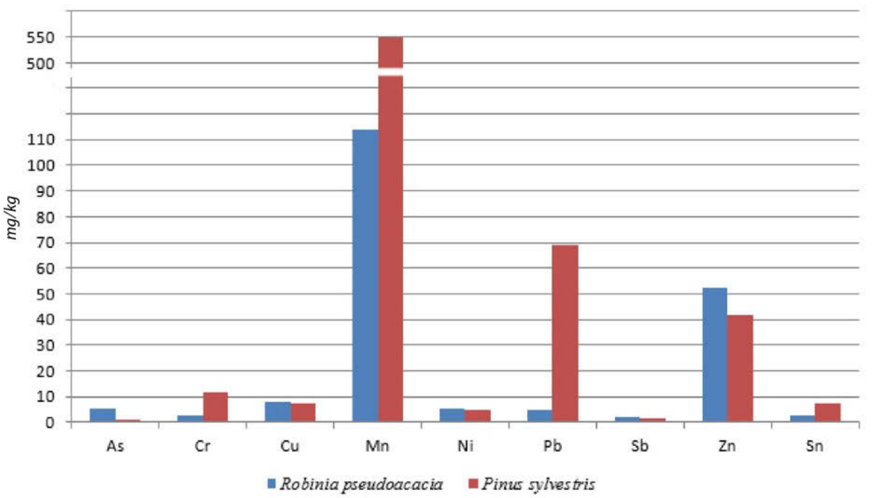


Fig. 1. Inorganic contaminants concentration in foliage biomass of Black locust and Crimean pine.

with medium concentration (41.5–69.2 mg/kg of dry mass), Pb, Zn; and (3) substances with low concentration (0.8–11.9 mg/kg of dry mass), Sb, Cr, As, Cu, Ni and Sn.

A comparative analysis of concentration of chemical substances under research in foliage biomass of Black locust and Crimean pine has enabled to reveal significant differences in the concentration of metals such as manganese, arsenic, chromium, lead and tin.

The most significant difference is recorded for the concentration of lead: in the needles of Crimean pine, the concentration of this element was more than 12 times higher than that in Black locust leaves. More significant accumulation in Black locust foliage biomass compared to the concentration in Crimean pine needles was found for only two metals: arsenic (6.7 times) and antimony (2.2 times). The reverse trend was observed in relation to manganese, chromium and tin: their accumulation in Crimean pine needles exceeded those found in the leaves of Black locust, which are 4.5, 4.4 and 2.9 times, respectively. The presence of identical concentrations in foliage biomass of Crimean pine and Black locust was found for copper, nickel and zinc. These substances are physiologically significant for plants; therefore, their concentration is identical in assimilation apparatus of different tree species without any signs of either damage and lesions of different abiotic and biotic aetiology, or intoxication, which demonstrates physiologically optimal value for the performance of physiological and biochemical reactions.

Excessive gross content and significant concentration of mobile forms of inorganic contaminants in soils result in their accumulation and higher concentration in vegetal tissues. However, this process is species specific. Therefore, to characterise remediation potential for the tree species under research, the coefficient of biological accumulation of metals by foliage fraction of their aboveground live biomass was calculated (Table 2).

T a b l e 2. Values of bioaccumulation coefficients for inorganic contaminants.

Tree species	**Chemical substances**								
	As	**Cr**	**Cu**	**Mn**	**Ni**	**Pb**	**Sb**	**Zn**	**Sn**
Robinia pseudoacacia	0.178	0.029	0.282	0.691	0.127	0.152	0.043	0.924	0.069
Pinus pallasiana	0.031	0.127	0.281	3.104	0.107	1.702	0.020	0.734	0.184

According to scale developed by Avessalomov (1987), manganese and lead are recognised as elements of high accumulation ($10 > K_{bac} \geq 1$); all other metals under research are considered amongst elements of weak accumulation ($1 > K_{bac} \geq 0.1$). High coefficient of biological accumulation values for these metals was recorded only for the specimens of Crimean pine. The data presented for another studied tree species, Black locust, has shown very low values of coefficient of biological accumulation for substances such as chromium, antimony and tin.

The results of determining the accumulative properties of assimilative fraction of arboreous plants have allowed to establish that when growing on mine rock, this fraction of live biomass is capable of accumulating inorganic contaminants from 1.46 to 2134.35 kg/ha for Crimean pine and from 4.42 to 441.08 kg/ha for Black locust, depending on the age of the model trees (Table 3).

T a b l e 3. Content of inorganic contaminants in foliage biomass of Black locust and Crimean pine.

Age group, years	Foliage biomass, kg·ha^{-1}	Elements of the group of heavy metals, kg·ha^{-1}								
		As	Cr	Cu	Mn	Ni	Pb	Sb	Zn	Sn
1–20	3880 1830	20.91 1.46	10.63 21.87	30.11 14.16	441.08 934.42	21.30 8.44	18.20 126.54	6.84 1.46	202.65 75.87	10.20 13.63
21–40	3480 4180	18.76 3.34	9.53 49.95	27.00 32.35	395.61 2134.35	19.11 19.27	16.32 289.05	6.12 3.34	181.76 173.30	9.15 31.14
41–60	3370 3350	18.18 2.68	9.23 40.03	26.15 25.93	383.10 1710.54	18.50 15.44	15.80 231.65	5.93 2.68	176.02 138.89	8.86 24.96
61–80	2510 3130	13.53 2.50	6.88 37.4	19.48 24.23	285.34 1598.21	13.78 14.43	11.77 216.44	4.42 2.50	131.10 129.77	6.60 23.32
81–100	2630 –	2.10 –	31.43 –	20.36 –	1342.90 –	12.12 –	181.86 –	2.10 –	109.04 –	19.59 –

Notes: numerator – *Robinia pseudoacacia*, denominator – *Pinus pallasiana*.

Significant differences were found regarding nature of accumulation of individual metals, which is mainly reasoned by their content in mine rock, nature of intake and translocation of a metal in vegetal tissues and by different potential for accumulation of the two tree species under research.

When researching the content of inorganic contaminants in Black locust foliage biomass, it was determined that the lowest accumulation in assimilation organs is characteristic for metals such as Sb and Sn, whereas for Crimean pine, minimal accumulation is observed for Sb and As. On the contrary, translocation of manganese in assimilation fraction of both the hardwood and coniferous wood species occurs most intensively. Significantly higher concentrations of this metal were recorded in foliage biomass of Crimean pine, especially in the second age group, which is primarily due to the formation of predominant assimilation live biomass in trees of the specified age group.

The second position in terms of gross content in foliage fraction of both investigated wood species is presented by lead and zinc, which are elements with synergistic action when accumulated in soils (Grishko et al., 2012). Upon intake in the aboveground live biomass, lead with low mobility is accumulated in substantial quantities in assimilative fraction of trees of all ages ranging from 126.54 kg/ha (young-age stands) to 289.05 kg/ha (middle-age group). An average content of the analysed element for Crimean pine included in this research is 209.11 kg/ha, whereas that for Black locust is only 15.52 kg/ha[1], which is 13.5 times less. Such differences can be explained by different lifetime of assimilative apparatus in coniferous and broadleaved tree species.

Another tendency of gross content variability depending on the age of Crimean pine and Black locust trees was found for zinc. Unlike lead, this metal is easily available to plants, and its accumulation increases linearly with increasing concentrations in soils (Eide, 2006). Sequestration of this metal is greater in foliage biomass of Black locust and gradually decreases with increasing age of the model trees, which is directly proportional to decrease in the ability to form assimilative organs for this tree species. The difference between maximal and minimal content of this metal in the young-age and the mature age groups is 35%. For Crimean pine needles, it was found that zinc is accumulated in lower quantities, with maximum accumulation in live biomass of trees belonging to the middle-age group.

Distribution of copper in photosynthetic organs of the studied tree species did not reveal any particular age-dependent variability and specificity of its accumulation. A tendency towards age-driven decrease in gross content of nickel in foliage biomass was found for Black locust (Table 3). The mentioned pattern was also found for the coniferous species with a noticeable decrease in the concentration of this metal in the needles of overmature trees. The research did not reveal species-specific peculiarities of build-up, redistribution and subsequent accumulation of this metal.

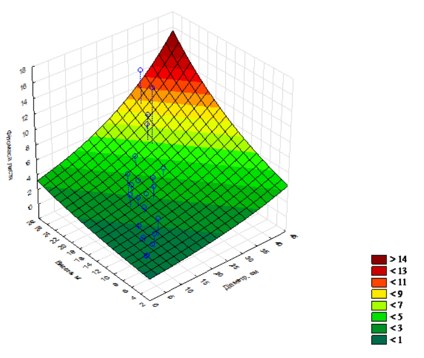
According to the results of our research, it was found that for Crimean pine, processes of translocation of metals such as As, Sb and Ni to assimilation apparatus are slowed down as compared to Cr, Mn and Zn, whereas for Black locust, the minimal content in foliage is registered for metals such as Cr and Sn. The maximal gross content is found for manganese; Crimean pine accumulates this metal at a 5 times higher rate than Black locust, and lead is accumulated 16 times more intensively by Crimean pine than by Black locust.

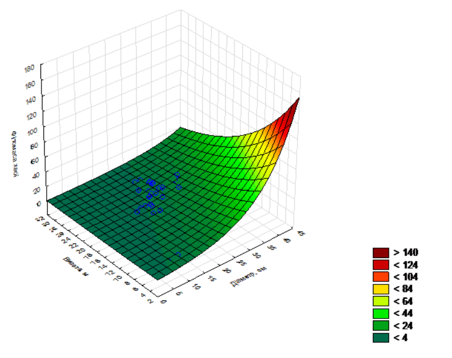
The content of chromium is highest in specimens of Crimean pine aged between 21 and 40 years and lowest in young-age group. The content of this metal in foliage biomass of *R. pseudoacacia* logically decreases with increase in age and reaches its minimum – 6.88 kg/ha – in leaves of mature model trees.

In order to determine the dependency of accumulation of inorganic contaminants in assimilative organs of the two tree species, a search for an adequate mathematical model

T a b l e 4. Equations describing dependency of foliage biomass on the main biometric indices of model trees.

Robinia pseudoacacia		*Pinus pallasiana*	
Equation type	R^2	Equation type	R^2
$P_{fol} = 0.7909 \cdot \exp(0.3512\, d) \cdot \exp(0.0499\, h)$	0.46	$P_{fol} = 3.2256 \cdot \exp(0.0891\, d) \cdot \exp(0.0683 h)$	0.42





T a b l e 5. Main distributional statistics of indices of model trees' foliage fraction.

Index	Minimal value	Maximal value	Mean value	Standard deviation	Skewness	Kurtosis
Robinia pseudoacacia						
Foliage biomass, kg	0.14	13.36	3.61	0.02	1.877	3.755
Diameter, cm	2.70	40.00	17.22	0.09	0.255	-0.294
Height, m	3.70	25.80	15.05	0.07	-0.208	-0.789
Pinus pallasiana						
Foliage biomass, kg	0.95	19.62	5.82	0.03	1.493	3.761
Diameter, cm	4.00	38.90	20.61	0.12	-0.468	0.754
Height, m	4.20	30.00	19.06	0.42	-1.268	1.027

was performed, in order to account for dependency of formation of foliage biomass on the main mensurational indices of model trees (Tables 4, 5).

The presented dependencies are the most significant and envisage further practical application of the developed two-factor models accounting for two mensurational indices – height and diameter of a sample tree.

Discussion

To consider all the variety of soil and geochemical conditions when establishing maximum permissible concentrations of inorganic substances is hardly possible. Therefore, certain concentrations of chemical substances in mine rock reveal an understanding of their phytotoxic effects on arboreous plants. An important point in substantiating soil toxic safety for formation of artificial plantation systems is the consideration of concentrations of inorganic contaminants that alter physiological and biochemical processes and are toxic to plants. Regarding location and transformation of chemical substances in the soil and their availability to plants, the literature presents data proving different directions of migration of inorganic contaminants from soil to plants and their absorption. According to Appenroth (2010), inorganic contaminants are predominantly concentrated in a 10-cm layer; however, given low pH values, which is characteristic for the substrate under research, a large proportion of metals are transferred to soil solution, which makes them more accessible to root systems of plants.

The results of comparative analysis of determined actual concentrations of chemical substances in foliage biomass of aboveground live biomass of Crimean pine and Black locust with concentrations indicated by different authors as optimal for functioning of plants are shown in Table 6.

According to the results of our research, the highest migration capacity and sequestration in assimilation fraction of aboveground live biomass of the two tree species are observed for zinc and manganese, which are physiologically significant substances for plant metabolism. It is worth noting that gross content of these metals in mine rock does not exceed the values indicated in MPC (Mn) or are slightly excessive (Zn). The obtained data are

T a b l e 6. Limits of fluctuations of optimal concentrations of chemical substances in plants.

Inorganic contaminants, mg/kg	Mineev V.H., 1990	Chertko N.K., 2008	Kovalskyy V.V., 1974	Kabata-Pendias A., 1989	Nieber et. al., 1978	Concentrations in foliage biomass of the researched plants
Chromium	0.2 – 1.0	–	–	0.02 – 0.2	–	2.8 – 12.0
Copper	2.0 – 12.0	5.0 – 30.0	3.0 – 12.0	2.0 – 20.0	≤ 30	7.7
Nickel	0.4 – 3.0	≤ 1.0	–	0.1 – 2.7	–	4.6 – 5.5
Lead	0.1 – 5.0	1.5 – 14.0	–	0.05 – 5.0	≤ 30	4.7 – 69.1
Tin	0.8 – 6.0	–	–	–	–	2.6 – 7.5
Zinc	15.0 – 150.0	15.0 – 150.0	20.0 – 60.0	–	≤ 100	41.6 – 52.3
Manganese	–	20.0 – 300.0	20.0 – 60.0	17.0 – 334.0	–	113.7 – 510.6

consistent with the results obtained by Grishko et al. (2012), which indicate that the depth of penetration of chemicals in contaminated soils is usually not more than 20 cm, but in cases of severe contamination, they can penetrate down to a depth of 1.5 m. According to Grishko et al. (2012), amongst all the metals, zinc has the highest migration potential and uniformity of distribution in the soil layer of 0–20 cm. The research results state that a decrease in pH by two units leads to 3.8–5.5 times increase in mobility of zinc. Phytotoxicity of zinc is noted by many authors, especially on acid soils (Alexander, 2000; Eide, 2006; Fernandez et al., 2017). Manifestation of toxicity signs of zinc in plants is noted when its content in tissues reaches 300–500 mg/kg of dry matter. Tolerant species may weaken the effect of excessive zinc concentrations by metabolic adaptation and formation of complexes or by limiting the presence of the element in cells or by converting it into insoluble form in storage tissues. According to Prasad and Hagemeyer (1999), zinc concentration of 200 mg/kg of dry plant material causes a toxic effect on plants.

The literature does not contain data indicating an undoubted need for lead for functioning of any plant species; only information on growth stimulating effect of low concentrations of compounds of this metal is available. The described effects of inhibition of plants' metabolism arise from low level of the element. Interaction of lead with other elements under different environmental conditions does not allow to reliably determine which metal concentrations are toxic to plants. Data on interaction of lead with other microelements are available only for zinc and cadmium. Stimulating effect of Pb^{2+} ions on the absorption of cadmium by plant roots may be a secondary effect associated with the disturbance of transmembrane transport processes. Antagonism of zinc and lead is manifested in a mutually unfavourable effect on the transport of both the elements from roots to aboveground part of plants (Itoh et al., 2006).

Regarding migration of lead, it is mainly indicated that the element is able to be accumulated in the soil. According to Lin et al. (2004), this metal accumulates only in the surface layer, 0–2.5 cm, and its ions are characterised by low mobility even at low pH values. For different types of soils, the rate of leaching of this metal varies from 4 to 30 g/ha/year. However, our research has found that when the total amount of lead in the substrate was within the range of indicative permissible concentrations, Crimean pine needles revealed a significant sequestration capacity in relation to lead, which is confirmed by high value of accumulation coefficient.

Significant migration potential in acidic environment is observed for copper and nickel. Migration of the latter is complex: on one hand, this metal enters plants from the soil in a form of soil solution; on the other hand, its amount in the soil is replenished as a result of destruction of soil minerals, dieback of plants and microorganisms. Coefficients of biological accumulation of these metals in assimilation fraction of the researched tree species that were calculated in the course of our research have shown a considerable similarity: Crimean pine needles and Black locust leaves concentrated copper (K_{bac}Cu = 0.28) and nickel (K_{bac}Ni = 0.11) with the same intensity, provided a significant excess over MPC of these substances in the substrate.

High concentrations of copper may cause toxic effects on plants. Copper is referred to as inactive metal, which actively binds mainly with cell membranes in plant roots. This is confirmed by the high metal concentrations in the soil, and a significant decrease in its concentration in assimilation fraction of plants. Owing to its important role in functioning of enzymes and variable valency, other ions with similar protein affinity may exhibit antagonistic interaction. The mechanism for copper and zinc absorption is identical, and therefore, each of them, because of mutual competition, can inhibit the absorption of the other by a root system. Signs of copper deficiency in plants are observed at different content in cells: content of copper below 2 mg/kg is unfavourable for most plants (Pietrzykowski et al., 2014).

Currently, the necessity of nickel for plants is a controversial question, but toxicity of its high concentrations is obvious. For different plant species, a range of toxic concentrations of nickel varies widely, and concentrations of excessive and toxic levels vary from 10 to 100 mg/kg of soil. With an excess of nickel, absorption of nutrients is drastically reduced. Jarup (2003) has found a fact of reduction in inflow and transfer of a number of elements – Zn, Cu, Ca, Mg and Mn – in plants. However, when nickel concentration is excessive, inhibition of the activity of meristem was observed, which was expressed in the suppression of differentiation of tissues, decrease in number of cell layers and vascular beams. Before the appearance of visually noticeable symptoms of acute toxicity, elevated concentrations of nickel in plant tissues suppress transpiration and photosynthesis processes, whilst replacing the central magnesium atom with the nickel atom. Thapa et al. (2012) indicate that mobility of nickel in soil depends on the concentration of organic matter, mainly humus acids, and pH of the medium. The determined coefficients of biological accumulation for nickel in relation to the researched tree species have shown a considerable resemblance: foliage of both tree species was absorbing and concentrating the element with same insignificant intensity (K_{bac}Ni = 0.11).

Antimony is not considered to be a vital metal for plants. It is known that its soluble forms are actively absorbed by plants from the soil. The physiological impact of antimony is similar to that of arsenic: it binds with thiol groups of proteins and participates in enzymatic reactions as a competitor of vital metabolites. In our studies, given the high content of antimony in technosol and excess over MPC, its concentration in foliage of the researched tree species was taking place identically: concentration of Sb in leaves of Black locust and needles of Crimean pine was between 0.02 and 0.04 mg/kg, which is very low for accumulation. Kubatbekov et al. (2012) indicate that the content of antimony in tissues of trees and shrubs growing in areas of ore mineralisation was 7−50 mg/kg, whereas according to our data, the concentration of this metal in foliage of Black locust was 1.7 mg/kg and in needles of Crimean pine was only 0.8 mg/kg, which cannot compete with plants that are recognised as antimony accumulators.

Conclusion

Mine rock, which served as a substrate for growing arboreous plants of a remediation stand, was characterised by excessive content of inorganic contaminants, with the exception of manganese. Actual concentrations of metals in mine rock exceeded those stated in state MPC norms: Pb exceeded 1.3 times, Zn 2.5 times, Sb 9.0 times, Cu 9.2 times, Ni 10.8 times, As 12.9 times, Cr 15.7 times and Sn, 20.3 times.

Per unit of foliage biomass of Black locust and Crimean pine, the metals were divided into three groups: (1) substances with excess concentration (113.7–510.6 mg/kg), Mn; (2) substances with medium concentration (41.5–69.2 mg/kg), Pb, Zn; and (3) substances with low concentration (0.8–11.9 mg/kg), Sb, Cr, As, Cu, Ni, Sn.

In Crimean pine needles, sequestration of lead was 12 times higher in comparison with its content in leaves of Black locust. The tendency for higher accumulation in Crimean pine needles was detected in relation to manganese, chromium and tin: their accumulation exceeded the corresponding values in Black locust leaves by 4.5, 4.4 and 2.9 times, respectively. Higher accumulation in Black locust foliage biomass compared to Crimean pine was detected for arsenic (6.7 times) and antimony (2.2 times). Same concentrations in foliage biomass of the two researched tree species for physiologically significant metals have been established for copper, nickel and zinc.

Determination of content of heavy metals in mine rock and foliage biomass – Crimean pine needles and Black locust leaves – tree species used for biological reclamation of mine dumps shows stabilisation of content of heavy metals in the substrate. According to the bioaccumulation coefficient, Crimean pine can be considered a hyperaccumulator of lead, which substantiates its use as a phytoremediation agent.

References

Alekseenko, V.A., Pashkevich, M.A. & Alekseenko A.V. (2017). Metallisation and environmental management of mine site soils. *Journal of Geochemical Exploration*, 174, 121–127. DOI: 10.1016/j.gexplo.2016.06.010.

Alexander, M. (2000). Aging, bioavailability, and overestimation of risk from environmental pollutants. *Environ. Sci. Technol.*, 34, 4259–4265. DOI: 10.1021/es001069.

Allen, H.E, Huang, C.P., Bailey, G.W. & Bowers A.R. (1995). *Metal speciation and contamination of soil*. Boca Raton, FL: Lewis Publishers.

Appenroth, K.J. (2010). Definition of "heavy metals" and their role in biological systems. In *Soil heavy metals. Soil biology*, 19 (pp. 19–29). Berlin, Heidelberg: Springer. DOI: 10.1007/978-3-642-02436-8_2.

Avessalomov, I.A. (1987). *Geochemical indicators in the study of landscapes (in Russian)*. Moscow: Publishing House of Moscow University.

Brown, P.H., Welch, R.M. & Madison J.T. (1990). Effect of nickel deficiency on soluble anion, amino acid and nitrogen levels in barley. *Plant Soil*, 125, 19–27.

Chodak, M. & Niklińska M. (2010). The effect of different tree species on the chemical andmicrobial properties of reclaimed mine soils. *Biol. Fertil. Soils*, 46(6), 555–566. DOI: 10.1007/s00374-010-0462-z.

Chudzińska, E., Celiński, K., Pawlaczyk, E. & Diatta J. (2016). Trace element contamination differentiates the natural population of Scots pine: evidence from DNA microsatellites and needle morphology. *Environ. Sci. Pollut. Res. Int.*, 23(21), 22151–22162. DOI: 10.1007/s11356-016-7472-9.

Dmuchowski, W. & Bytnerowicz A. (1995). Monitoring environmental pollution in Poland by chemical analysis of Scots pine (*Pinus sylvestris* L.) needles. *Environ. Pollut.*, 87, 87–104. DOI: 10.1016/S0269-7491(99)80012-8.

Eide, D.J. (2006). Zinc transporters and the cellular trafficking of zinc. *Biochim. Biophys. Acta, Molecular Cell Research*. 1763(7), 711–722. DOI: 10.1016/j.bbamcr.2006.03.005.

Fernández, S., Poschenrieder, C., Marcenò, C., Gallego, J.R., Jiménez-Gámez, D., Bueno, A. & Afif E. (2017). Phytoremediation capability of native plant species living on Pb-Zn and Hg-As mine wastes in the Cantabrian range, north of Spain. *Journal of Geochemical Exploration*. 174, 10–20. DOI: 10.1016/j.bbamcr.2006.03.005: 10.1016/j.gexplo.2016.05.015.

Grishko, V.M., Syschykov, D.V., Piskova, A.M., Danilchuk, O.V. & Mashtaler O.V. (2012). *Heavy metals: intake in soil, translocation in plants and environmental hazards (in Ukrainian)*. Donetsk.

Hüttl, R. (1998). Ecology of post strip-mine landscapes in Lusatia, Germany. *Environmental Science Pollution*, 1**,** 129–135. DOI: 10.1016/S1462-9011(98)00014-8.

Hüttl, R. & Weber E. (2001). Forest ecosystem development in post-mine landscapes: a case study of the Lusatian lignite district. *Naturwissenschaften*, 88, 322–329. DOI: 10.1007/s001140100241.

Itoh, Y., Miura, S. & Yoshinaga S. (2006). Atmospheric lead and cadmium deposition within forests in the Kanto district, Japan. *J. For. Res.*, 11(2), 137–142. DOI: 10.1007/s10310-005-0196-1.

Jarup, L. (2003). Hazards of heavy metal contamination. *Br. Med. Bull.,* 68, 167–182. DOI: 10.1093/bmb/ldg032.

Kaar, E. (2002). Coniferous trees on exhausted oil shale opencast mines. *Metsanduslikud Uurimused (Forestry Studies),* 36, 120–125.

Kabata-Pendias, A. (2011). *Trace elements in soil and plants*. Boca Raton: CRC Press. DOI: 10.1201/b10158.

Khokhotva, A.P. (2010). Adsorption of heavy metals by a sorbent based on pine bark. *Journal of Water Chemistry and Technology,* 32(6), 336–340. DOI: 10.3103/S1063455X10060044.

Kubatbekov, T.S., Aitmatov, M.B. & Ibraimakunov M. (2012). Antimony in natural technogenic conditions of the biosphere: water, soil, plants (in Russian). *Bulletin of the Russian University of Peoples' Friendship*, 4, 56–60.

Kuznetsova, T., Mandre, M., Klõseiko, J. & Pärn H. (2010). A comparison of the growth of Scots pine (*Pinus sylvestris* L.) in a reclaimed oil shale post-mine area and in a Calluna site in Estonia. *Environ. Monit. Assess.*, 166, 257–265. DOI: 10.1007/s10661-009-0999-1.

Lakyda, P.I. (2003). *Phytomass of Ukrainian forests (in Ukrainian)*. Ternopil: Sbruch.

Lin, Q., Chen, Y.X., He, Y.F. & Tian G.M. (2004). Root-induced changes of lead availability in the rhizosphere of *Oryza sativa* L. *Agric. Ecosyst. Environ.*, 104, 605–613. DOI: 10.1016/j.agee.2004.01.001.

Marko-Worłowska, M., Chrzan, A. & Łaciak T. (2011). Scots pine bark, topsoil and pedofauna as indicators of transport pollutions in terrestrial ecosystems. *J. Environ. Sci. Health,* 46, 138–148. DOI: 10.1080/10934529.2010.500896.

Marmiroli, M., Pietrini, F., Maestri, E., Zacchini, M., Marmiroli, N. & Massacci A. (2011). Growth, physiological and molecular traits in Salicaceae trees investigated for phytoremediation of heavy metals and organics. *Tree Physiol.*, 31, 1319–1334. DOI: 10.1093/treephys/tpr090.

Pietrzykowski, M. & Socha J. (2011). An estimation of Scots pine (*Pinus sylvestris* L.) ecosystem productivity on reclaimed post-mine sites in Poland (central Europe) using of allometric equations. *Ecological Engineering*, 37(2), 381–386. DOI: 10.1016/j.ecoleng.2010.10.006.

Pietrzykowski, M., Socha, J. & van Doorn N.S. (2014). Linking heavy metal bioavailability (Cd, Cu, Zn and Pb) in Scots pine needles to soil properties in reclaimed mine areas. *Sci. Total Environ.*, 470–471, 501–510. DOI: 10.1016/j.scitotenv.2013.10.008.

Pöykiö, R., Hietala, J. & Nurmesniemi H. (2010). Scots pine needles as bioindicators in determine the aerial distribution pattern of sulphur emissions around industrial plants. *World Academy of Science, Engineering and Technology*, 44, 116–119.

Prasad, M.N.V. & Hagemeyer J. (1999). *Heavy metal stress in plants. From molecules to ecosystems.* Berlin Heidelberg: Springer-Verlag. DOI: 10.1007/978-3-662-07745-0.

Risto, P., Perämäki, P. & Niemelä M. (2005). The use of Scots pine (*Pinus sylvestris* L.) bark as a bioindicator for environmental pollution monitoring along two industrial gradients in the Kemi-Tornio area, northern. *International Journal of Environmental Analytical Chemistry,* 85, 127–139. DOI: 10.1080/03067310412331330758.

Saarelaa, K.-E., Harjua, L., Rajandera, J., Lillb, J.-O., Heseliusb, S.-J., Lindroosd, A. & Mattsson K. (2005). Elemental analyses of pine bark and wood in an environmental study. *Sci. Total Environ.*, 343, 231–241. DOI: 10.1016/j.scitotenv.2004.09.043.

Shahid, M., Pourrut, B., Dumat, C., Nadeem, M., Aslam, M. & Pinelli E. (2014). Heavy-metal-induced reactive oxygen species: phytotoxicity and physicochemical changes in plants. *Rev. Environ. Contamin. Toxicol.,* 232, 1–44. DOI: 10.1007/978-3-319-06746-9_1.

Thapa, G., Sadhukhan, A., Panda, S.K. & Sahoo L. (2012). Molecular mechanistic model of plant heavy metal tolerance. *Biometals,* 25, 489–505. DOI: 10.1007/s10534-012-9541-y.

Verbruggen, N., Hermans, C. & Schat H. (2009). Molecular mechanisms of metal hyperaccumulation in plants. *New Phytol.*, 181(4), 759–776. DOI: 10.1111/j.1469-8137.2008.02748.x.

Wuana, R.A. & Okieimen F.E. (2011). Heavy metals in contaminated soils: a review of sources, chemistry, risks and best available strategies for remediation. *ISRN Ecology,* 2011, 20. DOI: 10.5402/2011/402647.

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SPATIOTEMPORAL DYNAMICS OF SOIL PENETRATION RESISTANCE OF RECULTIVATED SOIL

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Abstract

Zadorozhnaya G.: Spatiotemporal dynamics of soil penetration resistance of recultivated soil. Ekológia (Bratislava), Vol. 37, No. 1, p. 82–89, 2018.

This article examines changes in the spatial distribution of soil penetration resistance in ordinary chernozem (Calcic Chernozem) and in the recultivated soil in 2012 and 2014. The measurements were carried out in the field using an Eijkelkamp penetrometer on a regular grid. The depth of measurement was 50 cm, the interval was 5 cm. The indices of variation of soil penetration resistance in space and time have been determined. The degree of spatial dependence of soil penetration resistance has been determined layer by layer. The nature of temporal dynamics of soil penetration resistance of chernozem and technical soil has been described. A significant positive relationship of the structure of chernozem in the two years of the research has been shown. Significant correlations between the data of different years in the technical soil were found to be mostly negative.

Key words: soil penetration resistance, recultivation, geostatistics, spatial distribution of data.

Introduction

The specific role of soil in ecosystems gives a high degree of relevance to research of anthropogenic soil processes (Brady, Weil, 2002; Karpachevskij et al., 2007). The strongest anthropogenic impacts are suffered by the soil that is formed by the extraction of mineral deposits through open-cut mining (technical soils). They go through the stages of complete destruction of the soil body and creation of a new one (Yeterevska et al., 2008; Uzbek et al., 2010). Substrates are formed by waste disposal areas of the mining industry. Theoretically, such soils can be considered as models of young natural soil. From a practical standpoint, the study of such soils is necessary to solve the problems of their economic use (Ditsch, Collins, 2000). It is known that soil formation under the man's influence proceeds more actively than in adjacent conditionally natural ecosystems (Gerasimova et al., 2003; Medvedev, 2014). Cases of increase in speed of soil processes in conditions of contamination with foreign compounds, secondary salinization during irrigation, periodic deep ploughing have been described (Nikiforova, Solntseva, 1982; Solntseva, Rubilina, 1987; Gerasimova et al., 2003). Due to the youth of the reclaimed soils, a large number of particular processes occurring in them remain unclarified.

To assess the condition of the soil body, the soil penetration resistance index is applied (Bölenius et al., 2006; Medvedev, 2009; Alvarez et al., 2009). This emulates the living conditions of soil inhabitants (Lipiec, Tarkiewicz, 1990; Vanags et al., 2004; Godefroid, Koedam, 2004; Langmaack et al., 2002). At the same time, the variation in soil penetration resistance correlates with the change in most properties important for soil fertility (Vachel, Ehrlich, 1988; Grunwald et al., 2001; Vanags et al., 2004; Topp et al., 2003; Bajla, Minarik, 2003; Bölenius et al., 2006). The objective of this study is a comparative analysis of temporal changes in the spatial variation of soil penetration resistance of recultivated soil and natural soil.

Material and methods

The soil penetration resistance was studied in the field environment to a depth of up to 50 at 5 cm intervals. A hand penetrometer Eijkelkamp was used in the study (Young et al., 2000; Grunwald et al., 2001; Bengough et al., 2001; Cecilia et al., 2012; Zhukov, Zadorozhnaya, 2016). The average error of the measurement results of the device is ± 8%. Soil penetration resistance was measured using a cone with a cross-section of 2 cm² in each section of the test site.

Ordinary chernozem (Calcic Chernozem) was selected for study as natural soil (control). The test site is located in a steppe area adjacent to the south-eastern slope of the Kamenistaya Gully (southern outskirts of the city of Dnipro, Ukraine), 48°38'67"N, 35°09'05" E. The soil penetration resistance of the technical soil was studied at a recultivation site in the Nikopol manganese-ore basin (city of Pokrov, Ukraine), 47°38'55 "N, 34°08'33" E. Sod-lithogenic soil on red-brown clays was chosen as the substrate. The landfill was created 40 years ago. Currently, crops are not cultivated on the test sites, though cereals and beans grow there in feral condition.

The material was collected in June 2012 and 2014. The test sites are a regular grid of 7×15 m. The distance between the sampling points was 3 m. Accordingly, the dimensions of each test site were 18×42 m.

In statistical calculations, the methods of descriptive statistics were used. To determine the level of spatial dependence of soil penetration resistance parameters, geostatistical analysis of data was applied (Verones et al., 2006; Webster, Oliver, 2007; Diggle, Ribeiro, 2007; Calderon et al., 2008; Chien et al., 1997). The spatial dependence level (SDL) was calculated by the formula:

$$SDL = \frac{C_0}{C_0 + C_1} \times 100,$$

where C_0 is the nugget-effect, C_1 is the partial sill.

The indices for C_0, C_1 and the range (R) were obtained on the basis of simulations of variograms of spatial variability of soil penetration resistance (Legendre, Fortin, 1989; Grunwald et al., 2001; Webster, Oliver, 2007).

The degree of conjugation of the spatial distribution of the soil body indices in different years of the study was established by means of correlation analysis.

Results

Chernozem has less absolute soil penetration resistance factors than recultivated soil. The average values of chernozem soil penetration resistance did not exceed 2.67 ± 0.07 MPa (Table 1). The technical soil penetration resistance value was much higher. The maximum values in 2012, were 8.42 ± 0.33 MPa, and in 2014 were 6.71 ± 0.15 MPa. The reason for this is the direct dependence of soil penetration resistance on humidity (Umarova et al., 2013). Recultivated soil quickly loses moisture due to the low content of organic matter and the absence of water-resistant structure (Zhukov et al., 2013; Demidov et al., 2013). Soil penetration resistance data were more variable in the upper soil layers. The greatest values of the variation coefficient corresponded to the layer of 5–10 cm from the surface in all cases.

T a b l e 1. Parameters of spatial variation of soil penetration resistance.

Distance from the surface, cm	Ordinary chernozem				Recultivated soil			
	x ± SE, MPa	CV, %	SDL, %	R, m	x ± SE, MPa	CV, %	SDL, %	R, m
2012								
0–5	2.04 ± 0.07	32.71	27.8	6.91	3.26 ± 0.09	41.1	2.09	4.49
5–10	2.14 ± 0.08	39.75	17.21	6.32	4.57 ± 0.17	42.7	6.60	4.31
10–15	2.12 ± 0.08	38.02	28.39	4.53	5.58 ± 0.19	40.0	13.07	3.52
15–20	2.15 ± 0.07	31.04	30.4	4.20	6.31 ± 0.22	39.4	12.81	3.34
20-25	2.13 ± 0.06	28.57	26.87	4.38	6.96 ± 0.25	35.4	13.09	3.79
25-30	2.14 ± 0.06	28.12	25.67	5.07	7.39 ± 0.27	36.5	17.13	4.06
30–35	2.15 ± 0.06	27.49	28.16	5.22	7.79 ± 0.29	37.3	15.61	4.30
35–40	2.23 ± 0.06	26.53	32.22	4.76	8.24 ± 0.31	38.7	18.96	4.11
40-45	2.40 ± 0.06	26.35	34.73	4.92	8.42 ± 0.33	38.2	21.29	4.19
45–50	2.67 ± 0.07	25.25	32.72	4.90	8.06 ± 0.35	29.6	18.92	4.04
2014								
0–5	1.45 ± 0.04	26.91	22.27	9.83	2.05 ± 0.09	47.31	18.52	3.86
5–10	1.54 ± 0.05	33.09	20.60	9.06	3.89 ± 0.14	38.01	21.48	5.01
10–15	1.52 ± 0.05	32.21	17.30	10.14	4.90 ± 0.10	20.57	25.10	5.60
15–20	1.54 ± 0.03	22.92	31.01	6.72	5.44 ± 0.06	11.70	31.13	3.67
20-25	1.51 ± 0.03	20.63	31.00	7.44	5.64 ± 0.06	11.58	34.23	3.62
25-30	1.48 ± 0.03	22.45	30.22	6.51	5.93 ± 0.12	20.20	42.71	3.53
30–35	1.52 ± 0.04	24.07	32.17	5.42	5.95 ± 0.14	24.97	52.47	4.57
35–40	1.71 ± 0.04	24.14	27.68	7.05	6.11 ± 0.14	23.97	46.48	3.73
40-45	2.08 ± 0.05	24.74	26.09	6.82	6.46 ± 0.15	23.90	50.80	4.53
45–50	2.46 ± 0.06	24.28	25.56	6.34	6.71 ± 0.15	22.29	60.87	4.09

Notes: x – is an average value (MPa); SE – is the standard error; CV – is the coefficient of variation (%); SDL – is the spatial dependence level (%); R – is the radius of influence (m).

The lower the SDL value, the greater the spatial dependence of these data. A higher level of spatial dependence is described in the data for the upper layers of the soils studied. In the upper layers, the bulk of the roots of plants and soil animals is concentrated. Their vital activity has a structuring effect on the soil and increases the level of spatial dependence.

The semivariogram range (R) shows the distance at which the geostatistic model first flattens out or in our case the distance over which the mutual spatial influence of the soil masses spreads. In general, this indicator is higher in chernozem than in technical soil. During the two years of study, the radius of influence decreased with depth, while in recultivated soil it increased.

Based on the geostatistical data, maps of the layered distribution of soil penetration resistance in the soils studied were designed (Fig. 1).

Darker areas indicate increased soil penetration resistance. The transition to the areas of lesser soil penetration resistance has a gradient character. With increased depth the pattern

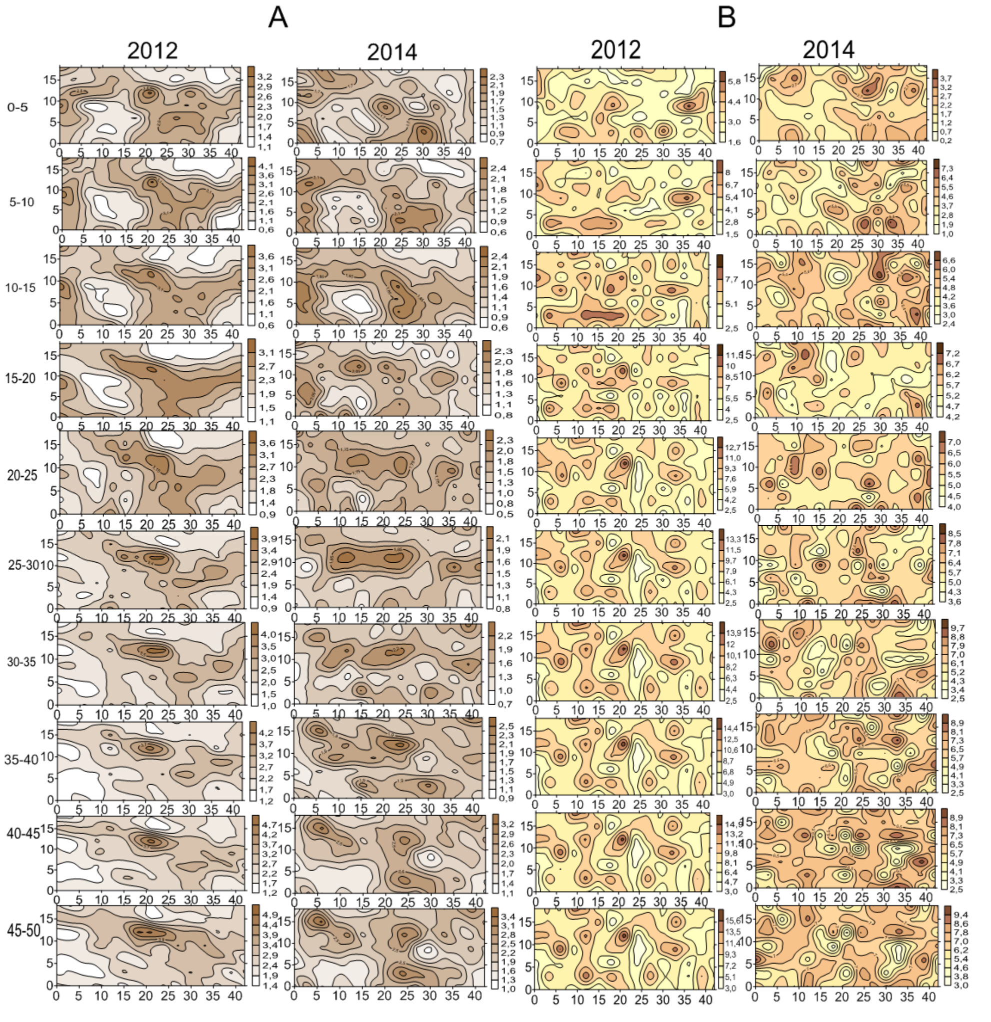

Fig. 1. Maps of spatial distribution of soil penetration resistance indicators in chernozem (A) and in technical soil (B) by layers in 2012 and 2014.
Notes: The abscissa is the length of the polygon, m, the ordinate is the width of the polygon, m; 0–5 ... 95–100 cm is the depth of soil from the surface.

changes. One can observe a three-dimensional separation of the soil material in terms of increased soil penetration resistance. The maps of the same sites changed considerably over two years. Less solid areas appeared in place of harder ones, and vice versa.

In order to determine the quantitative measure of the soil structure relationship in different years, a correlation analysis was performed (Tables 2, 3).

T a b l e 2. Coefficients of Pearson correlation of soil penetration resistance in ordinary chernozem in 2012 and 2014.

Years		**2014**									
		0–5	5-10	10-15	15–20	20-25	25–30	30–35	35-40	40-45	45-50
2012	0–5	**0.45**	**0.61**	**0.59**	**0.47**	**0.26**	**0.23**	0.08	0.06	0.13	0.24
	5–10	**0.44**	**0.67**	**0.71**	**0.58**	**0.36**	**0.30**	0.07	0.03	0.11	**0.26**
	10–15	0.43	0.64	0.71	0.60	0.38	**0.28**	0.03	0.02	0.06	0.19
	15–20	**0.32**	**0.54**	**0.63**	**0.54**	**0.38**	**0.25**	-0.01	0.00	0.03	0.19
	20-25	0.21	**0.38**	**0.46**	**0.43**	**0.38**	**0.32**	0.07	0.16	0.14	0.26
	25-30	0.16	**0.26**	**0.31**	**0.37**	**0.42**	**0.42**	**0.23**	**0.26**	0.19	0.26
	30–35	0.12	**0.20**	**0.22**	0.19	**0.30**	**0.40**	**0.36**	**0.34**	**0.23**	**0.26**
	35–40	0.10	0.14	0.11	0.11	**0.31**	0.45	0.46	0.43	0.28	0.31
	40-45	0.10	0.13	0.06	0.04	**0.28**	**0.48**	**0.51**	**0.50**	**0.33**	**0.36**
	45–50	0.14	**0.26**	0.18	0.18	**0.33**	**0.48**	**0.44**	**0.37**	**0.25**	**0.31**

Notes: 0–5, ..., 45–50 is the distance from the surface (cm); statistically significant correlations ($p < 0.05$) are shown in bold.

T a b l e 3. Pearson correlation coefficients of soil penetration resistance of recultivated soil in 2012 and 2014.

Years		**2014**									
		0–5	5-10	10-15	15–20	20-25	25–30	30–35	35-40	40-45	45-50
2012	0–5	-0.12	-0.05	-0.06	**-0.20**	0.03	0.07	**-0.19**	-0.17	**-0.19**	**-0.23**
	5–10	**-0.28**	**-0.21**	-0.19	-0.06	-0.05	-0.14	**-0.27**	**-0.21**	**-0.30**	**-0.30**
	10–15	**-0.26**	**-0.28**	**-0.28**	0.03	-0.11	**-0.23**	**-0.32**	**-0.34**	**-0.30**	**-0.27**
	15–20	-0.17	**-0.25**	-0.14	**0.20**	-0.02	-0.08	-0.17	**-0.21**	**-0.21**	**-0.20**
	20-25	-0.15	**-0.28**	-0.18	0.17	-0.08	-0.13	-0.17	-0.17	-0.15	-0.15
	25-30	-0.15	**-0.24**	-0.16	**0.20**	-0.07	-0.10	-0.14	0.15	-0.08	-0.12
	30–35	-0.13	**-0.20**	-0.14	**0.20**	-0.02	-0.07	-0.16	-0.14	-0.08	-0.11
	35–40	0.15	**-0.21**	0.15	**0.20**	-0.04	-0.09	-0.16	0.15	-0.10	-0.13
	40-45	-0.14	**-0.21**	-0.16	**0.20**	-0.04	-0.11	-0.16	0.15	-0.09	-0.12
	45–50	-0.16	**-0.22**	-0.16	**0.20**	-0.04	-0.12	-0.17	0.15	-0.09	-0.12

Notes: see Table 2.

The spatial distribution of chernozem soil penetration resistance data in different years is strongly correlated. The structure of the upper soil layers (0–5, ..., 15–20 cm) in 2014 reliably depends on that of 2012. A similar pattern is observed in the data of the lower layers (30–35, ..., 45–50 cm) of the studied soil mass. Layers of 20–25 and 25–30 cm carry data about the past structure of the entire soil profile. The structure of the layer of 45–50 cm depends also on the structure of the surface layers (0–5, 5–10 cm), which was observed 2 years previously.

The spatial distribution of soil penetration resistance of the technical soil also has much in common with the data of the preceding years. Over the two years of study, reliable correlations were observed in almost all the studied soil layers. The structure of the upper layers (0–5, ..., 15–20 cm) exerts the strongest influence on the lower layers of the soil. The most dependent layers are those of 5–10 and 15–20 cm. It should be noted that the correlation is mostly negative. Reliable positive coefficients are observed only in the data of the layer of 15–20 cm in 2014. A reliable negative correlation can be traced in all soil layers except the 20–25 cm layer.

Discussion

The results of the study show that the soil has a memory of the past structure (Targulian, Goryachkin, 2008). A high level of dependence of soil penetration resistance data distribution was revealed over the two years of research. The discovered relations in the technical soil were weaker than those of the chernozem. Reliable correlations between the data of the technical soil for the different years were mostly negative. Negative dependencies also indicate the existence of a relationship. However, they show that in the areas of increased soil penetration resistance it decreased over two years and vice versa. Most likely, the driver of such processes is vegetation. From the point of view of solid phase structural organization, anthropogenic soil is considered as a simpler system with limited buffering (Anand et al., 2002; Gerasimova et al., 2003, Serafim et al., 2008). The latter affects the stability and structure of plant aggregations. Intraspecific and interspecific relationships of plants are determined by trophic dependencies and environmental transformation (Chesson, 2000). In chernozem these relationships were formed long ago. Recultivated soils are very young. Within these soils, transformations of vegetation on them are very clearly expressed from year to year. There is a rapid change in species composition and quantitative relationships. In this context, important work has been conducted on plant communities on technical soil (Andrusevich, Stirz, 2014). This research found an expansion in a range of species with different environmental preferences in comparison with the natural community. It indicates the variability of ecological environmental regimes for the vegetation of anthropogenic soil. The share of young plants on technical soil is over 1.5 times higher than it is for the natural community. This indicates the state of dynamic transformation of vegetation on technical soils. In turn, the structure of the plant community influences the structure of the soil (Zhukov, Zadorozhnaya, 2015). Depending on the characteristics of the root systems, this effect is significant (Montagu et al., 2001; Vanags et al., 2004; Godefroid, Koedam, 2004). Rapid changes in the configuration of the plant community lead to dynamic restructuring of the soil. The presence of a large number of negative correlations between the structure of technical soil during the different years of the study confirms our assumptions.

Conclusion

The chernozem studied had lower indices of absolute soil penetration resistance than the recultivated soil. The maximum average value of soil penetration resistance of the chernozem was 2.67 ± 0.07 MPa, while for technical soil it was 8.42 ± 0.33 MPa.

The values of black and technical soil penetration resistance were more variable in the upper layers of the soil. The greatest values of the coefficient of variation correspond to the layer of 5–10 cm from the surface in all cases (33.09–42.7%).

A higher level of spatial dependence is described in the data for the upper layers of the soils studied.

The radius of influence in the chernozem decreased with depth, while it increased in the reclaimed soil. Absolute values of the radius of influence in the chernozem were higher than those of technical soil.

A high level of dependence of soil penetration resistance data distribution was revealed over the two years researched. The discovered relations in the technical soil were weaker than those of the chernozem. Reliable correlations between the data of the different years in the technical soil were mostly negative.

References

Alvarez, C.R., Taboada, M.A., Gutierrez Boem, F.H., Bono, A., Fernández, P. L. & Prystupa P. (2009). Topsoil properties as affected by tillage systems in the Rolling Pampa Region of Argentina. *Soil Sci. Soc. Am. J.*, 73, 1242–1250. DOI: 10.2136/sssaj2008.0246.

Anand, M., Tucker, B.C. & Desrochers R. (2002). Ecological monitoring of terrestrial ecosystem recovery from man-made perturbation: assessing community complexity. In *Proceedings of the 10th International Conference on Modelling, Monitoring and Management of Air Pollution* (pp. 341–350). July 1–3, 2002, Segovia, Spain. Southampton: WITPress.

Andrusevich, E.V. & Shtirts Yu.A. (2014). Ecologiocal diversity of vegetation on lithogenic soil in the reclamation land of the Nicopol manganese ore basin. *Industrial Botany*, 14, 115–127.

Bajla, B.C. & Minarik J. (2003). Navrh metody na meranie okamzitej vlhkosti pody u hrot epenetrometra. *Acta Technologica Agriculturae,* 4, 93–96.

Bengough, A.G., Campbell, D.J. & O'Sullivan M.F. (2001). Penetrometer techniques in relation to soil compaction and root growth. In K.A. Smith & C.E. Mullins (Eds.), *Soil and environmental analysis. Physical methods* (pp. 377–403). New York: Marcel Dekker.

Bölenius, E., Rogstrand, G., Arvidsson, J., Thylén, L. & Stenberg B. (2006). On-the-go measurements of soil penetration resistance on a Swedish Eutric Cambisol. In *Proceedings of International Soil Tillage Research Organization 17th Triennal Conference* (pp. 867–870). Kiel.

Brady, N.C. & Weil R.R. (2002). *The nature and properties of soils*. Upper Saddle River: Prentice Hall.

Calderon, V.C.A., Martinez, M.L.J. & Giraldo Henao R. (2008). Spatial variability of soil properties and yield relationship in a mango crop (*Mangifera indica* L.). *Revista Brasileira de Fruticultura* 30(4), 1146–1151. DOI: 10.1590/S0100-29452008000400049.

Cecilia, M., Jesus, H.C. & Cortes C.A. (2012). Soil penetration resistance analysis by multivariate and geostatistical methods. *Engenharia Agricola*, 32(1), 91–101. DOI: 10.1590/S0100-69162012000100010.

Chesson, P. (2000). Mechanisms of maintenance of species diversity. *Annu. Rev. Ecol. Syst.*, 31, 343–366.

Chien, Y.J., Lee, D.Y., Guo, H.Y. & Houng K.H. (1997). Geostatistical analysis of soil properties of mid – west Taiwan soils. *Soil Sci.*, 162, 291–298.

Demidov, A.A., Kobets, A.S., Gritsan, Y.I. & Zukov A.V. (2013). *Spatial agroecology and land reclamation: monograph*. Dnepropetrovsk: Publishing house 'Svidler AL'.

Diggle, P.J. & Ribeiro J.P. Jr. (2007). *Model based geostatistics.* Sao Paulo: Springer.

Ditsch, D.C. & Collins M. (2000). Reclamation consideration for pasture and hay lands receiving sixty–six centimeters or more precipitation annually. *Reclamation of drastically disturbed lands. Agronomy Monograph* (pp. 24–273). Madison: ASA, CSSA, SSSA.

Gerasimova, M.I., Stroganova, M.N., Mozharova, M.V. & Prokofieva T.V. (2003). *Anthropogenic soils (genesis, geography, recultivation) (in Russian)*. Smolensk: Oykumena.

Godefroid, S. & Koedam N. (2003). How important are large vs. small forest remnants for the conservation of the woodland flora in an urban context? *Glob. Ecol. Biogeogr.*, 12, 287–298. DOI: 10.1046/j.1466-822X.2003.00035.x.

Grunwald, S., Mc Sweeney, K., Rooney, D.J. & Lowery B. (2001). Soil layer models created with profile cone penetrometer data. *Geoderma*, 103(1–2), 181–201. DOI: 10.1016/S0016-7061(01)00076-3.

Karpachevskij, L.O., Zubkova, T.A., Tashninova, L.N. & Rudenko R.N. (2007). Soil cover and forest biogeoceonosis-

parcelar structure (in Russian). *Russian Forest Sciences*, 6, 107–113.
Langmaack, M., Schrader, S., Rapp-Bernhardt, U. & Kotzke K. (2002). Soil structure rehabilitation of arable soil degraded by compaction. *Geoderma*, 105, 141–152. DOI: 10.1016/S0016-7061(01)00097-0.
Legendre, P. & Fortin M.J. (1989). Spatial pattern and ecological analysis. *Vegetatio,* 80, 107–138. DOI: 10.1007/BF00048036.
Lipiec, J. & Tarkiewicz S. (1990, 1991). The influence of water content and bulk density on penetration resistance of two soils. *Zeszyty Problemowe Postępów Nauk Rolniczych*, 338, 99–105.
Medvedev, V.V. (2009). Soil penetration resistance and penetrographs in studies of tillage technologies (in Russian). *Eurasian Soil Science*, 42(3), 299–309. DOI: 10.1134/S1064229309030077.
Medvedev, V.V. (2014). The contents and laws of soil anthropogenous evolution (in Russian). *Gruntoznavstvo (Soil Science)*, 15(1–2), 17–30. DOI: 10.15421/041402.
Montagu, K.D., Conroy, J.P. & Atwell B.J. (2001). The position of localized soil compaction determines root and subsequent shoot growth responses. *J. Exp. Bot.*, 52, 2127–2133. DOI: 10.1093/jexbot/52.364.2127.
Nikiforova, E.M. & Solntseva N.P (1982). Geochemistry of technogenic flow and aureoles of pollution in coal mining areas (using the Kizelovsky basin as an example) (in Russian). In *Geochemistry of landscapes and geography of soils* (pp. 100–128). Moscow: MSU.
Serafim, M.E., Vitorino, A.C.T., Peixoto, P.P.P., Souza, C.M.A. & Carvalho D.F. (2008). Intervalo hidrico otimo em um latossolo vermelho distroferrico sob diferentes sistemas de producao. *Engenharia Agricola*, 28(4), 654–665. DOI: 10.1590/S0100-69162008000400005.
Solntseva, N.P. & Rubilina N.E. (1987). Morphology of soils transformed in coal mining. *Eurasian Soil Science*, 2, 104–108.
Targulian, V.O. & Goryachkin S.V. (Eds.) (2008). *Soil memory: Soil as a memory of biosphere-geosphere-anthroposphere interactions*. Moscov: LKI Publishers.
Topp, G.C., Lapen, D.R., Edwards, M.J. & Young G.D. (2003). Laboratory calibration, in–field validation and use of a soil penetrometer measuring cone resistance and water content. *Vadose Zone Journal,* 2, 633–641. DOI: 10.2136/vzj2003.6330.
Uzbek, I.Kh., Volokh, P.V, Dyrda, V.I. & Demidov A.A. (2010). *Reclamation of disturbed lands as a sustainable development of complex techno-ecosystems: monograph (in Russian).* Dnipropetrovsk: Porogi.
Umarova, A.B., Shein, E.V., Milanovskiy, E.Yu. & Dembovetskiy A.V. (2013). Specific water regime in technogenic soils: Preferential water flow formation. *Procedia Environmental Sciences,* 19, 558–563. DOI:10.1016/j.proenv.2013.06.063
Vachel, J. & Ehrlich P. (1988). Vyuzifi penetrometricki metody mereni pevnosti zemin v prbzkumech pro odvodneni. *Vedpr. Vu zurod. Zemedel.,* 5, 131–140.
Vanags, C., Minasny, B. & McBratney A.B. (2004). The dynamic penetrometer for assessment of soil mechanical resistance. In *Supersoil 2004: Proceeding of the 3th Australian New Zealand Conference. University of Sydney, Australia.* URL: http://www.regional.org.au/au/asssi/supersoil2004/s14/poster/1565_vanagsc.htm.
Verones J.V., Carvalho, M.P., Dafonte, J., Freddi, O.S., Vidal Vazquez, E. & Ingaramo O.E. (2006). Spatial variability of soil water content and mechanical resistance of Brazilian ferralsol. *Soil Tillage Res.*, 85(1–2), 166–177. DOI:10.1016/j.still.2005.01.018
Webster, R. & Oliver M.A. (2007). *Geostatistics for environmental scientist.* Cornwall: John Wiley & Sons.
Yeterevska, L.V., Momot, G.F. & Lehtsier L.V. (2008). Recultivated soils establishment, diagnostic, eststimation. *Gruntoznavstvo (Soil Science)*, 9(3–4), 147–150.
Young, G.D., Adams, B.A. & Topp G.C. (2000). A portable data collection system for simultaneous cone penetrometer force and volumetric soil water content measurements. *Can. J. Soil Sci.*, 80, 23–31. DOI:10.4141/S99-025.
Zhukov, A.V., Zadorozhnaya, G.A. & Lyadskaya I.V. (2013). Aggregate structure of industrial soils of the Nikopol manganese ore basin. *Biological Bulletin of Bogdan Chmelnitskiy Melitopol State Pedagogical University,* 3(3), 274–286.
Zhukov, A.V. & Zadorozhnaya G.A. (2015). Ecomorphic organisation of the soil body: geostatistical approach (in Ukrainian). *Studia Biologica*, 9(3–4), 119–128.
Zhukov, A. & Zadorozhnaya G. (2016). Spatial heterogeneity of mechanical impedance of atypical chernozem: the ecological approach. *Ekológia (Bratislava),* 35(3), 263–278. DOI: 10.1515/eko-2016-0021.

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THE USE OF MULTI-CRITERIA ANALYSIS FOR IDENTIFYING AREAS SENSITIVE TO LAND DEGRADATION AND WATER RETENTION

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Abstract

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The article presents a method of selecting critical areas (4th river basin) in terms of landscape degradation, with an emphasis on water retention, from a relatively larger unit (3rd river basin). For this purpose, indicators that point directly or indirectly to soil and landscape degradation or water retention were selected with regard to the scale of processing. The indicators were processed in a multi-criteria context using principal component analysis, which, based on the spatial layout pattern of the indicators, assigns weights of importance. These weights were then subsequently used to calculate the aggregation index, which indirectly indicates the sensitivity of the area to degradation and, in particular, water retention. Two catchment areas of the 3rd order – Čížina and Kyjovka – with different soil, climatic and economic conditions were selected for the study. Among the indicators of water retention in the landscape, our analysis included the share of agricultural land in the total area, the share of arable land, the average size of the field block, soil degradation according to the degradation model, runoff curve number, potential water erosion and surface drainage. The resulting procedure can be used to evaluate smaller areas. For a more detailed solution, a number of other methods and indicators could be used, which are also outlined in the article.

Key words: sensitivity analysis, landscape, degradation, retention, modelling.

Introduction

All human life depends on the landscape, with all its parts including soil and water. If we look at the statistics, more than half of the arable land in the world is moderately or heavily degraded and the damage is caused not only by agricultural production loss and diminished livelihoods but also by the lost value of ecosystem services previously provided, including water filtration, erosion prevention, the nutrition cycle and the provision of clean air (ELD Initiative, 2015).

The resulting land degradation (LD) is a global process and the result of various factors, including climatic variations and human activities, and it progressively leads to a reduction in soil fertility, which is a phenomenon commonly regarded as soil degradation (SD) (Jie et al., 2002; Fullen, 2003). Land degradation and desertification are caused by natural and anthropogenic processes (Gisladottir, Stocking, 2005; Johnson, Lewis, 2007; Imeson, 2012) and lead to a reduction in land productivity with ecological and socio-economic consequences. The role of anthropogenic pressures is assumed (Bajocco et al., 2011).

The question is how to assess the degradation of land. In a number of studies, the use of visual observation, field measurements, social enquiries, environmental indicators derived from statistical sources, remote sensing, and mathematical models has been proposed (Basso et al., 2000; D'Angelo et al., 2000; Bathurst et al., 2003; Gad, Lotfy, 2008; Simeonakis et al., 2007; Costantini et al., 2009; Santini et al., 2010; Salvati et al., 2013, 2016).

A number of methodological approaches use indicators, where their selection is very important and ensure the most effective use of available data (Kosmas et al., 2003; Rubio, Recatala, 2006). An example of these approaches can be the ESA (e.g., Rubio, Bochet, 1998; Simeonakis et al., 2007; Thornes, 2004).

The paper deals with water retention in the landscape. This is mainly influenced by changes in the landscape and affects the runoff process and water storage capacity, which are consequently related to other parameters such as field capacity (FC) and saturated hydraulic conductivity (Ks) (Marshall et al., 2014). Surface runoff closely relates to landscape degradation (Kosmas et al., 2000) and can be determined using the runoff curve number (CN) model (Hawkins et al., 2009). Due to its simplicity, this model has been used to identify the direct surface runoff in agricultural basins (Mishra, Singh, 2006). CN relates to the water retention potential of soil (S) and the curve number model considers many factors including changes in land-use, soil type, land management, treatment, antecedent soil moisture and surface condition (Michel et al., 2005) and is involved in many complex and water retention simulation models (e.g., Soulis, Dercas, 2007; Singh et al., 2008). According to Mantey and Tagoe (2013), the main parameters for CN model are hydrological soil groups (HSGs), land use and the digital elevation model.

Runoff processes are also associated with the most serious soil degradation factor, which is water erosion. A wide range of approaches can be used to model it, for example, the spatially explicit erosion model PESERA (Kirkby et al., 2004) that takes into account climate, soil, land use and relief.

There are a number of methodological approaches to the study of degradation effects. They are, however, mostly assessed individually without any link to other influences. Our task was to find suitable indicators, sensitive to water retention and soil degradation threat, and process them, if possible, in a multi-criteria context in a way that could classify the analysed areas according to their sensitivity to water retention. In the most sensitive areas, the remedial measures would be preferentially applied.

Material and methods

The areas of interest for our study are the catchment areas of Čížina and Kyjovka, as shown in Fig. 1. The source of the river Čížina lies southwest of Horní Benešov (49.9528425N, 17.5540864E) at an altitude of 630 m and flows into the river Opava from the right near Brumovice at 280 m above sea level. The catchment area is 102.7 km^2, the flow length is 20.8 km^2 and the average discharge at its confluence with the Opava is 0.45 m^3s^{-1}. Geologically, most

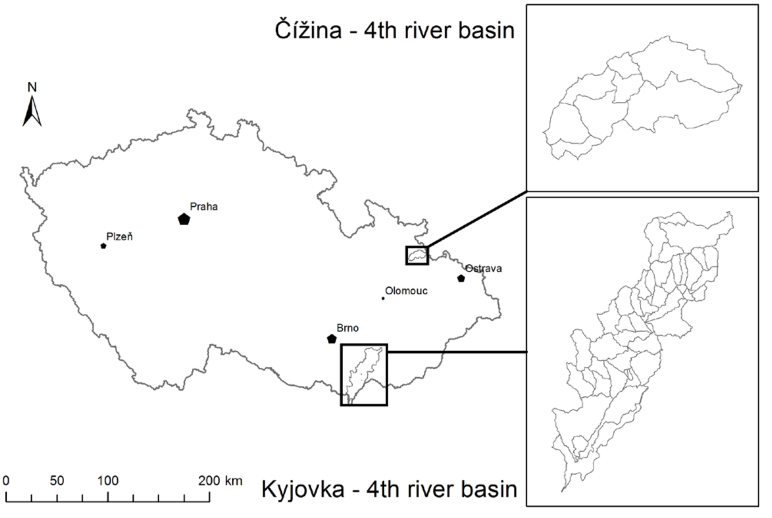

Fig. 1. Čížina and Kyjovka basins – our areas of interest.

of the catchment area consists of Culmian greywackes, sandstones, conglomerates and slates, with loess and glacial deposits in the lower part of the river.

Climatically, it is within the climatic regions of moderately warm MW9 and MW4 (centre), with the upper part of the catchment area in the moderately warm MW2 region. Opava meteorological station registers an average annual temperature of 8.0 °C and an average rainfall of 640 mm/yr. The most common soil type in the catchment area is Cambisol; the agricultural land occupies about 57% of the area, of which 66% is arable land and 33% permanent grassland. Forest accounts for 38% of the area.

The Kyjovka River is a left-hand tributary of the Dyje River. Its source is located in the Chřiby Mountains, on the southern slope of Vlčák hill (561 m above sea level) at an altitude of 512 m above sea level (49.1612092N, 17.2834028E), near the village of Stará Huť. Together with its tributaries, it drains a water catchment area of 665.8 km^2. The average flow of the river at its mouth is about 1.09 $m^3 s^{-1}$. The length of the river is 86.7 km. Geologically, the Kyjovka flows through two major units of the Western Carpathians, the Carpathian flysch belt and the Vienna Basin.

The catchment area falls into three climatic regions – very warm (VW), warm (W3) and moderately warm (MW2) with an average total rainfall of 500 to 650 mm. The most common soil type is chernozem, agricultural land occupies about 60% of the area, of which 83% is arable land, 7% permanent grassland and 8% vineyards. Forest represents about 29% of the area.

The catchment area of the Čížina incorporates a total of 9 further sub-basins of the fourth order. The Kyjovka basin incorporates another 48 sub-basins of the fourth order. Two water catchments with different soils (chernozems vs. Cambisols), climatic conditions (very warm, dry to warm, slightly humid, slightly warm, slightly humid to slightly cold, wet region) and economic conditions have been deliberately chosen for the study.

As indicators of water retention in the landscape, the following factors were included: the share of agricultural land in the total area, the percentage of arable land, the average size of the field block, soil degradation, CN curve, potential water erosion and surface drainage. The choice of this input data was influenced by the fact that this data is freely available or can be calculated based on the available free data.

Mainly vector geo-data was used with the exception of potential water erosion. In the case of Total Degradation factor from the Degradation Model of the Palacky University in Olomouc, the data had to be generalized to individual sub-basins because the original model only contains data for cadastres. Drainage values were obtained from the information system of the Czech Office of Surveying and Cadastre. The Land-parcel identification system (LPIS) was used to calculate potential water erosion, CN curves, and farmland size. Another basis was the Estimated pedologic ecological unit (EPEU) data set freely provided by the State Land Office. The altitude model was derived from the 4th Generation Digital Surface Model of the Czech Republic with a pixel resolution of 5x5 m; the calculations were performed in statistical software R and the maps were processed in ESRI – ArcGIS 10.2.2 GIS software.

The actual work deals with the methodology of the selection of interest areas in terms of water retention in the landscape. After studying different methodological approaches, we chose a model designed by Salvati et al. (2011), the output of which is the value of the aggregation risk index for each sub-basin, in a normalized range from 0 to 1, where 1 is the most sensitive area. The aggregation index was obtained on the basis of the multidimensional statistical analysis, which reduces the complexity of the input data bases, removes the interdependencies between the variables and each assigns a weighting parameter in terms of the importance of the observed phenomenon.

A fourth-order river basin was chosen as the basic mapping unit.

Our processing involved several steps:

1. Normalization of input data.
2. Principal component analysis (PCA) analysis of the normalized matrix.
3. Calculation of the synthetic vulnerability index of the catchment area.

Values of input parameters for each of 9 (Čížina) and 48 (Kyjovka) sub-basins respectively were obtained and modified as follows:

If the influence of the individual variable is negative with increasing value of the variable (i.e., it shows the need for solution), the variable is normalized to the range 0–1 according to the formula:

$$x_i' = \frac{x_{i,j} - x_{i,\min}}{x_{i,\max} - x_{i,\min}}.$$

Otherwise, according to the formula:

$$x_i' = 1 - \frac{x_{i,j} - x_{i,\min}}{x_{i,\max} - x_{i,\min}}, \text{ where}$$

$x_{i,j}$ represents the value of the i-th monitored variable in the j-th water sub-basin, $x_{i,\max}$, or $x_{i,\min}$ symbolizes the maximum, respectively the minimum value of the monitored variable across the whole water sub-basin.

The dependencies of the individual variables are shown in Table 1 using emoticons, where :-) means a positive effect of the variable and :-(a negative effect.

PCA analysis was applied to the resulting matrix of normalized variables. The weights of individual factors were determined by multiplying the contribution of each variable (V_i) to the most important variables (explaining 90% variability) by the proportion of their variance (C_k). The sum of these products for all the most important variables represents the individual weight (w_i) attributed to each indicator as the formula expresses:

$$w_i = \sum_{k=1}^{m} (V_i C_k)$$

Subsequently, the relative weights (W_i) were calculated by the ratio of the absolute weights to the sum of the weights of all the indicators.

$$W_i = \frac{w_i}{\sum_{i=1}^{6} w_i}$$

Each of these weights describes the extent to which a single factor contributes to the overall vulnerability of the territory.

The resulting IC sensitivity index is calculated based on the linear combination of normalized factors and their calculated significance weights.

$$IC = W_{ag} * Ag' + W_{ar} * Ar' + W_{fb} * Fb' + W_{cn} * CN' + W_{ero} * Ero' + W_{dm} * DM' + W_{sdr} * Sdr' \quad ,$$

where individual abbreviations represent normalized values:
Ag – agricultural land, Ar – arable land, Fb – average size of field block, CN – average value of CN curve, Ero – potential water erosion, DM – Total Degradation according to UPOL model and Sdr – surface drainage in individual sub-basins.

From the nature of the method, IC results are within the range of (0.1), where 1 indicates the highest sensitivity to degradation hazard (retention).

Results

The range of input parameters for both catchment areas is shown in Table 1. PCA analysis was used to evaluate the sensitivity of sub-basins to water retention in the landscape, which evaluates the spatial significance of the individual components and quantifies it in similarly calculated weights of the individual factors.

In the catchment area of Čížina, the percentage of arable land (W_{Al} = 18.1%) was the most important parameter, the second most important factor was surface drainage (W_{Sdr} = 17.1%).

T a b l e 1. Basic descriptive statistics of input variables included in the calculation.

		DM [0.1]	Ero [t/ha/yr.]	Ar [-]	Fb [ha]	Ag [-]	CN [0.100]	Sdr [-]	Area [ha]
Kyjovka	Min	0.30	0.1	0.01	1.8	0.04	46.7	0.000	37.8
47 sub-basins	Max	0.76	37.0	0.79	27.3	0.85	87.8	0.729	6449.7
	Avg	0.51	16.1	0.48	8.3	0.54	71.0	0.094	1304.2
Čížina	Min	0.48	1.7	0.04	1.2	0.27	67.8	0.000	15.2
9 sub-basins	Max	0.66	15.6	0.59	14.0	0.67	80.5	0.262	3130.4
	Avg	0.60	8.11	0.25	7.94	0.48	71.41	0.09	1085.92

Notes: Ag – agricultural land, Ar – arable land, Fb – average size of field block, CN – average value of CN curve, Ero – potential water erosion, DM – Total Degradation according to UPOL model and Sdr – surface drainage in individual sub-basins.

T a b l e 2. Results of PCA analysis for individual variables.

	Dependency	Code	Čížina – weight	Kyjovka – weight
Percentage of agricultural land	:-)	Ag	15.1%	17.8%
Percentage of arable land	:-)	Ar	18.5%	17.4%
Average size of fields in ha (arable)	:-)	Fb	10.8%	9.4%
Soil degradation (UP model)	:-)	DM	13.5%	11.6%
CN curve	:-)	CN	14.1%	10.5%
Potential water erosion in t/ha/yr	:-)	Ero	10.9%	23.7%
Melioration data - surface drainage	:-)	Sdr	17.1%	9.6%

In the Kyjovka basin, the most important factor is clearly potential water erosion (W_{Ero} = 23.7%). Detailed results for all the factors are summarized in Table 2.

In the catchment area of Čížina, the worst sub-basin is 2-02-01-074 with IC = 0.87 and the values of the most important factors Ar = 59% (worst among the sub-basins) and Sdr = 23% (2nd worst among the sub-basins). In the catchment area of Kyjovka, the worst sub-basin is 4-17-01-076 with an IC = 0.73 and the most significant ERO = 32.3 t/ha/yr (worst among sub-basins) and Ag = 73% (12th worst among sub-basins). The graphical representation of the resulting IC values for each river basin is shown in Figures 2 and 3.

Discussion and conclusion

The aim of our research was to propose suitable and easily accessible indicators, which determine or affect degradation of the agricultural landscape, with an emphasis on water retention. On the basis of these indicators, it should be possible to identify the most problematic areas at the level of 4th order river basin. Another objective was to choose a suitable method of processing which would reflect the importance of indicators in terms of their spatial distribution pattern. The results of processing are weights showing the importance of indicators for the studied research topic.

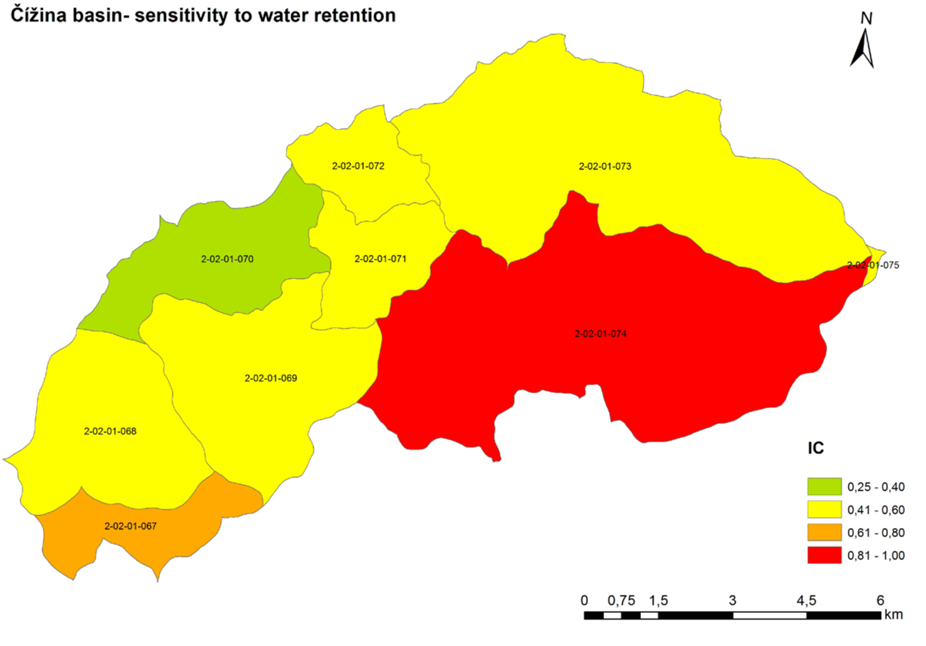

Fig. 2. Sensitivity of the 4th order of the Čížina River Sub-basins to land degradation with the emphasis on water retention.

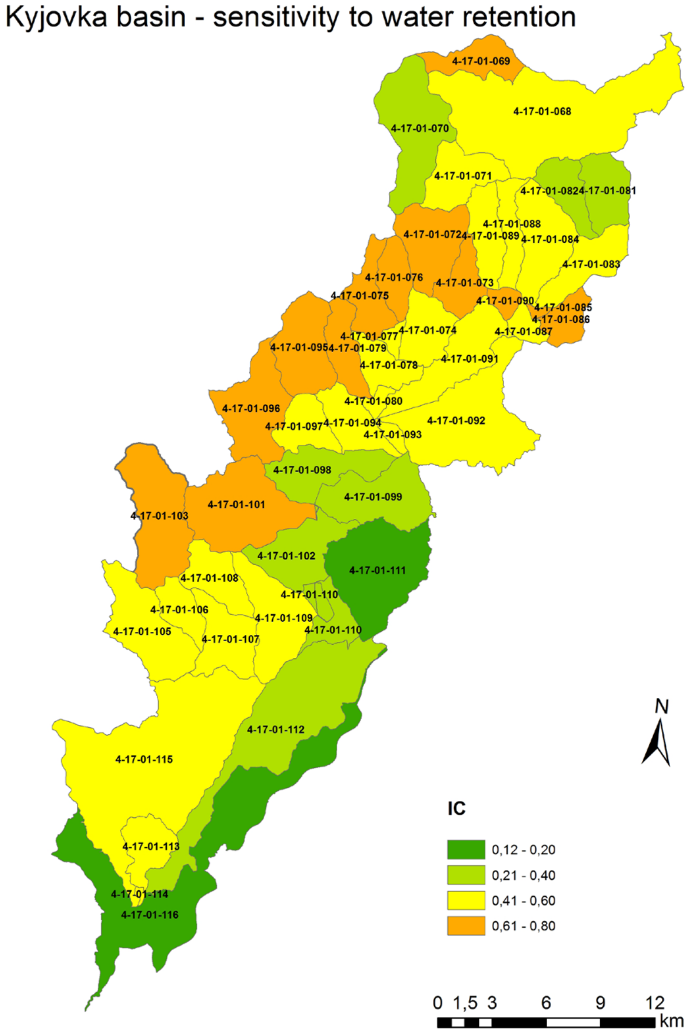

Fig. 3. Sensitivity of the 4th order of the Kyjovka Sub-basins to land degradation with the emphasis on water retention.

There are a number of methods of selecting indicators that affect water retention in the landscape at the level of individual sites. Geroy et al. (2011) studied retention with selected soil characteristics, which they investigated in relation to morphometric parameters of the landscape, especially aspect. Krnáčová et al. (2016) developed an algorithm that can indirectly derive the hydro-limits of soils from the soil ecological unit classification system used in Slovakia. In the Czech Republic, the method of CN curves is often used to analyse the

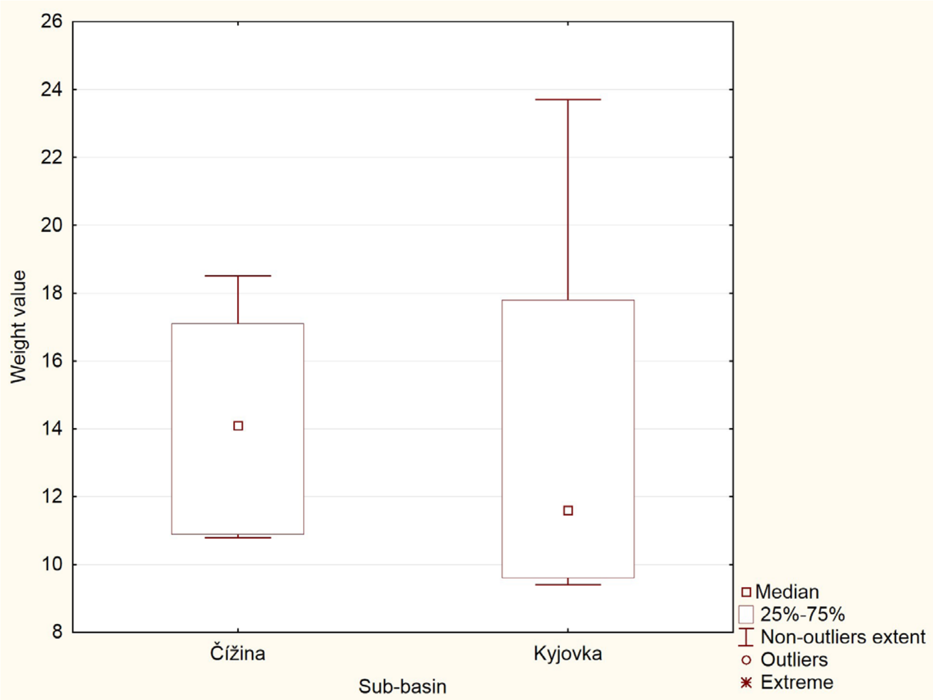

Fig. 4. Comparison of weight distribution between Čížina and Kyjovka basins.

retention capacity of the landscape. Based on the combination of the hydrologic group of soils and land use, this can determine the water retention capacity of the studied area. Another approach may be to use rain-flow data in relation to water retention. Palát et al. (2013) use multidimensional statistics tools to derive the relationship of retention characteristics to rainfall levels of the river basins.

The assessment of landscape degradation is elaborated by Salvati and Zitti (2005) who used several variables and indicators in the ESA Index (ESAI), including the assessment of climate, soil quality, vegetation cover and land management, taken as significant factors leading to land degradation. In other papers (Salvati et al., 2009, 2011), the authors describe a multivariate approach to derive the weights to be assigned to each selected indicator. The resulting Multivariate Soil Degradation Vulnerability Index (MSDVI) provided an estimation of the level of land vulnerability by aggregating more indicators. Krnáčová and Krnáč (1995) used the method of factor analysis in conditions close to CZ conditions for the identification of significant factors of the ecosystem and their relationship to environmental variables.

A number of similar studies have been conducted in Mediterranean areas, notably MEDALUS – The Mediterranean Desertification and Land Use project, using the aforementioned four quality indicators to map different types of environmental sensitivity to desertification (Basso et al., 2000; Lavado Contador et al., 2009; Ladisa et al., 2010; Salvati, Bajocco, 2011; Jafari, R., Bakhshandehmehr, L., 2016). Under CZ conditions, the issue of degradation of agricultural land has been dealt with by Šarapatka and Bednář (2015), who created an aggregated index of the total degradation, which takes into account the influence of several key degradation factors.

In our research, we used a method similar to Salvati et al. (2011). The resulting weights of individual indicators were determined on the basis of PCA analysis, and then linearly converted into a single aggregate index expressing the area's sensitivity to degradation, and water retention in particular. The aim of our work was not to directly and specifically determine individual landscape areas with problematic retention and degradation loads, but the creation of a tool for the evaluation of smaller areas, which would capture relatively problematic areas, where a number of other methods and indicators can be used for detailed research.

The method of selection of problem areas was tested in the catchment areas of Čížina and Kyjovka. These river basins differ in size, flow, climatic conditions and pedogeographic conditions, but have a similar percentage of agricultural land, land block size, drainage area, and average CN curve size. The difference lies in the structure of agricultural land; in both cases, arable land prevails (83 and 66%), but more significantly in the case of Kyjovka. The opposite is the case with permanent grassland (33 and 7%), higher in Čížina. Most of the Kyjovka river basin is occupied by arable land, which occurs evenly throughout the basin except its northern part. During long periods of rainy weather, the retention capacity of soil and vegetation is exhausted, and the overwhelming majority of the precipitation flows off the surface. It is therefore not surprising that the most important factor computed by the method is potential water erosion, in the case of the Kyjovka, which reaches average values from 0.1 to 37 t/ha/yr.

The Boxplot diagram of weighing results (Fig. 4) shows a more or less even weight distribution of individual parameters in the catchment area of Čížina, where the weight of none of the parameters significantly exceeds the average value of 13%. It is a different case in Kyjovka, where factors with higher overall impact are more pronounced – mainly erosion, but also the share of agricultural and arable land. From this, we can assume that the number of individual cases, which, in the presence of higher values (47 sub-basins in Kyjovka compared with 9 in Čížina), can bring a more significant definition of decisive factors, is important in the application of the proposed method. In the areas identified by the proposed method, we are currently examining selected soil characteristics, from which it is possible to indirectly derive soil hydro-limits and propose specific remedial measures in the landscape.

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References

Bajocco, S., Salvati, L. & Ricotta C. (2011). Land degradation versus fire: a spiral process? *Progress in Physical Geography,* 35(1), 3–18. DOI: 10.1177/0309133310380768.

Basso, F., Bove, E., Dumontet, S., Ferrara, A., Pisante, M., Quaranta, G. & Taberner M. (2000). Evaluating environmental sensitivity at the basin scale through the use of geographic information systems and remotely sensed data: an example covering the Agri basin – Southern Italy. *Catena,* 40, 19–35. DOI: 10.1016/S0341-8162(99)00062-4.

Bathurst, J.C., Sheffield, J., Leng, X. & Quaranta G. (2003). Decision support system For desertification mitigation in the Agri basin, southern Italy. *Physics and Chemistry of the Earth,* 28, 579–587. DOI:10.1016/S1474-7065(03)00104-9.

Costantini, E.A.C., Urbano, F., Aramini, G., Barbetti, R., Bellino, F., Bocci, M., Bonati, G., Fais, A., L'Abate, G., Loj, G., Magini, S., Napoli, S., Nino, P., Paolanti, M., Perciabosco, M. & Mascone F. (2009). Rationale and methods for compiling an atlas of desertification in Italy. *Land Degrad. Dev.,* 20, 261–276. DOI: 10.1002/ldr.908.

D'Angelo, M., Enne, G., Madrau, S., Percich, L., Previtali, F., Pulina, G. & Zucca C. (2000). Mitigating land degradation in Mediterranean agro-silvo-pastoral systems: a GIS-based approach. *Catena,* 40, 37–49. DOI: 10.1016/S0341-8162(99)00063-6.

ELD Initiative (2015). The value of land: Prosperous lands and positive rewards through sustainable land management. Available from www.eld-initiative.org

Fullen, M.A. (2003). Soil erosion and conservation in northern Europe. *Progress in Physical Geography,* 27, 331–358. DOI: 10.1191/0309133303pp385ra.

Gad, A. & Lotfy I. (2008). Use of remote sensing and GIS in mapping the Environmental sensitivity areas for desertification of Egyptian territory. *Earth Discussions,* 3(2), 41–85. DOI: 10.5194/eed-3-41-2008.

Geroy, I.J., Gribb, M.M., Marshall, H.P., Chandler, D.G., Benner, S.G. & McNamara J.P. (2011). Aspect influences on soil water retention and storage. *Hydrological Processes,* 25, 3836–3842. DOI: 10.1002/hyp.8281.

Gisladottir, G. & Stocking M. (2005). Land degradation control and its global environmental benefits. *Land Degrad. Dev.,* 16, 99–112. DOI: 10.1002/ldr.687.

Hawkins, R.H., Ward, T.J., Woodward, D.E. & Van Mullem J.A. (2009). *Curve number hydrology: State of the practice.* Washington: U.S.D.A.

Imeson, A. (2012). *Desertification, land degradation and sustainability*. London: Wiley. DOI: 10.1002/9781119977759.

Jafari, R. & Bakhshandehmehr L. (2016). Quantitative mapping and assessment of environmentally sensitive areas to desertification in central Iran. *Land Degrad. Dev.,* 27, 108–119. DOI: 10.1002/ldr.2227.

Jie, C., Jing-Zhang, C., Man-Zhi, T. & Zi-Tong G. (2002). Soil degradation: a global problem endangering sustainable development. *Journal of Geographical Sciences,* 12(2), 243–252. DOI: 10.1007/BF02837480.

Johnson, D.L. & Lewis L.A. (2007). Land degradation: Creation and destruction. Lanham: Rowman & Littlefield.

Kirkby, M.J., Jones, R.J.A., Irvine, B., Gobin, A., Govers, G., Cerdan, O., van Rompaey, A.J.J., Le Bissonais, Y., Daroussin, J., King, D., Montanarella, L., Grimm, M., Vieillefont, V., Puigdefabregas, J., Boer, M., Kosmas, C., Yassoglou, N., Tsara, M., Mantel, S., van der Lynden, G.W.J. & Huting J. (2004). *Pan-European soil erosion risk assessment: The PESERA map version 1*. Ispra: JRC.

Kosmas, C., Danalatos, N.G. & Gerontidis S. (2000). The effect of land parameters on vegetation performance and degree of erosion under Mediterranean conditions. *Catena,* 40, 3–17. DOI: 10.1016/S0341-8162(99)00061-2.

Kosmas, C., Tsara, M., Moustakas, N. & Karavitis C. (2003). Identification of indicators for desertification. *Annals of Arid Zones,* 42(3–4), 393–416.

Krnáčová, Z. & Krnáč S. (1995). Application of exploratory factor-analysis model to agroecological system study. *Ekológia (Bratislava),* 14(2), 123–140.

Krnáčová, Z., Hreško, J. & Vlachovičová M. (2016). An evaluation of soil retention potential as an important factor of water balance in the landscape. *Moravian Geographical Reports,* 24(3), 44–54. DOI: 10.1515/mgr-2016-0016.

Ladisa, G., Todorovic, M. & Trisorio Liuzzi G. (2010). Assessment of desertification in semi-arid Mediterranean environments: the case study of Apulia region (southern Italy). In P. Zdruli, M. Pagliali, S. Kapur & A. Faz Cano (Eds.), *Land degradation and desertification: assessment, mitigation and remediation* (pp. 493–516). New York: Springer. DOI: 10.1007/978-90-481-8657-0.

Lavado Contador, J., Schnabel, S., Gómez Gutiérrez, A. & Pulido F.M. (2009). Mapping sensitivity to land degradation in Extremadura, SW Spain. *Land Degrad. Dev.,* 20, 129–144. DOI: 10.1002/ldr.884.

Mantey, S. & Tagoe N.D. (2013). Spatial modelling of soil conservation service Curve number grid and potential maximum soil water retention to delineate flood prone areas: A case study. *Research Journal of Environmental and Earth Sciences,* 5(8), 449–456.

Marshall, M.R., Ballard, C.E., Frogbrook, Z.L., Solloway, I., McIntyre, N., Reynolds, B. & Wheater H.S. (2014). The impact of rural land management changes on soil hydraulic properties and runoff processes: results from experimental plots in upland UK. *Hydrological Processes,* 28, 2617–2629. DOI: 10.1002/hyp.9826.

Michel, C., Andreassian, V. & Perrin C. (2005). Soil conservation service curve number method: How to mend a wrong

soil moisture accounting procedure? *Water Resource Research,* 41, W02011. DOI: 10.1029/2004WR003191.
Mishra, S.K. & Singh V.P. (2006). A relook at NEH-4 curve number data and antecedent moisture condition criteria. *Hydrological Processes,* 20(13), 2755–2768. DOI: 10.1002/hyp.6066.
Palát, M., Sr., Palát, M., Jr. & Prudký J. (2013). Modelling of natural water retention in the catchment basin of the Opava river during flood. *Beskydy,* 6(2), 109–116. DOI: 10.11118/beskyd201306020109.
Rubio, J.L. & Bochet E. (1998). Desertification indicators as diagnosis criteria for desertification risk assessment in Europe. *J. Arid Environ.,* 39,113–120. DOI: 10.1006/jare.1998.0402.
Rubio, J.L. & Recatala L. (2006). The relevance and consequences of Mediterranean desertification including security aspects. In W.G. Kepner, J.L. Rubio, D.A. Mouat & F. Pedrazzini (Eds.), *Desertification in the Mediterranean region: A security issue* (pp. 133–165). Dordrecht: Springer. DOI: 10.1007/1-4020-3760-0_05.
Salvati, L. & Zitti M. (2005). Land degradation in the Mediterranean basin: linking bio-physical and economic factors into an ecological perspective. *Biota,* 5, 67–77.
Salvati, L., Zitti, M., Ceccarelli, T. & Perini L. (2009). Developing a synthetic index of land vulnerability to drought and desertification. *Geographical Research,* 47 (3), 280–291. DOI: 10.1111/j.1745-5871.2009.00590.x.
Salvati, L. & Bajocco S. (2011). Land sensitivity to desertification across Italy: past, present, and future. *Applied Geography,* 31, 223–231. DOI: 10.1016/j.apgeog.2010.04.006.
Salvati, L., Bajocco, S., Ceccarelli, T., Zitti, M. & Perini L. (2011). Towards a process-based evaluation of land vulnerability to soil degradation in Italy. *Ecological Indicators,* 11, 1216–1227. DOI: 10.1016/j.ecolind.2010.12.024.
Salvati, L., Bajocco, S., Ceccarelli, T. & Perini L. (2013). Amplifying (or reversing) the territorial disparities in land vulnerability to soil degradation: The case of Italy. *Professional Geographer,* 65, 647–663. DOI: 10.1080/0033124.2012.724351.
Salvati, L., Zitti, M. & Perini L. (2016). Fifty years on: long-term patterns of land sensitivity to desertification in Italy. *Land Degrad. Dev.,* 27, 97–107. DOI: 10.1002/ldr.2226.
Santini, M., Caccamo, G., Laurenti, A., Noce, S. & Valentini R. (2010). A multi-model GIS framework for desertification risk assessment. *Applied Geography,* 30(3), 394–415. DOI: 10.1016/j.apgeog.2009.11.003.
Simeonakis, E., Calvo-Cases, A. & Arnau-Rosalen E. (2007). Land use change and land degradation in southeastern Mediterranean Spain. *Environ. Manag.,* 40, 80–94. DOI: 10.1007/s00267-004-0059-0.
Singh, P.K., Bhunya, P.K., Mishra, S.K. & Chaube U.C. (2008). A sediment graph model based on SCS-CN method. *J. Hydrol.*, 349(1–2), 244–255. DOI: 10.1016/j.jhydrol.2007.11.004.
Soulis, K., & Dercas N. (2007). Development of a GIS-based spatially distributed continuous hydrological model and its first application. *Water International,* 32(1), 177–192. DOI: 10.1080/02508060708691974.
Šarapatka, B. & Bednář M. (2015). Assessment of potential soil degradation on agricultural land in the Czech Republic. *J. Environ. Qual.*, 44(1), 154–161. DOI: 10.2134/jeq2014.05.0233.
Thornes, J.B. (2004). Stability and instability in the management of Mediterranean desertification. In J. Wainwright & M. Mulligan (Eds.), *Environmental modelling: Finding simplicity in complexity* (pp. 303–315). Chichester: Wiley.