Phytoindication assessment of the effect of reconstruction on the light regime of an urban park

O. M. Kunakh*, O. I. Lisovets*, **, N. V. Yorkina***, Y. O. Zhukova*

*Oles Honchar Dnipro National University, Dnipro, Ukraine
**Dnipro State Agrarian and Economic University, Dnipro, Ukraine
***Bogdan Khmelnitsky Melitopol State Pedagogical University, Melitopol, Ukraine

Introduction

The factors that affect plant communities vary depending on the hierarchical scale of the study (Ganjouri et al., 2012; Angeler et al., 2013). At the global and continental level, the species richness of plant communities is mainly influenced by climatic factors (Currie, 1991; Warmeck, 2000; Xu et al., 2016). At the regional level, in addition to climatic factors, relief effects and landscape diversity become important (Isbell et al., 2017; Putchkov et al., 2019; Harrison, 2020). The factors determining species richness vary by region. Elevation range and net primary productivity are important predictors of plant community diversity. Also, elevation range is positively related to plant species richness. This indicates that elevation range often becomes the dominant controlling factor in regions with sufficient energy (Xu et al., 2016). At the same time, the role of ecological features of individual plant species and their assemblages, as well as the peculiarities of their evolutionary history, is increasing. On the local scale, climatic factors are of secondary importance due to their small spatial variations within the study areas. However, on the local scale, the habitat characteristics such as topography, light exposure, tree crown cover, and soil properties attract special attention (Bryadyrenko, 2015; Samec et al., 2021). The light condition is important for ecology of forest and park plantations (Jennings, 1999). Solar radiation in the form of light is a leading factor in many biological, ecological, physiological, and hydrological processes (Van der Zande et al., 2011; Zhukov et al., 2021). The undergrowth light environment is a key determinant of vegetation structure and ecosystem processes, and varies spatially perhaps more than any other resource used by plants. The light exposure of undergrowth varies along vegetation structure gradients from a grassland with no tree cover to a forest with nearly complete tree cover (Martens et al., 2000). Radiation in forest and park stands affects photosynthesis, transpiration, vegetation structure (Peng et al., 2014), stand development dynamics (Zavala et al., 2007), tree plant growth (Grant, 1997), and production efficiency (Englund et al., 2000). Significant positive correlations were found between incident solar energy and aboveground biomass in forest ecosystems (Zavitkovski, 1976). The influence of solar radiation also manifests in the forest litter (Malisev et al., 2017; Turovou et al., 2018).

Light passing through canopy affects germination and growth of understory (Anderson & Denmead, 1969; Grant, 1997; Kyureh et al., 1999), regeneration and succession (Sakai & Akiyama, 2005; Van der Zande et al., 2010), soil condition (von Arc et al., 2012; Muselman et al., 2013), and biodiversity (Battisti et al., 2013; Serce et al., 2017; Chaplygina et al., 2018; Tsai et al., 2018). The light regime in forest and park plantations...
depends on canopy structure, site characteristics, atmospheric conditions, and sun altitude (Jones et al., 2003; Alexander et al., 2013; Bode et al., 2014; Brygadyrenko, 2016). These factors create a complex light pattern of the underground, which expresses not only in horizontal heterogeneity, but also vertical variation at any given time (Peng et al., 2014).

The vegetation is able to improve the urban microclimate through shading (Bartels & Chen, 2010; Adler et al., 2011; Li et al., 2018; Storch et al., 2018) and by evaporation (Dimoudi & Nikolopoulos, 2003; Georgi & Dimitriou, 2010; Duarte et al., 2015). In turn, the urban vegetation may adapt to climate change itself (Harnada & Ohta, 2010). Urban trees can compensate or reverse the effects of heat islands. Mitigating the effects of urban heat islands has the potential to reduce energy use for air conditioning and improve urban air quality (Adbari et al., 2001). Influencing the microclimate with plants can improve the comfort of the city population (Lin et al., 2010; Shahidian et al., 2010). The air temperature in the shade of trees in summer is significantly lower than in areas without trees. Multi-level plant communities are the most effective in terms of their cooling and moisturizing effect (Zhang et al., 2013). The planting of grass in combination with shade trees can have a pronounced effect on the reduction of air temperature (Shashnas-Bar et al., 2010). The structural characteristics of urban green space plant communities are important factors in mitigating the effects of heat. The cooling and moisturizing effects are controlled by canopy density, leaf area index and average leaf angle (Qin et al., 2014), as well as canopy density, canopy area, tree height, and solar radiation (Zhang et al., 2013). There are many factors that can influence the attractiveness of urban parks to residents, including features of plant community structure (Xue et al., 2017a, 2017b; Zhang et al., 2021). The density and light regime of park stands influenced the degree to which parks were preferred by different resident groups. Moderately dense plantings were the most preferred. A curvilinear effect of respondents’ age on preference for moderately and densely vegetated scenes was found. Respondents in their early 40s expressed the greatest preference for moderate and dense vegetation compared to younger and older respondents. The preference for moderate and dense vegetation also increased as respondents’ level of education increased (Bjerke et al., 2006). Separate physical attributes have no predictive power to explain residents’ preferences for particular environmental conditions. Perception-based variables proved to be the most effective, with parkland openness and soil surface smoothness proving to be particularly useful predictors (Kaplan et al., 1989; Strumre, 1994b, 1994a). Forest patches with dense vegetation support a diversity of wild-life habitats. On the other hand, the aesthetic quality of the natural environment is associated with more open grassy areas, where random groups of trees and shrubs are represented (Parsons, 1995). The structural characteristics of plant communities affect the ability to mitigate the thermal environment and influence the perceived suitability of park areas for outdoor recreation (Li et al., 2018; Dorman et al., 2020).

Urban habitats differ notably in the diversity of their vascular plant flora (Lossov et al., 2011; Shekhtnova & Mal’ beva, 2015). Socio-economic, environmental, and conservation factors influence the diversity and composition of plant communities in urban parks (Figueroa et al., 2015). Anthropogenic impacts alter the structural development of urban forests, affecting both positively through the formation of gaps, the appearance of deadwood, and the formation of ecosystem complexity, and negatively through the shifts in composition and successional trajectories (Alasmery et al., 2020; Matsalu et al., 2021). Plants are an important component of landscape design (Maltsve & Maltsve, 2018; Goncharenkova & Yatsenko, 2020). They perform a number of functions in a holistic landscape (Maltsve et al., 2017). Landscaping projects focus on the importance of plants in improving ecology and the environment. A park reconstruction allows space to be created for plants without altering the original ecological environment in the park, thereby achieving an optimal visual experience for tourists (Li, 2020). The individual plants are sensitive to a variety of environmental properties and thus can be indicators of the ecological factors. The sensitivity of an individual species to the action of environmental factors can vary according to Shelford's law of tolerance (Shelford, 1931), so a reliable assessment of environmental properties can be made on the basis of studying the species composition of the plant community. Phytoindication methods based on Ellenberg indicator values (EIVs) (Ellenberg, 1979) or Didulh (Didulh, 2011) are often used to assess habitat conditions. The indicator scales can be ranges (Didulh, 2011) or such that indicate the ecological optimum of the species (Ellenberg, 1979). In the Ellenberg system, the ecological optima of individual plant species are expressed in the form of ordinal numbers. The average values of Ellenberg indices calculated for vegetation sites make it possible to estimate specific habitat conditions (Diekmann, 2003). To assess the properties of the environment using the range scales of Didulh (Didulh, 2011), either the weighted average method (Didulh, 2011) or the ideal indicator method (Buzuk, 2017) is used. Using phytoindication as a surrogate for directly measured environmental variables saves time and reduces research costs (Dzwonko, 2001). Numerous studies confirm the usefulness and validity of the phytoindication method. The correlations between phytoindication estimates of environmental properties and the results of physical and chemical field measurements were proven (Ellenberg et al., 1991; Diekmann, 1995; Ertsei et al., 1998; Schaffers & Sykora, 2000). Nevertheless, there are also problems associated with the use of indicator analyses. Statistical inferences should not be made in analyses that relate phytoindicator values to other variables derived from species composition, as this can lead to highly biased results and misinterpretations (Zeleny & Schaffers, 2012, Wildi, 2016). The phytoindicator scales demonstrate the maximum effectiveness in studies on large temporal or spatial scales (Pignatti et al., 2001). There are few studies on the relationship between the morphological properties of plants and indicator scale values of light (L-numbers). A study on annual weeds showed a significant negative correlation of seed weight with L-numbers. In annalas, heavier seeds are more common in species from more shady habitats. A likely advantage of this may be stronger support for germinating seeds when energy requirements cannot be satisfied by photosynthesis alone (Bartelheimer & Poschlod, 2016). The L-numbers of perennial grasses are negatively correlated with germination rate (Grime et al., 1981). A possible biological explanation could be that high-intensity habitats are prone to drought, so slow germination increases the risk of desiccation (Bartelheimer & Poschlod, 2016). There are few studies on the relationship between light indicator values and light intensity in forests. The values of this indicator are good predictors of relative light intensity in deciduous forests (Diekmann, 1995; Dzwonko, 2001). The positive correlation of indicator values of temperature and light with the average annual temperature was explained by the adaptation of heliophilius species to locations where the high temperatures are always associated with intense solar radiation (Murcén & Guarino, 2015).

The phytoindication approach was shown to be effective in assessing the impact of silvicultural practices on ecological regimes (Hamrez & Hänell, 1997). Phytosociological method is applied to assess ecological lands for the conservation of vascular plant diversity in urban environments (Zhukov et al., 2017; Dyderski et al., 2017). According to the mean Ellenberg indicator values calculated for the five land use types within urban environments, densely built-up areas were characterized by higher light availability, soil nutrients, soil pH, and lower soil moisture compared with open built-up areas (Godefroid & Ricotta, 2018). Anthropogenic parts of cities are typically dry (Grinn et al., 2008), unshaded (Chocholoušová & Pyšek, 2003), and dominated by building materials that tend to be highly alkaline (Deutz et al., 2017). Environmental conditions, as determined by Ellenberg values, were found to be similar for areas of dense development and industrial zones (Godefroid & Ricotta, 2018). In Europe, a shift in the structure of urban plant communities toward shade-tolerant species has been occurring over the past 120 years. This trend was revealed by means of Ellenberg indicator values analysis. Such trends are mainly due to an increase in the number of parks and trees in urban areas. The climate change and the presence of artificially irrigated areas in the city led to an increase in hygrophilous and drought-tolerant species. The temperature index showed a significant increase in the number of macrothermal species that are adapted to a warmer climate as a response to the urban heat island effect (Salinoto et al., 2019). Ellenberg’s indicator values revealed that since 1940, the flora of the Brussels region has become more nitrophilic and shade-tolerant, with changes in relation to moisture and soil reaction as well as temperature (Godefroid, 2001).

Municipalities use ecological restoration of urban forests as a measure to improve air quality, mitigate urban heat island effects, improve storm-water infiltration, and provide other social and environmental benefits. 

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However, the dynamics of plant communities following urban forest restoration are poorly documented. The objectives of this study were to: (1) evaluate the impact of urban park reconstruction on the condition of herbage cover; (2) perform the phytoindication of changes in the light regime caused by the reconstruction of the park and (3) to find out the dependence of reliability of phytoindication estimation on the number of species in the relevé.

Materials and methods

Study area. The study was conducted in the recreational area of the Botanical Garden of Oles Honchar Dnipro National University (Ukraine). The tree plantation was created after the Second World War in the location of a natural oak forest. In 2019, a 2.8 ha area of the park was reconstructed. The samples were taken within polygons, two of which were in the reconstruction area and two of which were in a similar section of the park where no reconstruction was performed. During the reconstruction, walkways were rebuilt, shrubs were removed, old, damaged trees were removed, and tree crowns were trimmed. Juvenile trees were planted in the places of the removed old trees. Old outbuildings, which greatly im paired the aesthetic perception of the park, were also removed. Transport and construction machinery was involved in the reconstruction. The works were carried out during the warm period of the year.

Data collection. Each polygon consisted of 105 sample points. The points were located along 7 transects with 15 sample points in each. The distance between points in the transects, as well as the distance between transects, was 3 m. The adjacent sample points were in close proximity to each other. The vascular plant species lists were composed for each 3 × 3 m sample point along with a visual assessment of species cover using the nine-degree Braun-Blanquet scale (Westhoff & Van Der Maarel, 1978). The projective cover of plant species was recorded at soil, understory (up to 2 m in height), and canopy (above 2 m in height) levels. The species were all identified to species level in all plots. The seedlings and saplings of tree species were subsequently excluded from the analysis.

Phytoindicator procedure. The scales were used for phytoindication according to Didud (2011). Species live along environmental gradients. The distribution of a species along a gradient is known as a response curve. The concept of the species response curve is the theoretical basis for the development of phytoindication approaches (Didud, 2012). The conceptual form of the response curve is traditionally thought to be unimodal and symmetric (bell-shaped or Gaussian) (ter Braak & Looman, 1986), but various other forms of response curves can be observed (Huisman et al., 1993; Lawesson & Oksanen, 2002). The actual distributions of species are most often asymmetric, so they cannot be described by a Gaussian distribution (Austin et al., 1994). The asymmetric distributions of species are extremely common. The tailed of the distributions tend to be directed toward more favorable values of environmental factors (Austin et al., 1990; Austin, 2013). The asymmetric nature of species distributions requires adjustments to the procedure of determining the species optimum as an indicator value for evaluating environmental properties. To model the response function of species in a gradient of environmental conditions, a β-function was proposed (Austin, 1976; Austin et al., 1994):

\[ V = k \times (x - a)^{\alpha} \times (b - x)^{\gamma} \]

where \( V \) is a measure of the species abundance; \( k \) is a constant; \( a \) and \( b \) indicate the smaller and larger boundary of the species in the gradient of variable \( x \); \( \alpha \) and \( \gamma \) are distribution shape parameters. In the range of values of \( a \) and \( b \), the \( \beta \)-function can exhibit a considerable variety of possible shapes of distributions from a close approximation of the Gaussian distribution to a highly asymmetric distribution. To solve the problem in general form, the analytical representation of the \( \beta \)-function could be simplified by setting the dependence of the unknown shape indices \( a \) and \( \gamma \) on the known parameters \( a \) and \( b \) (Zhukov et al., 2018). Then the function takes the following form:

\[ V = k \times (x - a)^{\alpha} \times (b - x)^{\gamma} \]

where \( x \) is normalized to range 0–1. For the case where \( a = 1-b \), the \( \beta \)-function will be symmetric. For the case where \( a = 0 \) or \( b = 1 \), the maximum of the function will be either at \( x = 0 \) or \( x = 1 \), respectively. For transient values of \( a \) and \( b \), the \( \beta \)-function is asymmetric and gradually shifts between the two extremes of the distribution (Zhukov et al., 2018). In the equation, the parameter \( k \) has the character of normalization coeffi cient. At the point of optimum, the index of species abundance reaches the highest value of 1 (or 100%). At least one species in the community can be assumed to be under optimal conditions and to be dominant. Therefore, the abundance index of the dominant species (projective cover, crown density, number of shoots) can be taken as the maximum 100% in the community among all plants of the same layer. The relative abundance of a species in the community can be estimated as the ratio of the observed abundance of a species (\( p \)) to the abundance of the dominant species (\( p_{dom} \)) within a layer: \( p/p_{dom} \). The information about the abundance of a species in a community greatly increases its phytoindication value. If the response function of the species is estimated, the abundance of the species in the community significantly narrows the range of conditions that can respond to the observed abundance index. The phytoindication scales were normalized to the range of 0–1. The distribution of plant projective cover along the ecological factor gradient was modeled using beta function. The range scales values according to Didud (2011) were used as the threshold values of the function. Thus, modeling the response of species along the environmental factor gradient made it possible to recalculate “tabular” values of the range phytoindication scale into new local scales that account for species abundance in particular communities. Further, for the purpose of phytoindication of environmental factors, we used the ideal indicator method of Buzuk (2017). This approach is based on the assumption that ecological conditions can be most accurately indicated by a species with zero tolerance, i.e., a species whose minimum and maximum threshold points coincide. Obviously, no such species exists, but the properties of the ideal indicator can be calculated by determining the dependence of the upper and lower threshold values on the tolerance index, which is the difference of these upper and lower threshold values. The regression dependence for both the maximum threshold value and the minimum threshold value on tolerance pass through the same point on the ordinate axis, which corresponds to zero tolerance. Such a situation corresponds to a hypothetical species that can inhabit only a given habitat with an appropriate ecological regime. This virtual species is the ideal indicator.

The original ideal indicator procedure does not take into account species abundance. Our methodology does not use “tabulated” values of threshold points, but rather recalculated with taking into account species abundance.

Permutation test. The estimation of phytoindication parameters can be carried out for objects of different dimensions. As the area of the object under study increases, the number of species naturally increases and thus, the accuracy of phytoindication assessment increases. However, such estimation is accurate for the whole studied area, but there is no reason to believe that it is valid for each individual site within the studied area. Reducing the sample area decreases the number of species in the description, and thus the accuracy of the estimate, but the resolution of the method increases. To the data we obtained, we applied the permutation method, which allowed us to estimate the probability of difference between the phytoindicator estimates in a given site and the values of environmental factors estimated based on a random sample of species from the list of all species that occur in the polygon. Note that the standard permutation test procedure uses null distribution created by calculating the test statistics using randomized data. To account for the similarity problem, a modification of this first step has been proposed, the specificity of which is that the test statistic is calculated not using randomized averages of the real indicator values, but using (non-randomized) mean randomized indicator values instead (i.e., randomizing the species indicator values among the species in the table instead of randomizing the calculated averages) (Zelený & Schaffers, 2012). The feature of our variant of the permutation procedure is that the null distribution is generated by randomly choosing species from the list of species for the polygon, rather than by randomly choosing indicator values that occur in species within the same polygon. The experimental value of the statistics and the values of the randomly generated sample were compared using the ascending function from the ade4 library (Dray & Dufour, 2007).

Results

A total of 65 plant species were found within the studied polygons. The phanerophytes were represented by 11 species, the nonphanerophytes were represented by two species, the hemicyryptophytes were represented...
by 29 species, the therophytes were represented by 16 species, and the geophytes were represented by 7 species. At each individual site, the number of species in the herb layer ranged from 5 to 19 (Table 1). Differences between the polygons in the number of herbaceous species in the individual site were statistically significant (F = 55.3, P < 0.001). The number of herbaceous species in the park area after reconstruction was higher compared to the unreconstructed area (F = 119.7, P < 0.001). Tree canopy crown closure varied widely, from 0% to 95%. The differences in this indicator between the polygons are statistically significant (F = 381.7, P < 0.001). The crown closure in the reconstructed area was significantly lower than such in the untreated conditions (F = 90.8, P < 0.001). The polygons in the reconstruction zone (polygon 1 and 2) did not differ from each other in terms of crown cover (planned comparison F = 0.38, P = 0.55). The polygon 4 had significantly more crown closure than polygon 3 in the untreated conditions (planned comparison F = 191.5, P < 0.001).

The grass height on average was 0.60 ± 0.01 m. The differences in grass height between the polygons were statistically significant (F = 107.9, P < 0.001). The grass height in the reconstruction zone was higher than without reconstruction (F = 26.1, P < 0.001). The average projective cover of the grass was 66.9 ± 1.0%. The differences between the polygons in projective grass cover were statistically significant (F = 58.1, P < 0.001). The differences between the zones after and without reconstruction were statistically insignificant (F = 1.53, P = 0.21).

The phytoindication assessment showed that the light regime varied 3.8 (the conditions suitable for the scyphophytes – plants of typical foliage forests) to 8.75 (the conditions suitable for the sub-heliophytes – plants of light forests and shrubberies, or high herbaceous communities; lower layers are in the shade). The typical light level was 6.39 ± 0.06, which corresponds to the conditions that favour hemi-scyphophytes – plants of little-closed foliage forests. The light mode was statistically significantly different between the polygons (F = 5.4, P < 0.001). The light regime in the park area after reconstruction was statistically significantly different from the regime in the untreated park area (F = 646.7, P < 0.001). The lighting regime after the reconstruction was favourable to sub-heliophytes, and in the unreconstructed area the regime favoured hemic-scyphophytes. The probability < 0.1 of difference between the phytoindication estimate of light at a given site and the estimate obtained from a randomized sample from a list of species at the same extent range occurred in 20.1% and 19.1% of cases at polygons 1 and 2 (Fig. 1). This value is 34.3% and 49.5% at polygons 3 and 4. This probability correlates positively with the number of species (r = 0.26, P < 0.001). The proportion of hemic-scyphophytes in the plant community increases with increasing light regime (r = 0.36, P < 0.001), and the proportion of therophytes decreases (r = −0.35, P < 0.001). This pattern is most pronounced for the untreated sites.

Tree canopy crown closure negatively correlated with grass height (r = −0.32, P < 0.001) and with herbaceous layer projective cover (r = −0.16, P = 0.001). The correlation between grass height and herbaceous layer projective cover was statistically significantly positive (r = 0.44, P < 0.001). The tree canopy crown closure, grass height, and herbaceous layer projective cover were able to explain 86.0% of the phytoindication assessment of the lighting regime variation (Table 2). The regression coefficients indicated that the tree canopy crown closure, grass height, and herbaceous layer projective cover negatively affected the light regime. This pattern was common to the entire park plantation, but the patterns identified had their own specificity depending on the type of polygon. The positive regression coefficients for the Polygon×Crown closure predictor indicate that the rate of decrease in light intensity with increasing crown cover in polygons 1 and 2 was less than such for polygon 4, which was considered the reference. This result can be illustrated graphically (Fig. 2). The pairwise correlation coefficients of the tree canopy crown closure and the phytoindication assessment of the lighting regime were statistically insignificant for polygons 1 and 2 (Fig. 3), whereas the pairwise coefficient for polygon 3 was greater in modulo than for polygon 4 (P < 0.001). According to GLM results, the rate of change in light with changes in grass height did not differ between polygons 3 and 4, whereas this relationship was weaker for polygons 1 and 2. The pairwise correlation coefficients indicated a positive correlation between grass height and illumination for polygon 4 (Fig. 4). Obviously, taking into account the influence of other factors, which is carried out in the GLM procedure allows us to obtain an adequate assessment of the relationship between the indicators.

### Table 1

Descriptive statistics of number of herbaceous plant species, tree canopy crown closure, grass height, herbaceous layer projective cover, and phytoindication assessment of the lighting regime of park plantation vegetation cover

| Parameter                  | Polygon | x ± SE | Minimum | Maximum |
|----------------------------|---------|--------|---------|---------|
| Number of herbaceous plant species | 1 (N = 105) | 9.79 ± 0.18 | 6 | 16 |
|                           | 2 (N = 105) | 11.02 ± 0.24 | 5 | 19 |
|                           | 3 (N = 105) | 8.69 ± 0.14 | 6 | 13 |
|                           | 4 (N = 105) | 7.40 ± 0.26 | 5 | 17 |
| Total (N = 420)            | 9.22 ± 0.12 | 5 | 19 |

Table 2

General linear model of the effect of the tree canopy crown closure, grass height, and herbaceous layer projective cover on the phytoindication assessment of the lighting regime (Radj2 = 0.86, F = 198.0, P < 0.001)

| Predictors | Level of effect | β-regression coefficient x ± SE | | |
|------------|-----------------|---------------------------------|---|---|
| Crown closure | – | –0.20 ± 0.09 | –0.51 ± 0.06 | 0.020 |
| Grass height | – | –0.28 ± 0.09 | –0.45 ± 0.11 | 0.001 |
| Projective cover | – | –0.26 ± 0.07 | –0.39 ± 0.13 | 0.001 |
| Polygon | 1 | 0.74 ± 0.11 | 0.53 ± 0.06 | 0.96 |
| 2 | 0.47 ± 0.11 | 0.24 ± 0.09 | 0.69 |
| 3 | 0.41 ± 0.09 | 0.22 ± 0.06 | 0.59 |
| Polygon×Crown closure | 1 | 0.29 ± 0.06 | 0.16 ± 0.04 | 0.41 |
| 2 | 0.40 ± 0.08 | 0.25 ± 0.06 | 0.56 |
| Polygon×Grass height | 3 | –0.81 ± 0.08 | –0.98 ± 0.05 | 0.001 |
| Polygon×Grass height | 1 | 0.14 ± 0.05 | –0.23 ± 0.03 | 0.005 |
| Polygon×Grass height | 2 | 0.17 ± 0.07 | –0.30 ± 0.03 | 0.015 |
| Polygon×Grass height | 3 | –0.08 ± 0.07 | –0.22 ± 0.05 | 0.241 |
| Polygon×Projective cover | 1 | –0.23 ± 0.09 | –0.40 ± 0.05 | 0.013 |
| Polygon×Projective cover | 2 | –0.18 ± 0.10 | –0.39 ± 0.02 | 0.080 |
| Polygon×Projective cover | 3 | –0.10 ± 0.09 | –0.28 ± 0.07 | 0.251 |
| Crown closure × | – | 0.06 ± 0.07 | –0.07 ± 0.19 | 0.343 |
| Grass height | Crown closure × | – | 0.10 ± 0.09 | –0.28 ± 0.08 | 0.275 |
| Projective cover | – | 0.32 ± 0.09 | 0.13 ± 0.50 | 0.001 |

According to GLM results, the relationships between projective grass cover and light did not differ between polygons 3 and 4. The relationship was weaker for polygons 1 and 2. The pairwise correlation coefficients indicated a negative correlation for polygons 1, 2, and 3 and no correlation for polygon 4 (Fig. 5). The influence of grass height and density had a positive synergistic effect on the light regime.
Fig. 1. The distribution histogram of probabilities of difference between observed phytoindication estimates and those obtained as a result of estimates for species populations, which are the result of random sampling from the total list of species in the polygon according to the permutation test results: the abscissa axis is the P-value, the ordinate axis is the number of observations; a is polygon 1 (N = 105), b is polygon 2 (N = 105), c is polygon 3 (N = 105), d is polygon 4 (N = 105)

Fig. 2. The dependence of the phytoindication assessment of the light regime on the crown density of tree plants and shrubs: X-axis is crown closeness, %; Y-axis is the assessment of the light regime, points; a is polygon 1 (N = 105), b is polygon 2 (N = 105), c is polygon 3 (N = 105), d is polygon 4 (N = 105)

Discussion

The ground vegetation is of great value as an indicator of the management impact on the structure and function of forests and park plantations (Halpern & Spies, 1995; Brunet et al., 1996; Rédei et al., 2020; Oettel & Lapin, 2021; Stefanovska et al., 2021). The trajectories of successional vegetation dynamics differ between forests after reconstruction and without reconstruction. The restoration procedures created areas favourable to the germination and growth of species adapted to high-light conditions and disturbed soils (Johnson & Handel, 2015). The data obtained in our study indicate that the reconstruction of the park affected the number of vascular plant species. Changes in the light regime of the park plantation as a result of thinning of the canopy of tree plants and the destruction of the shrub layer created conditions for the growth and development of phytotrophic species of the herbaceous layer. These results are in agreement with findings that indicate that the soil and light conditions influence the species richness of ground vegetation depending on vegetation type (Härdtle et al., 2003). In humid forests, species richness has a strong positive correlation with soil moisture, while light conditions and nutrient supply mostly have no effect on species richness. In meso- and eutrophic forests, species richness is closely correlated with soil activity and base and nitrogen availability. Improving light conditions for terrestrial vegetation does not increase the number of typical forest species because most of them are shade-tolerant and have very limited light requirements. Species richness in acidophytic forests depends mainly on canopy closure and internal light conditions. Therefore, soil moisture, nutrient input, and light
availability must be evaluated differently in terms of their influence on the number of terrestrial vegetation species and with respect to individual forest communities, as they vary with site conditions. If the relationship between species richness and environmental factors, such as soil conditions, is not analyzed for individual forest communities, but for the whole forest community, relationships may be found that do not help to explain the species richness of a particular forest community and thus have only limited utility for appropriate forest management recommendations (Härdtle et al., 2003). Our results also highlight the fact that the impact of park reconstruction is also superimposed on the specific characteristics of a particular park site.

The plant community transformations should also be considered in the context of changes in the information value of vegetation for phytoidication of ecological regimes. The widespread use of phytoidicator scales to resolve ecological problems raises the question of the overall reliability of the results obtained (Otýpková, 2009). The average values of phytoidicator scales can be robust indicators even in conditions of incomplete species list in a plant community. The robustness of phytoidicator values to incomplete floristic lists is due to their association with the most stable structural features of the community. Improved local estimates of environmental factors can be obtained from the features of rare species (Ewald, 2003). Community species composition and number of species depend on the size of the survey area (Rosenzweig, 1995). It was found that phytoidication estimates of environmental factors do not differ between sites of different sizes in both homogeneous and heterogeneous sites (Otýpková, 2009). The author of the study believes that this result is a consequence of the fact that species that occur on increasingly larger plots have similar values of phytoidication scales to those found earlier. In a number of other studies, phytoidication of ecological regimes was conducted with the assumed ability to measure properties with a high degree of accuracy at a fine spatial scale. This assumption was supported by arguments that were based on geostatistical calculations (Zhukov et al., 2018).

Fig. 3. The dependence of the phytoidication assessment of the light regime on the height of the grass stand: X-axis is grass height, m; Y-axis is the assessment of the light regime, points; a is polygon 1 (N = 105), b is polygon 2 (N = 105), c is polygon 3 (N = 105), d is polygon 4 (N = 105)

Fig. 4. The dependence of the phytoidication assessment of the light regime on the projective cover of grass stand: X-axis is the projective cover of herbage, %; Y-axis is the assessment of the light regime, points; a is polygon 1 (N = 105), b is polygon 2 (N = 105), c is polygon 3 (N = 105), d is polygon 4 (N = 105)
The number of species in the geobotanical relevés affects the reliability of phytoindication estimates. When the number of species in the description is 15–20 and more, the method of phytoindication provides a generally adequate assessment of ecological regimes (Didulik, 2012). There is a point of view that there should not be less than 5 species in the description for sufficient accuracy of ecological assessments (Tsatsenkin, 1970). The conclusions should be taken into account as based on expert evaluations and such that have no sufficient statistical basis. Our studies show that the number of species on 3×3 meter sites was not less than 5, which suggests sufficient reliability of the obtained phytoindication estimates.

There is a problem of logical circle (tautology) when trying to explain vegetation patterns using biondication (Szymura et al., 2014). The problem arises from the fact that phytoindication estimates of environmental factors have two sources. These are the phytoindication values of individual species that indicate their ecological features and the composition of species in the geobotanical record from which the phytoindication assessment is made. The effect of retaining information on species composition and their similarity or difference compared to other compositions in the phytoindication assessment is called the “similarity problem” (Zelený & Schaffers, 2012; Zhukov et al., 2018). To account for the problem of ecological similarity between species, a modification of this first step in the permutation test process is that randomization of species indicator values among species in the description is applied instead of randomization of the calculated mean indicator values (Zelený & Schaffers, 2012). In our study, we aimed at a somewhat different task, namely to evaluate the specificity of indicative assessments at sites 3 × 3 m in size compared to the polygon as a whole. In other words, we wanted to solve the issue of the reliability of phytoindication within the chosen spatial scale. The permutation test we proposed showed that the park reconstruction resulted in change in the vegetation cover so the level of specificity of individual sites decreased compared with the situation observed within the untreated polygons. This observation may be due to anthropogenic homogenization, which leads to the evening out of ecological conditions in the area of anthropogenic impact. The environmental homogenization as a result of reconstruction was accompanied by increase in the species diversity of the plant community due to increase in the number of photophilic species.

The permutation test also provides an indication of the reliability of phytoindication assessments. The reliability of phytoindication estimates in the context of the ability to indicate site-specific features of the ecological regime was found to decrease with an increase in species richness. Shelford's law of tolerance (Shelford, 1911, 1931) predicts that species in the optimum zone are the least sensitive to changes in environmental conditions. This explains the fact that the approach of a community to the optimum conditions for most species is followed by a decrease in its phytoindicator value. As an alternative hypothesis, we can consider the scenario of increase in the role of interspecific interactions with increase in the species diversity of the community. The diversity is a factor of community stability, and therefore independence from fluctuations in environmental conditions (Thienbaul & Loreau, 2005; Zymaraeva et al., 2021). The phytoindication concept is based entirely on the ecological niche theory (Hutchinson, 1965). However, ecological niche theory is not the only concept that is applied to explain community structure. Neutrality theory also has considerable explanatory power (Hubbell, 2001; Hu et al., 2006; Adler et al., 2007; Eitenne, 2009; Zhukov et al., 2019), especially at the detailed spatial level (Zhukov et al., 2018; Zhukov et al., 2019). The neutral effects are related to dispersal. The creation of new ecological space as a result of reconstruction led to decrease in crown density, resulting in increase in projective cover of herbaceous vegetation. The effect of the mentioned ecological properties on the indication assessment of light is unidirectional: increase in crown density, height and projective cover of vegetation leads to decrease in light indicator values. The mentioned patterns were common for the park as a whole, while significant deviations from typical dependencies were observed as a result of the reconstruction. The role of the crown cover factor in structuring the herbaceous layer was most important in the untreated part of the park.

For the area after the reconstruction, crown cover was not important in the variability of the light regime. The most probable reason was the delayed development of vegetation cover after the abrupt anthropogenic transformation. The vegetation cover has not yet had time to adapt to the ecological conditions, which suddenly changed after the trimming of branches of tree plants and the removal of shrubs. The response of the light indicator to the effect of grass height in the reconstruction and untreated zones was opposite. In the reconstruction zone, the increase in height was associated with the decrease in the light level of the ecotope. The result quite predictably was that the higher grass stand was able to absorb more solar energy and had a greater ability to form shade. The trend of increasing light levels with increasing plant height is apparent in areas of the park without reconstruction. This pattern can be explained by the location of higher herbaceous plants in areas without tree vegetation, which are better supplied with light than the polygon as a whole.

Under reconstruction, the amount of light decreased with increasing projective cover of herbaceous vegetation. No such dependence was found in the untreated conditions. Thus, the leading driver of the light regime of the forest park plantation was crown closure. After reconstruction, the leading role in varying of the light regime transferred to the grass cover. Obviously, the return of the leading role of the crown closure in spatial variation of light regime should be considered as a marker in the process of vegetation recovery after reconstruction.

The vascular plants can be divided into life forms. The influence of environmental factors is different for different groups of plants. Among the main life forms, the highest level of relative species richness in the temperate climate zone is achieved by the hemerophytes (Raunkiaer, 1937). The proportion of phanerophytes is maximal in the humid tropics, the proportion of therophytes is maximal in the desert, and the proportion of hemerophytes is maximal in the dry grassland (Whittaker, 1960). Our results show that as the light regime increases in the plant community, the proportion of hemerophytes increases and the proportion of therophytes decreases. The hemerophytes in the temperate zone show a positive correlation with the light regime, while the inverse relationship was found for the therophytes (Shary et al., 2019).

The prospect of further research is to investigate the dependence of indicative reliability of the assessment of other environmental factors with the help of phytoindication depending on the number of species. In addition to the indication of traditional ecological factors it is of particular interest to clarify the question of the dynamics of hemeroby indicators as a result of park reconstruction.

Conclusion

The park reconstruction resulted in the increase in the number of photophilic herbaceous plant species. The phytoindication assessment of the light regime was an effective tool to assess the degree of transformation of the park plantation and the dynamics of vegetation cover after reconstruction. The proposed permutation test allowed us to demonstrate the possibility of using the phytoindication method at fine-scale spatial level, which is important for monitoring the state of green park plantations. The phytoindication led to homogenization of ecological conditions of particular parts of the park. The reliability of fine-scale assessments of phytoindication decreased with increase in species richness. The spatial organization of tree crowns was the leading factor in structuring of the herb layer in the untreated areas of the park. After reconstruction, the role of tree crowns decreased, which was due to the effect of delayed response of grass cover to abrupt changes in the state of the crown.

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