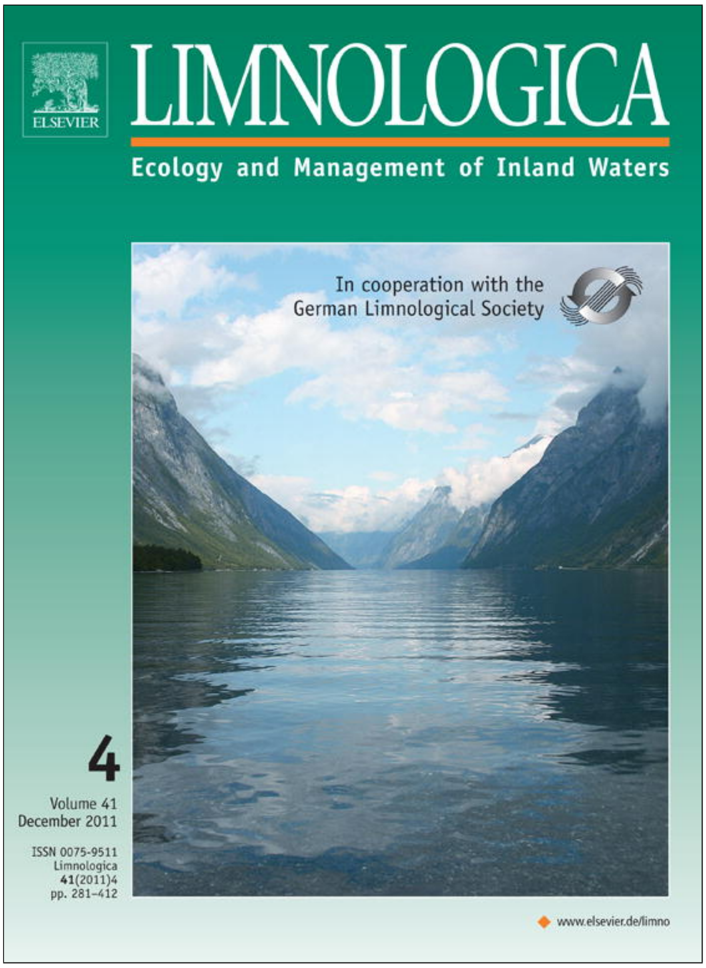
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Spatio-temporal changes in land cover and aquatic macrophytes of the Danube floodplain lake

Helena Ot'ahel'ová[a,*], Ján Ot'ahel'[b], Robert Pazúr[b], Richard Hrivnák[a], Milan Valachovič[a]

[a] Institute of Botany, Slovak Academy of Sciences, Dúbravská cesta 9, SK-845 23 Bratislava, Slovak Republic
[b] Institute of Geography, Slovak Academy of Sciences, Štefánikova 1, SK-814 73 Bratislava, Slovak Republic

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ABSTRACT

The aquatic vegetation of Číčov Lake in the Danube floodplain, which is listed in the Ramsar Convention, was investigated to address three main questions: (1) how have landscape composition and the structures of the lake and its buffer zone changed from the mid-20th century; (2) how have species richness and the abundance of the aquatic macrophyte assemblage in this lake ecosystem changed over the last 34 years; and (3) which landscape metrics can best explain these temporal changes for floating-leaved macrophytes? Two methodological approaches, remote sensing and botanical field surveys, were applied. Historical (1949, 1970, 1990) and contemporary (2006) aerial photographs were analysed to determine land cover. Landscape configuration and structure were analysed using eight landscape metrics selected in advance to measure spatio-temporal changes and the fragmentation of the lake ecosystem and its corresponding buffer zone. The species diversity, abundance and distribution of true aquatic macrophytes were surveyed eleven times in five survey stretches between 1973 and 2007.

At the landscape level, a decrease in the area covered by floating-leaved macrophytes, as well as an increase in open water surface and fragmentation of the land cover classes in the lake ecosystem, were recorded from 1949 to 2006. Overall, 30 true aquatic macrophytes were found from 1973 to 2007. Species richness did not change considerably, but the abundance of aquatic species fluctuated over the years. Three groups of true aquatic vegetation, based on common structural characteristics, were found in 1973–1983, 1989–2002, and 2004–2007 over the last 34 years. The landscape metrics NP, PD, LPI, and SHDI, which all express patterns of landscape fragmentation mostly indicate temporal changes in floating-leaved macrophytes.

Introduction

The reduction or deterioration of aquatic and semi-aquatic habitats arouses interest in their structure and functional processes. Historical data can provide a baseline to determine long-term ecological changes and clarify the spatial organisation and temporal succession of communities (Bravard et al., 1986; Dömötörfy et al., 2003; Sharov et al., 2009). Several studies have reported comparisons between the current species richness, abundance and distribution and historical levels of aquatic vegetation in lake ecosystem. These documented changes occurred mostly during the second half of the last century, in response to human impacts, mainly changes in water level (Egertson et al., 2004; Hellsten et al., 2006), eutrophication (Ozimek and Kowalczewski, 1984), the intensification of agriculture (Wade, 1999; Duigan et al., 2006), stocking of herbivorous fish (Hutorowicz and Dziedzic, 2008) and changes in land use along the lakeshores and catchments (Papastergiadou et al., 2007, 2008; Partanen et al., 2009).

To evaluate spatial distributions and historical changes in wetland and lake ecosystems, we considered remote sensing and GIS as the most suitable tools. Our assumption was supported by previous studies focusing on the application of remote sensing techniques in botanical research (Iverson, 1988; Remillard and Welch, 1992; Lehmann and Lachavanne, 1997; Janauer, 1997; Ot'ahel' et al., 1994; Valta-Hulkkonen et al., 2004; Zhu et al., 2007) and the analysis of land–water, or terrestrial–lake ecosystem interactions (Osborn and Kovacic, 1993; Kratz et al., 1997; Magnuson et al., 2005; Papastergiadou et al., 2008). The knowledge of successional changes in the aquatic vegetation, in the context of the adjacent landscape, is useful for determining the practical management of the floodplain. Aquatic plants are important components of the freshwater ecosystem. Using remote sensing methods, aquatic plants are usually identified to the level of growth forms: emergent helophytes (Papastergiadou et al., 2007) and floating-leaved hydrophytes (Dömötörfy et al., 2003). In general, the practical identification and assessment of submerged macrophytes is dif-

* Corresponding author. Tel.: +421 2 52968508; fax: +421 2 54771948.
E-mail address: helena.otahelova@savba.sk (H. Ot'ahel'ová).

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ficult and limited when using historical black and white aerial photographs, or colour orthophotographs in the case of fertile water-bodies (Silva et al., 2008). To obtain complete data for aquatic plants to the species level in the lake ecosystem, botanical surveys and remote sensing techniques should be combined (Partanen, 2007; Zhu et al., 2007).

A general trend of deterioration in aquatic vegetation has been recorded in the floodplain lake system along the Danube River. Baart et al. (2006) analysed the historical change in the aquatic plants and neophytes of the Lobau floodplain, within the city limits of Vienna, from around 1846, prior to regulations being put into place for the Danube. They found that the frequency and quantity of aquatic neophytes were rising. Degradation of macrophyte habitats is considered to be responsible for changes in species composition, distributions and densities. Changes in the dynamics of submerged vegetation during the two decades following 1980 included an approximate 50% reduction in open water in the Danube Delta of Romania, restructuring the primary producers by suppressing aquatic vegetation in some lakes and decreasing the abundance of emergent and floating species (Cristofor et al., 2003).

Several authors have previously studied the aquatic vegetation of the Slovak Danubian corridor (Ot'ahel'ová, 1980; Adamec et al., 1993; Ot'ahel'ová and Valachovič, 2002; Ot'ahel'ová et al., 2007). However, the identification of land–water, or terrestrial–lake, ecosystem interactions through remote sensing techniques has been quite scarce. This study was undertaken in an attempt to fill this gap through the identification of temporal changes in the structure of aquatic vegetation in relation to land cover in the buffer zone of Číčov Lake. This lake was selected due to its high status of protection, not only within Slovakia, but also as a part of the larger Inland Danube Delta in Europe. The accessibility of historical aerial photographs since 1949 and our own botanical data since 1973 motivated us to combine both approaches to address three questions: (1) how have landscape composition and the structures of the lake and its buffer zone changed from the mid-20th century; (2) how have species richness and abundance of the aquatic macrophyte assemblage in this lake ecosystem changed over the last 34 years; and (3) which landscape metrics can best explain these temporal changes for floating-leaved macrophytes?

Methods

Study area

This study covered an area on the left bank of the Danube River in southern Slovakia (1800 km along the river; 47°46′N; 17°43′E; Fig. 1) and represents a floodplain with a georelief that declines along the Danube River (115–111 m a.s.l.) and from the southern to the northern part of study area (115–110 m a.s.l). The flow of ground water is connected with the hydrological regime of the Danube River, generally from north-west to south-east; the air circulation is also dominant in this direction. Číčov Lake is a relic of the Danube anabranch system, the so-called the Middle Danube Inland Delta.

The total study area covers 228.6 ha, including the lake's area of 47.9 ha. The length of the shoreline of the lake is 11.7 km, and the mean water depth is approximately 2.5–3 m (7.5 m maximum). It is a groundwater-fed and back-flooded lowland lake system, with prevailingly stagnant water levels. Číčov Lake is rich in nutrients, characterised by high alkalinity and levels of nitrate, ammonium nitrogen and phosphate. Klaučo (1992) reported an array hydro-chemical data, obtained in November 1992: pH 7.5 ± 3.5; conductivity 366.0 ± 35.4 μS cm^{-1}; BOD 3.6 ± 1.2 mg l^{-1}; Na^+ 9.2 ± 1.6 mg l^{-1}; K^+ 2.5 ± 0.3 mg l^{-1}; NH_4^- 0.24 ± 0.28 mg l^{-1}; NO_3^{2-} 0.8 ± 0.2 mg l^{-1}; PO_4^{2-} 0.047 ± 0.017 mg l^{-1}; SO_4^{2-} 40.05 ± 2.51 mg l^{-1}; Cl^- 16.30 ± 0.53 mg l^{-1}; and HCO_3^- 162.19 ± 31.82 mg l^{-1} (average values calculated from ten sampling plots within Číčov Lake). Transparency values, determined using a Secchi disc, varied between 185 and 210 cm (July 2004).

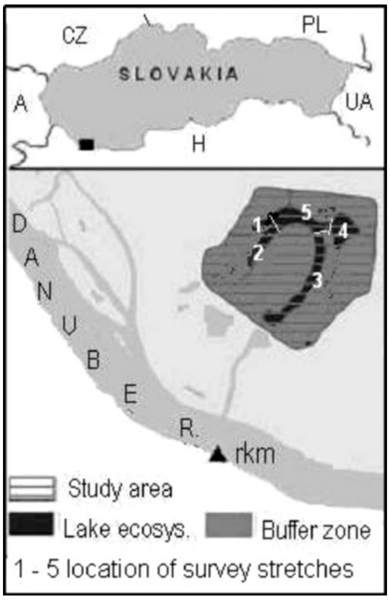

Fig. 1. Location of the study area.

Číčov Lake was disconnected from the river in 1903 when river regulations were put into place. It became a National Nature Reserve in 1964. Up until 1965, a weak hydrological connection existed between the lake and the inundation area via a discharge canal. The Danube experienced a large flood in 1965, which broke the levee and flooded the adjacent landscape (Gyalókay, 1982). The subsequently rebuilt levee impaired the superficial connection between the lake and the river. There is a long tradition of fisheries management in the region, and the lake was stocked with herbivorous grass carp (*Ctenopharyngodon idella* Val.) in 1985 in order to eliminate the aquatic vegetation (Nagy et al., 1992). Since 1990, it has been listed in the Ramsar Convention, an international treaty for the conservation and sustainable utilisation of the wetland. Subsequently, it was included in the extended Ramsar locality of Dunajské luhy in 1998.

Land cover data processing and landscape analysis

Land cover data were acquired using aerial photographs study area at scale of about 1:30,000 from the Topographic Institute of the Ministry of Defence of the Slovak Republic. The dates of the aerial photographs closest in date to the corresponding field surveys were used in this study. To identify and quantify land cover changes during the study period (1949–2006; Table 1), the scanned data were geometrically transformed to a Gauss-Krüger map projection, integrated through the ground control points and digitised in a GIS environment. A geometric correction was carried out using standard techniques (Schowengerdt, 1997; Mather, 2004). The delimitation of land cover-classes was based on Baker et al. (1979), Lillesand and Kiefer (1994), Ot'ahel' et al. (1994), Ihse (1995) and Cousins and Ihse (1998). In the land–lake interaction analysis, the land cover classes of the buffer zone and the land cover of the lake ecosystem were determined (Table 1). The nomenclature of the lake ecosystem classes was based on the growth forms of aquatic macrophytes (cf. Valta-Hulkkonen et al., 2003), and the categories of the buffer zone followed the CORINE land cover classification (Heymann et al., 1994; Bossard et al., 2000; Table 1).

Table 1
Land cover changes in Čičov Lake, interpreted from aerial photographs taken from1949 to 2006.

Period		1949	1970	1990	2006	1949/1970	1970/1990	1990/2006	1949/2006
Date of acquisition		26.7.	14.8.	20.6.	15.9.				
Aerial photographs		BW[a]	BW[a]	BW[a]	C[b]				
No.	Land cover class	Area in ha				Index of change in %			
Lake ecosystem									
1	Water surface	19.4	21.7	23.8	19.1	5.03	9.55	−19.61	−1.48
2	Floating leaved macrophytes	12.5	9.52	7.09	11.2	−10.6	−25.59	58.51	−10.31
3	Helophytes	13.9	13.5	13.3	12.6	−1.26	−1.42	−5.27	−8.98
4	Willow shrubs	2.85	2.84	3.54	4.96	−0.38	24.72	40.13	73.85
	Total	48.66	47.60	47.75	47.95				
Buffer zone									
5	Broad leaved forest	46.5	55.4	69.3	90.6	8.26	25.05	30.74	95.01
6	Transitional woodland shrubs	24.4	31.3	29.6	12.1	12.04	−5.21	−59.09	−50.38
7	Natural grassland	10.9	8.04	0.51	0	−11.51	−93.67	−100	−100
8	Arable land	28.1	60.9	33.6	33.1	50.82	−44.79	−1.47	17.95
9	Orchards	0	3.45	3.24	0	–	−6.08	−100	–
10	Pastures	18.3	13.4	35.1	33.2	−11.85	161.57	−5.48	80.96
11	Complex cultivation patterns	46.6	3.89	4.07	6.57	−40.13	4.58	61.45	−85.91
12	Dam	4.37	4.37	4.37	4.39	−0.15	0	0.39	0.39
13	Urban fabric	0.57	0.86	0.82	0.68	22.22	−4.86	−16.6	20.04
	Total	179.79	181.65	180.71	180.71				

[a] BW: black and white aerial photographs.
[b] C: colour aerial photographs.

Landscape changes were evaluated using ArcGIS 9.3 and Fragstat 3.3. Historical land cover maps of the floodplain lake and its buffer zone (shore distance from 100 to 435 m, prevailing 250 m; Fig. 2) were created from rectified aerial photographs. Patches of land cover classes were used to calculate spatial and temporal landscape composition and configuration in the context of patch characteristics (Forman, 1995) and related landscape indices. We used a set of landscape pattern metrics to quantify changes in the landscape and compare landscape compositions (Turner et al., 2001; Wu et al., 2002). Landscape structure was analysed according to the following landscape metrics: the total landscape area (TA), the number of patches (NP), patch density (PD), total edge lengths (TE), largest patch index (LPI), landscape shape index (LSI), mean patch area (AREA_MN), and Shannon's diversity index (SHDI). Basic definitions

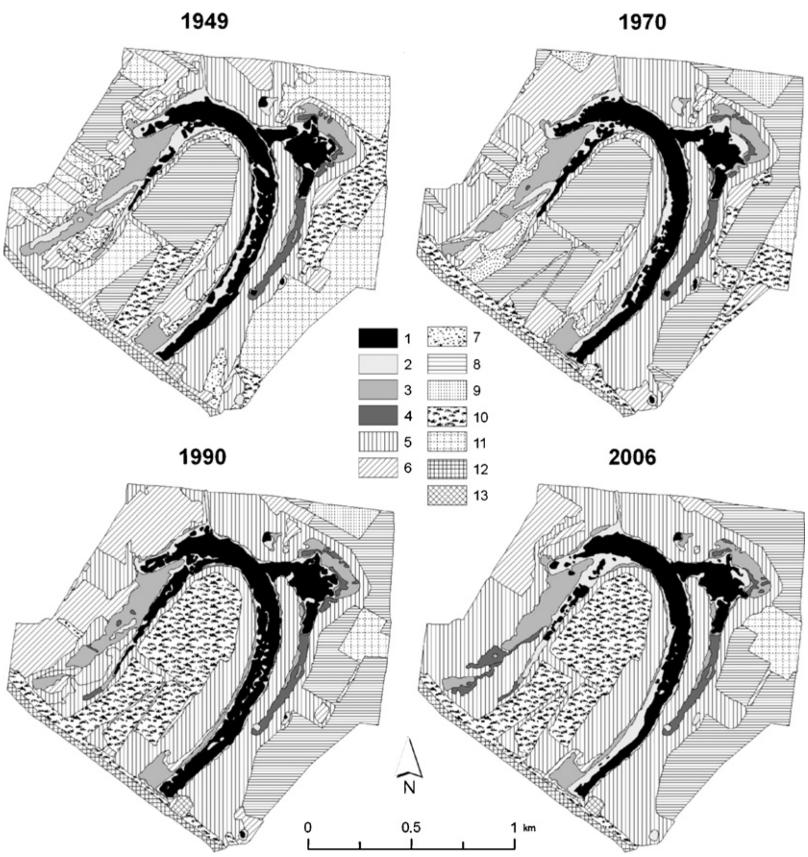

Fig. 2. Land cover maps for the years 1949, 1970, 1990, and 2006 (Captions No. 1–13 see Table 1).

Table 2
Definition of each landscape metric used, according to McGarigal and Marks (1995).

Metrics	Basic definition	Units
Total landscape area (TA)	Total class area	Hectares
Number of patches (NP)	Number of patches of the corresponding class	Number per 100 ha
Patch density (PD)	Density of patches of corresponding class	Meters
Total edge lengths (TE)	Lengths of all edge segment of corresponding class	None
Landscape shape index (LSI)	Standardized measure of total length of edge in the landscape	None
Largest patch index (LPI)	Study area comprised by the largest patch of corresponding class	Percent
Mean patch area (AREA_MN)	Average size of patches	Hectares
Shannon's diversity index (SHDI)	Measure of relative patch richness	Information

of each metric are given in Table 2; for full descriptions, refer to McGarigal and Marks (1995) or Wu et al. (2002).

Aquatic macrophytes: sampling procedure and data analysis

Field surveys of aquatic macrophytes were carried out (from a boat using a grapnel attached to a rope) eleven times over a period of 35 years (1973, 1976, 1979, 1981, 1983, 1989, 1991, 1998, 2002, 2004, and 2007) following a standard approach (Janauer, 2003; BS EN 14184, 2003). Five survey stretches were selected along the Číčov Lake (Fig. 1). In the same stretches aquatic macrophytes were sampled using a five-level estimator scale (1: rare, 2: occasional, 3: frequent, 4: abundant, 5: very abundant; Plant Mass Estimate PME; Kohler, 1978). PME data were transformed into "plant quantity" by applying the function $y = x^3$ (y = "plant quantity"; x = PME; Kohler and Janauer, 1995) and their numerical derivatives were calculated for each surveyed year. These derivatives are the Relative Plant Mass–RPM (percentage of "plant quantity" of each species weighted by the lake survey stretch length; Formula (1)) and the Mean Mass Total-MMT (index of mean PME of each species with regard to the full length of the survey stretches; Formula (2)).

$$\mathrm{RPM}\,[\%] = \frac{\sum_{i=1}^{n}(M^3 \cdot L_i)\cdot 100}{\sum_{j=1}^{k}\left[\sum_{i1}^{n}(M_{ji}^3 \cdot L_i)\right]} \quad (1)$$

where M_{ij} = estimated mass of the ith species in the jth survey stretch, and L_j = length of the jth survey stretch,

$$\mathrm{MMT} = \sqrt[3]{\frac{\sum_{i=1}^{n} M_i^3 \cdot L_i}{L}} \quad (2)$$

where M_{ij} = estimated mass of the ith species in the jth survey stretch, and L_j = length of the jth survey stretch, L = total length of the survey stretches.

The MMT index was used to calculate Shannon's index of species diversity and for the indirect ordination method. Principal component analysis (PCA) was performed using the CANOCO 4.5 for Windows package (Ter Braak and Šmilauer, 2002). The first gradient length in the detrended correspondence analysis was 1.138 SD units, indicating that a linear model of species response, such as that included in PCA, is appropriate for our data set. Species that appeared in only in one study year (*Lemna trisulca*, *Hippuris vulgaris*, *Persicaria amphibia* f. *natans*, *Potamogeton crispus*, and *P. pusillus*) were downweighted, and the species data were log-transformed. Temporal changes in the MMT for each detected macrophyte species were determined using the pairwise Spearman rank correlation coefficient, R_s, calculated in STATISTICA (StatSoft, 2001). The nomenclature of plant species used here follows that used by Marhold and Hindák (1998).

Results

Landscape composition

Over the period of 1949–2006, 13 land cover classes were identified in the study area: four in the lake ecosystem and nine in the buffer zone (Table 1; Fig. 2). When the changes in the areas of individual classes of the lake ecosystem were compared, the results showed that the total area of the lake diminished by 0.72 ha during the study period. The greatest decrease occurred in 1970, followed by a slight increase (Table 1). The areas of the floating-leaved macrophytes and open water surface changed the most, but relative changes were highest in willow shrubs. The direct increase in willow shrubs was at the expense of helophytes and was associated with the time gradient (Table 1; Fig. 3a).

The biggest changes in absolute value in the buffer zone were found in the broad leaved forest; its area has gradually doubled and, at the expense of the transitional woodland/shrubs in 2006. The biggest index of change was obtained for natural grasslands, which was changed by natural succession into transitional woodland shrubs or broad leaved forest (100%, Table 1). Pastures increased overall, but the changes varied over time. This class increased at the

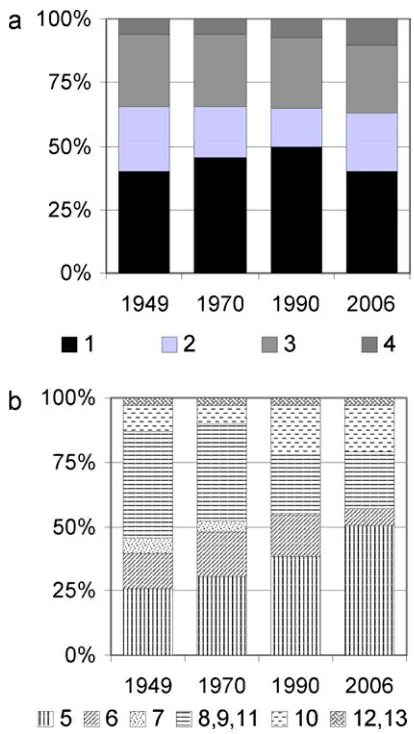

Fig. 3. Changes in the area (%) of land cover classes in the study area over the years 1949, 1970, 1990, and 2006, for the (a) Číčov Lake ecosystem and the (b) buffer zone (Captions no. 1–13 see Table 1).

Table 3
Changes of landscape metrics over 1949–2006 in the study area.

Landscape metrics	1949	1970	1990	2006
Lake ecosystem				
Total landscape area (TA) ha	48.66	47.60	47.75	47.95
Number of patches (NP)	99	190	224	108
Patch density (PD) *n*/100 ha	203.42	399.16	469.16	225.25
Largest patch index (LPI) %	35.70	45.12	48.90	39.11
Total edge length (TE) km	49.39	50.20	50.59	48.13
Landscape shape index (LSI)	17.70	18.19	18.31	17.38
Mean patch area (AREA_MN) ha	0.49	0.25	0.21	0.44
Shannon's diversity index (SHDI)	1.24	1.21	1.18	1.29
Buffer zone				
Total landscape area (TA) ha	179.79	181.65	180.771	180.71
Number of patches (NP)	66	56	37	11
Patch density (PD) *n*/100 ha	36.71	30.96	20.47	6.09
Largest patch index (LPI) %	11.58	12.90	19.98	27.89
Total edge length (TE) km	66.91	63.35	52.39	44.66
Landscape shape index (LSI)	12.47	11.78	9.75	8.31
Mean patch area (AREA_MN) ha	0.10	3.23	4.88	16.43
Shannon's diversity index (SHDI)	1.77	1.62	1.58	1.38

expense of arable land. Compared to 1949, complex cultivation patterns with dominating annual crops decreased considerably and, in 1970, changed to large-area arable land. The area of complex cultivation patterns was slightly increased in 1990 due to gardening. No distinct changes were identified in the other classes (Table 1; Fig. 3b).

Landscape structure

Changes in landscape structure for the years 1949, 1970, 1990, and 2006 are presented in Table 3. A significant increase in NP and PD of the lake ecosystem during 1970 and 1990 indicates a temporal increase in fragmentation, namely of floating-leaved macrophytes. An interesting side effect of the decrease and division of the areas covered with floating-leaved macrophytes (Fig. 4a) in 1949–1970 and 1970–1990 was an increase in the compactness of the water surface, as expressed in the LPI of the lake ecosystem. A significant increase in NP until 1990 had only a small effect on the SHDI, while its value was broadly affected by sensitivity to rare patch types (cf.

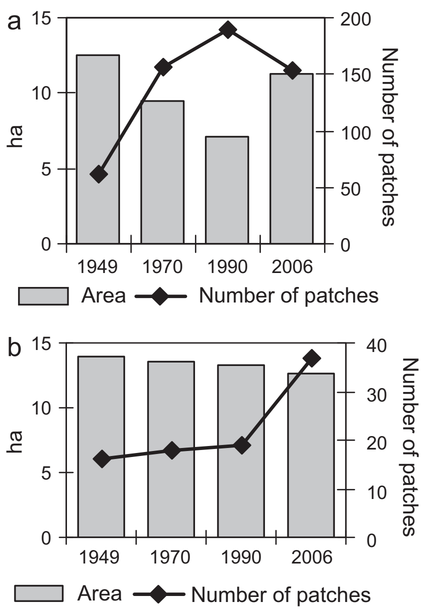

Fig. 4. Fragmentation patterns of (a) floating-leaved macrophytes and (b) helophytes, expressed as the size of the area and the number of patches in Čížov Lake, covering the period from 1949 to 2006.

McGarigal and Marks, 1995). Although the LSI and TE did not change significantly, important changes in the value of this metrics were found in the buffer zone, where all indices follow more general trends, confirming a decrease in the spatial diversity of the buffer zone. A dramatic decrease (six times less in 2006 compared to 1949) in NP was followed by a smoother decrease in PD, TE, LSI and SHDI. Homogenisation of the buffer zone areas also led to a doubling of the LPI in the study period. These landscape metrics point to a non-consistent trajectory of landscape patterns in the lake ecosystem and buffer zones.

Structural characteristics of aquatic macrophyte vegetation

Species diversity

Altogether, 30 true aquatic macrophyte species were recorded in Čížov Lake from 1973 to 2007 (detailed descriptions in Fig. 5). Species richness did not vary considerably (Average 16, and standard deviation SD ± 2); it only decreased from 1991 to 2002. The average value of Shannon's diversity index was 2.5 (SD ± 0.2); the highest value was found in the very first survey (Fig. 6).

Abundance and species richness

The total abundance of aquatic macrophytes, expressed as MMT, fluctuated moderately over the 34-year period (average 22.94, and SD ± 3.61). The biggest decrease was observed in 2002 (Fig. 6). *Myriophyllum spicatum* experienced the greatest fluctuation in MMT; it almost doubled with a transient decrease in 1989 and 1991. A similar tendency was observed in the cases of *Nymphaea alba* and *Ceratophyllum demersum*. In addition to *M. spicatum* ($R_S = 0.636$, $p < 0.05$), a positive statistical correlation with the time gradient was found for *Potamogeton nodosus* ($R_S = 0.661$). In contrast, a negative statistical correlation was found between the observation years and the MMT of *P. pectinatus* ($R_S = -0.777$), *Najas marina* ($R_S = -0.770$), *M. verticillatum* ($R_S = -0.725$), *P. lucens* ($R_S = -0.717$), *P. perfoliatus* ($R_S = -0.680$), and *Trapa natans* ($R_S = -0.683$). The ordination diagram (PCA), shows a gradual change over the years (Fig. 5). The most significant variability in the species data along the first PCA axis come from *P. lucens, N. marina, P. pectinatus, M. verticillatum, T. natans* with negative correlations and *M. spicatum* with positive correlation.

Relative plant mass

At the beginning of the monitoring of aquatic macrophytes, the variation in the size of the RPM patterns of the recorded species was relatively small over the whole lake area (Fig. 7). None of the 19 species present had an RPM > 16%, and most species occurred with less than 1%. However, considerable changes in RPM pattern size followed. In the last survey, the dominant species, *Myriophyllum spicatum*, reached an RPM of over 50%, while that of the subdominant *Nymphaea alba* was approximately 25%. The RPM of *Ceratophyllum demersum* was 15% and that of the 16 remaining species was less than 1%. *N. alba* was the dominant species in the lake in 1973–1983 and the value of RPM in the quoted period increased 2.5-fold. In 1989 and 1991, the values temporarily decreased (below 10%), but after 1998, they exceeded 20%. *Nuphar lutea* was subdominant at the beginning of 1973, but the RPM values gradually decreased by almost a half until 1983. This species became temporarily dominant in 1989 and 1991, but the RPM decreased below 10% by 1998, and, in the following years, it did not exceed 5%. The RPM of the species that were more frequent during the first half of monitoring time, such as *Najas marina, Trapa natans, Potamogeton lucens* and *M. verticillatum*, declined over the last 20 years.

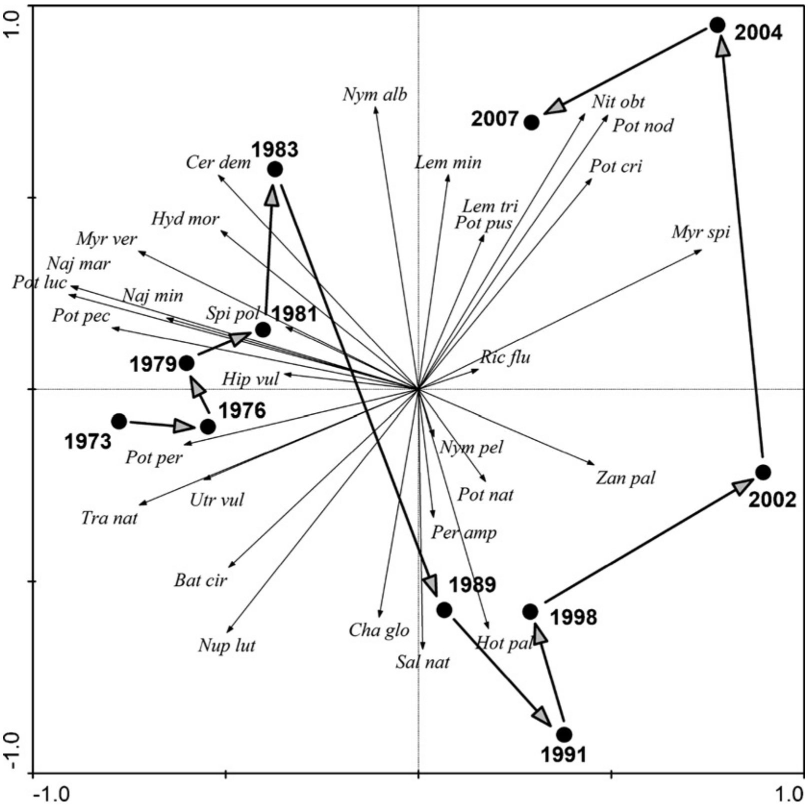

Fig. 5. PCA ordination diagram with species data displayed and the positions of samples (observation years). The first two axes explain 50.5% of the variation in the species data. Abbreviations of species: Bat cir, *Batrachium circinatum*; Cer dem, *Ceratophyllum demersum*; Cha glo, *Chara globularis*; Hip vul, *Hippuris vulgaris*; Hot pal, *Hottonia palustris*; Hyd mor, *Hydrocharis morsus-ranae*; Lem min, *Lemna minor*; Lem tri, *L. trisulca*; Myr spi, *Myriophyllum spicatum*; Myr ver, *M. verticillatum*; Naj mar, *Najas marina*; Naj min, *N. minor*; Nit obt, *Nitellopsis obtusa*; Nup lut, *Nuphar lutea*; Nym alb, *Nymphaea alba*; Nym pel, *Nymphoides peltata*; Per amp, *Persicaria amphibia* f. *natans*; Pot cri, *Potamogeton crispus*; Pot luc, *P. lucens*; Pot nat, *P. natans*; Pot nod, *P. nodosus*; Pot pec, *P. pectinatus*; Pot per, *P. perfoliatus*; Pot pus, *P. pusillus*; Ric flu, *Riccia fluitans*; Sal nat, *Salvinia natans*; Spi pol, *Spirodela polyrhiza*; Tra nat, *Trapa natans*; Utr vul, *Utricularia vulgaris*; Zan pal, *Zannichellia palustris*.

Discussion

In general, the relatively high degree of habitat variability in riverine ecosystems is responsible for their high biodiversity. Species diversity is a result of disturbance history, resource partitioning, habitat fragmentation and successional phenomena across the riverine landscape (Ward and Tockner, 2001). Anthropogenic influences on the hydrology of large rivers are mainly associated with regulation of river flow, including the modification of natural flow dynamics, dredging, channel straightening, bank stabilisation and the construction of artificial levees, which have disrupted the connectivity between ecological units of the floodplain/aquifer complex in many of the world's rivers (Ward et al., 1999). Hydrological connectivity between large rivers and the network of river channels was found to be important with respect to plant diversity (Bornette et al., 1998; Ot'ahel'ová et al., 2007). Moderate anthropogenic influences on hydrology, such as management, may result in a local enrichment of macrophyte productivity through the deposition of fertile sediment and the enhanced availability of dissolved nutrients (Barko and Smart, 1986; Lacoul and Freedman, 2006). Several studies have demonstrated that the structure of aquatic vegetation is correlated with the type of land use in the area (Hilli et al., 2007; Sass et al., 2010), including those from Danube river catchment area (Ot'ahel'ová et al., 2007).

In accordance with previous studies, we focused on changes in the structural characteristics of the floodplain lake ecosystem and the surrounding landscape over the last 50 years and changes in the assemblage of true aquatic macrophytes over the last 34 years. Except for a short-run flood in 1965, Číčov Lake has exhibited lentic conditions for approximately 100 years. Agricultural activities were reduced during the study period; however, herbivorous fish were stocked in the lake. Remote sensing and landscape metrics have helped us to explain the structure of the lake and its buffer zone in 1949, e.g., 46 years after the arm was disconnected. In the end of the 1940s the environments of Číčov Lake, with the fertile soils of the Podunajská Plain, were then exploited intensively by agriculture. Arable land prevailed and was most abundant in the complex cultivation pattern class, whereas forest consisted only of a narrow edge. Due to dominant air circulation patterns, relief inclination and the main direction of ground water flow the lake ecosystem was probably affected by fertiliser runoff from western part of the study area. We suggest that the most intensive impact of the terrestrification on the lake ecosystem took place before and during the period 1949–1970. The breaching of the levee by a flood of the Danube River in 1965 caused a significant disturbance of the lake ecosystem. The flood changed the shoreline, reduced the area of open water surface by 0.7 ha and formed a shore bank with terrestrial land cover classes in the western part of the lake. The spatial structure of floating-leaved macrophytes was changed distinctly by a decrease in area, and simultaneously, the fragmentation of its growth increased in 1970, according to the NP (190) and PD (399.16) indexes (Table 3; Fig. 4a). Contrary to flood disturbance shown by helophytes, no remarkable changes in structural characteristics were recorded in 1970 (Fig. 4b). This lack of change may be due to aerenchymatous helophytes, such as *Phragmites austaralis*, which are well adapted to flooded – anaerobic conditions (Lambers et al., 2008) or may be recovered under appropriate hydrological conditions (Dienst et al., 2004).

Botanical field surveys to the species level began in 1973. For interpretation of land–lake interactions, we used results from both

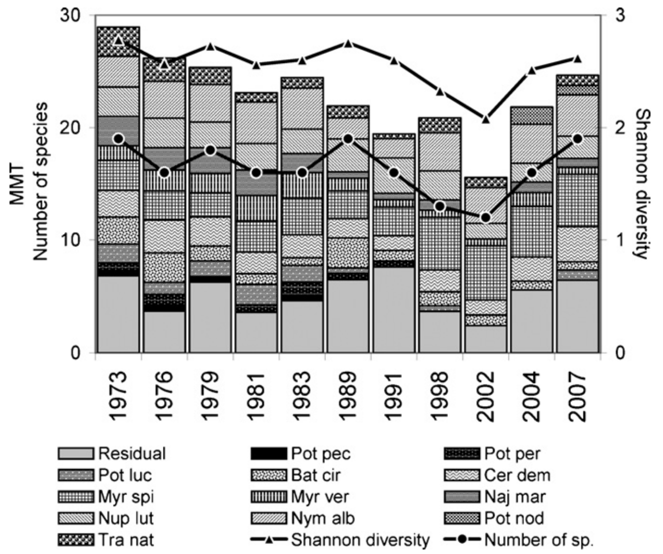

Fig. 6. Seasonal variation in species diversity and the abundance of aquatic macrophytes in Číčov Lake from 1973 to 2007 (for abbreviations of species, see Fig. 5; residual pattern includes minor abundant species).

methodological approaches in the followed period. Based on yearly changes in the structural characteristics of true aquatic vegetation over the last 34 years (Fig. 5), three groups were formed in the ordination space for years (A) 1973–1983, (B) 1989–2002, and (C) 2004–2007:

A: Čicov Lake had an average of 17 ± 1 species found in the first decade. Both Shannon's diversity index (2.64 ± 0.09) and the total abundance of true aquatic macrophytes (MMT = 25.61 ± 1.95) reached peak values. There were no noteworthy changes in macrophyte species richness and size abundances during the study period (Figs. 6 and 7). With the exception of *Nymphaea alba* (MMT = 3.3 ± 0.35), most species had a MMT value over 2 (*Myriophyllum spicatum*, *Nuphar lutea*, *Ceratophyllum demersum*, and *Najas marina*). *N. alba* was dominant during all survey periods. The intensity of land use in the buffer zone gradually decreased during the study periods. The area of the intensively used (ploughing) land cover classes (No. 8, 9, 11) decreased by about 10 ha in 1970 (Table 1; Fig. 3).

B: Changes in the structural characteristics of the aquatic macrophyte assemblage manifested in a reduction in the total number of species (15 ± 3), Shannon's index (2.44 ± 0.26), and total abundance (MMT = 19.46 ± 2.41). The biggest decrease of all metrics was found in 2002. During this period, increase of abundance of *M. spicatum* was recorded (average MMT = 3.61 ± 1.12). *N. lutea*, which was dominant in 1989 and 1991, was replaced by *M. spicatum* and later by *N. alba*, from 1998 until the last survey (Figs. 6 and 7). A dense, monospecific carpet of *M. spicatum* replaced the submerged species and invaded the bottom, which previously lacked vegetation, up to a depth of over 3 m. The abundance of submerged species, such as *C. demersum*, *M. verticillatum*, and *Najas* spp., decreased. A reduction in agricultural land use continued in the buffer zone in 1990. The area of the ploughing land cover classes (No. 8, 9, 11) decreased considerably by about 33.8 ha (Table 1; Fig. 3), and we identified an increase in areas covered by broad leaved forest (69.3 ha) and pastures (35.1 ha). Homogenisation of the buffer zone increased simultaneously according to the SHDI index (l.58, Table 3). The area of floating-leaved macrophytes was the lowest (7.09 ha) and exhibited the most fragmentation NP (224), PD (469.16) over the whole study period (Tables 2, 3). We saw no evidence of any impact of land use in the buffer zone on the change in the lake pattern. This change was caused by the other disturbances.

C: During the last monitored period, an increase in the total number species (16–19), Shannon's index (2.51–2.62), and the total abundance of true aquatic species (21.84–24.66) were recorded. This period could be considered a restoration period of lake environment, because number of species recorded in 2007 reached the number found in the first survey in 1973. However, the dominance of *M. spicatum* (RPM > 50%) in 2007 represents a change from the 1973 conditions. Subdominant *N. alba* and *C. demersum* achieved abundances similar to those in the first decade, in contrast to the decline of *N. lutea*. The intensity of agricultural land use (land cover classes No. 8, 9 and 11) decreased slightly (by about 1.24 ha) in 2006 because the orchards in the surrounding area were removed; however, the areas of complex cultivation pattern increased due to gardening. Agricultural land use and gardening were located in the eastern part of lake, but we do not expect there was nutrient runoff from these areas to the lake because the flow of ground water and air circulation are in opposite direction. Also, the extent of areas covered by broad leaved forests was the largest (90.6 ha, Table 1) and the landscape diversity of the buffer zone was the lowest in 2006, according to the SHDI index (1.38, Table 2). The area of floating-leaved macrophytes increased (11.2 ha) and their fragmentation decreased (NP = 108, PD = 25.25) in the lake ecosystem. These indices of landscape metrics from 2006 are very close to the indices of 1949. Simultaneous NP, PD, LPI and SHDI indices explained most of the relative temporal changes in the floating-leaved macrophytes.

Changes in the macrophyte patterns are possibly explained by environmental changes. As eutrophication is one of main factors that influence the assemblages of aquatic macrophytes in general (Wetzel, 1983), we first supposed that agricultural land use could explain the changes of structure of true aquatic vegetation. This hypothesis was not explicitly supported in our study. The observed changes in aquatic vegetation were not indicated to be in response to the reduction of ploughing land cover classes in the buffer zone (Table 1). Probably, the environment of the lake is naturally fertilised and nutrient availability is not a limiting factor for aquatic plants. Second, we attempted to explain temporal changes in the structural characteristics of the aquatic macrophyte assemblage as a result of management. One possible explanation for important changes recorded between A and B groups is that they may be a consequence of intensive stocking with the herbivorous grass carp (*Ctenopharyngodon idella*) in 1984 and 1985. Unfortunately, there are no quantitative data about the fish stock available. In 1993, the State Nature Conservancy of the Slovak Republic recommended fishing out this species but had little success (Szabóová, 1997). It is well known that grass carp reduces the biomass of select

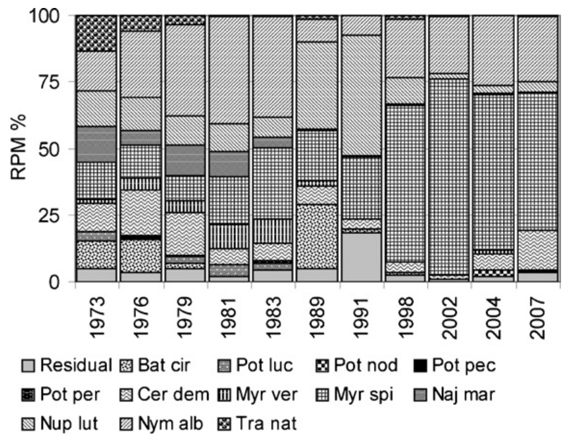

Fig. 7. Relative plant mass of aquatic macrophytes in Číčov Lake during 1973–2007 (for abbreviations of species, see Fig. 5; residual pattern includes minor abundant species).

aquatic plants by consuming them (van Zon, 1977). The efficacy of grass carp depends on several factors, such as water temperature, species composition and the stocking density of fish (Chaudhuri et al., 1976; Catarino et al., 1997). In general, grass carp usually live up to 5–11 years; some individuals can even reach 21 years of age (Shireman and Smith, 1983; Mandrak and Cudmore, 2004). The decrease in the total abundance of macrophytes in Číčov Lake was conspicuous even 17 years after grass carp stocking; however, the abundance of some species, in our case mainly *M. spicatum*, increased. Consequently, our findings concerning the changes in plant species composition are in accordance with those reported by Pípalová (2002), who studied the impact of the grass carp on aquatic vegetation in ponds of southern Czechia. She considered *M. spicatum*, *C. demersum,* and *Lemnaceae* species as unacceptable food for the grass carp. She also found that these species replaced other plants, which were dominant prior to carp stocking. Within the mentioned species, *M. spicatum* especially increased in occurrence in Číčov Lake. It is a cosmopolitan species, which grows rapidly, tolerates disturbance and stress and is a successful competitor (Nichols and Shaw, 1986). There is a general agreement in the literature about the rapid spread of *M. spicatum*, e.g. in North America or Great Britain (Boylen et al., 1999; Wade, 1999). The results of Agami and Waisel (2002) suggested that a competitive relationship exists between the submerged species *Najas marina* and *M. spicatum*, in accordance with our findings (Fig. 6). Plant competition has been recognised as an important factor determining the distribution of species in aquatic plant communities (Gopal and Goel, 1993; Wu and Yu, 2004). In spite of harvesting *M. spicatum* in a smaller part of the Číčov Lake in the autumn of 2002, its excessive spread was not prevented. In addition to management activities, the spread of *M. spicatum* in Číčov Lake is probably also favoured by suitable ecological conditions, such as the lack of the water level fluctuations and a nutrient-rich environment. The reduction of *N. lutea* and the increase *N. alba* are probably associated with the long-lasting lentic condition. *N. lutea* appears to be more tolerant of wave action and water movement than *N. alba* (Heslop-Harrison, 1955). A synergic effect, relatively stable hydrological conditions, the lake's morphometric context (Aznar et al., 2003; Léonard et al., 2008) and management are responsible for the structural features of the aquatic vegetation in Číčov Lake.

The combination of both methods allowed the careful examination of a floodplain lake in regard to the spatial distribution of floating-leaved macrophytes and the inventory of submerged species. The results from both approaches, interpreted through structural changes in the aquatic macrophyte community, pointed to the importance of the flood and management disturbances in the past. In addition, it is possible to reconstruct the vegetation of habitats, which were lacking in the historical field survey.

This study supports rating Číčov Lake as an aquatic habitat NATURA 2000. During the period of 1973–2007, 30 true aquatic macrophytes were identified. Among them, 15 are registered in the Red List of Slovakia. In the last survey, 11 species were confirmed directly in Číčov Lake; the others, except *Hippuris vulgaris*, were found in adjacent wetlands. Alien invaders, such as *Elodea canadensis* and *E. nuttallii*, which are rapidly spreading along the Danube River, have not yet been found in Číčov Lake.

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