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Spatiotemporal changes and drivers of trophic status over three decades in the largest shallow lake in Central Europe, Lake Balaton

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ABSTRACT

The over-enrichment of shallow lakes in nutrients has emerged as one of the main causes of water quality deterioration, and is today a major focus of water quality studies worldwide. In the present work, changes in trophic conditions over three decades (1985–2017) in the largest shallow freshwater lake in Central Europe, Lake Balaton, are assessed using the time series of 10 water quality variables measured at 4 sites, one in each basin of the lake. Using combined cluster and discriminant analyses, and assessing each of the four basins of the lake separately, it was possible to divide the history of the lake into three time intervals. Principal component and Sen's slope analyses highlight the fact that the oligotrophization of the lake took place at a different pace in each of these three major time intervals (1985–1994; 1995–2003; 2004–2017) along the lake's major axis. A significant decrease in the concentration of parameters indicating trophic conditions (e.g. chlorophyll-a, soluble reactive phosphorus) was first observed in the western basins, in the proximity of the main water input to the lake, followed by the eastward spread of this phenomenon. At the same time, the importance of external total phosphorus input to the lake was found to decrease eastwards, thereby diminishing its capacity to explain the variance of the water quality parameters in the lake. Over the time period covered by this study, various measures were taken to reduce the nutrient loads to the lake. These were, in the main, successful, as may be seen in the decade-by-decade overview of the lake's trophic state presented here. A brief review of similar cases from around the world only serves to reinforce the conclusion that a drastic reduction in external phosphorus loads arriving in similar shallow lakes will result in their oligotrophization, albeit with a time-lag of at least ten years.

1. Introduction

Nutrient over-enrichment deriving from intensive anthropogenic activity in the watersheds of lakes has emerged as one of the main causes of deterioration in water quality (e.g. Scheffer, 2013, Schindler, 1974, Schindler et al., 2016, Wetzel, 2001), leading eventually to the degradation of macrophyte vegetation, increased turbidity and, in extreme cases, anoxic conditions (Lau and Lane, 2002). The harmful effects of toxic cyanobacterial blooms endanger aquatic food production and supplies of water for recreation and drinking, leading, in turn, to economic losses, too.

In order to prevent the eutrophication of surface waters, inorganic nutrient inputs must be retained. Evidence shows that, of the inorganic nutrients, it is phosphorus (P) whose retention has the most beneficial effect on the trophic and ecological status of formerly eutrophic lakes (e.g. Sas, 1990; Schindler, 1974; Schindler et al., 2016; Vitousek et al., 2010). Neither can the role of N be neglected, since in estuaries or coastal environments it is a key factor (Carpenter, 2008), and excess reduction of trophic conditions has been achieved by managing not only P but N inputs as well (EPA, 2015). Nevertheless, interventions exclusively aimed at N loads will not lead to the desired oligotrophic states; this can only be achieved by reducing P as well (e.g. Carpenter,

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2008; Schindler et al., 2016; Welch, 2009). In spite of the fact that freshwater eutrophication has become a widespread problem over the past half-century and there have been many studies on how to prevent its harmful effects, globally, the number of toxic phytoplankton blooms has continued to increase (Ho et al., 2019; Hudnell, 2008).

A trophic classification of surface waters was first developed in the late 1960s in Europe (Rodhe, 1969), and further developed over succeeding decades. One of the most commonly used indices for the definition of the trophic state of lakes is the trophic state index (Carlson, 1977) relying primarily on the concentration of surface water chlorophyll-a, surface water total phosphorus concentration (TP) and the Secchi depth (Wen et al., 2019). Another widespread classification was formulated in the early '80s by the Organization for Economic Co-operation and Development (OECD), defining the classification of trophic status for freshwater lakes primarily on the basis of the concentration of TP and Chl-a in the water (Vollenweider and Kerekes, 1982). It is these parameters which still constitute the focus of more recently developed models for eutrophication (e.g. Markad et al., 2019, Wen et al., 2019). Therefore, the combined decadal assessment of these parameters is capable of yielding excess information on the effect of external measures aimed at shifting the trophic condition of lakes toward oligotrophization.

One of the most endangered ecosystems in this respect is shallow lakes, which are defined by being well mixed (that is, when subjected to an average wind velocity of 20 km h^{-1} for > 6 h they will mix through their water column (Chapman, 1996)), therefore besides their relatively large surface-to-depth ratio, they are characterized by intense lake-land, air-water and water-sediment interactions (Wetzel, 2001). These interactions render the eutrophication process and formation mechanisms of algal blooms particularly complicated (Qin et al., 2007); they also differ greatly between individual shallow lakes (Janssen et al., 2014). Examples of the adverse effects of algae blooms on shallow lakes have been reported all over the world, e.g. Asia (Qin et al., 2007); North America (López-López et al., 2016; Oberholster et al., 2006); Europe (Hatvani et al., 2014; Sebestyén et al., 2019); South America (Oliveira and Machado, 2013) and Africa (Muli, 1996).

When focusing on the eutrophication of shallow lakes, besides external nutrient loads, the resuspension-desorption of phosphorus from the sediment should also be taken into account, since it plays an important role in the overall nutrient dynamics of shallow lakes (Bloesch, 1995). Indeed, even in the case of reduced external nutrient loads, internal phosphorus load may prevent improvements in lake water quality. At high internal loading, TP concentrations may rise and phosphorus retention can be negative especially in summer (Hatvani et al., 2014; Søndergaard et al., 2003).

Lake Balaton, the largest (surface area 596 km^2) shallow (average water depth 3.2 m) freshwater lake in Central Europe (Fig. 1), has suffered from adverse anthropogenic effects over the last half century (see later and e.g. Hatvani et al., 2014, Padisák and Reynolds, 2003). The lake's watershed is approximately 5180 km^2 (Pomogyi, 1996), and it may be characterized as polymitic. The mean depth and surface area of the lake's geographical basins increases eastwards from 38 km^2 to 228 km^2, while their corresponding sub-watersheds decreases from 2750 km^2 to 249 km^2 (Istvánovics et al., 2007). The largest tributary, the River Zala, which enters the lake at its westernmost and smallest basin, Keszthely Basin (I. in Fig. 1), supplies ~50% of the lake's total water input and accounts for 35–40% of the lake's nutrient input (Istvánovics et al., 2007). The lake's only outflow is the Sió Canal, located at its easternmost end, and this was constructed in the nineteenth century to regulate the water level of the lake.

The accelerated anthropogenic activity (population growth, increasing waste water production, intensified use of fertilizers) in the catchment of Lake Balaton in the second half of the twentieth century resulted in a significant increase in external nutrient load (Hatvani et al., 2015) and a deterioration in the lake's water quality (Sebestyén et al., 2017), and by the end of the 1980s the P load carried by river Zala had doubled (Fig. 2a) compared to the beginning of the 1970s (Herodek et al., 1982; Istvánovics et al., 2007; Sagehashi et al., 2001). For this reason, a regional nutrient load control strategy was worked out for Lake Balaton (Somlyódy and van Straten, 1986), with the most important management measures being: (i) sewage diversion from the eastern and southern shoreline settlements; (ii) the construction of WWTPs in the western part of watershed; (iii) the downsizing of several large livestock farms (Hatvani et al., 2015); and (iv) the construction of the Kis-Balaton Water Protection System (KBWPS) (Hatvani et al., 2011; Kovács et al., 2010; Kovács et al., 2012a), the aim of which was the retention of nutrient loads borought by the Zala River which would have otherwise ended up in Lake Balaton directly; for further details see e.g. Clement et al., 1998; Hatvani, 2014; Hatvani et al., 2014; Hatvani et al., 2015. The combination of these measures and the ten-fold drop in fertilizer usage in the late 1980s (Hatvani et al., 2015) resulted in a TP load reduction of more than 50% compared to the 1980s (Hatvani, 2014). Nevertheless, the oligotrophization of Lake Balaton – and especially its easternmost basin – occurred, albeit with a delay, due to the presence of internal P loads from its sediment (Istvánovics et al., 2004).

As is the case with many temperate shallow lakes, primary production in Lake Balaton was considered to be P limited (Herodek, 1984), while, studies in past decades had focused on the importance of external vs. internal N loads (Présing et al., 2001; Présing et al., 2008). The role of P and N in algal biomass growth was investigated in a way similar to that employed by Schindler (1974) in the Experimental Lakes Area, and it was found that with an increased external P load, algal biomass grew, while N inputs increased the abundance of N-fixing cyanobacteria (Istvánovics et al., 1986). Thus, the more severe limitation of phytoplankton production by P, as compared to that caused by N is also acknowledged in the case of Lake Balaton (Istvánovics and Herodek, 1995; Istvánovics et al., 1986), though it should be recognized that N is found to limit primary production under extreme circumstances, e.g. an abrupt increase in algal biomass (Présing et al., 2008).

With regard to the internal P loads of Lake Balaton, it was found that their maxima are determined by the long-term behavior of the highly calcareous sediment. In the years when the internal P load approaches its maximum, a strong correlation can be observed between the biomass of phytoplankton and the estimated concentration of mobile P, under the influence of the carbonate content of the sediment (Istvánovics, 1988). Otherwise, the biomass of phytoplankton is kept below the highest possible level by physical constraints which depend on hydrometeorological conditions (Hatvani et al., 2014). Consequently, because of this delayed response in lakes (Sas, 1990) it is sometimes hard to find a direct correlation between external load reduction and water quality improvement, particularly over short time periods. In the case of Lake Balaton, thanks to the conscious efforts of the authorities, eutrophication has been successfully managed (Istvánovics et al., 2007). In spite of the global increase in intense lake phytoplankton blooming since the 1980s, in a worldwide study, Lake Balaton remains one of only six lakes, fewer than 10% of the total of 71, which exhibited an internationally acknowledged and statistically significant decrease in trophic status (Ho et al., 2019).

This certainly held true until late summer 2019, when, at the end of August, most of the lake became unexpectedly hypereutrophic due to the bloom of the flagellate *Ceratium furcoides* (Levander) Langhans 1925 and the blue-green *Aphanizomenon flosaquae* Ralfs ex Bornet & Flahault 1886. While both had previously been present in the lake (Padisák, 1985), their blooms had never before produced such an adverse effect. The 2019 bloom first occurred in the Keszthely Basin and spread to the Szigliget Basin, and then to almost the whole lake. The causes are still unclear, but it is obvious that this phenomenon was not to have been expected from the observed trends and the expectations which had formed as a result of the previous success of load reduction measures in and around the lake (Istvánovics et al., 2007). It is suspected that increased temperatures and specific hydrometeorological conditions

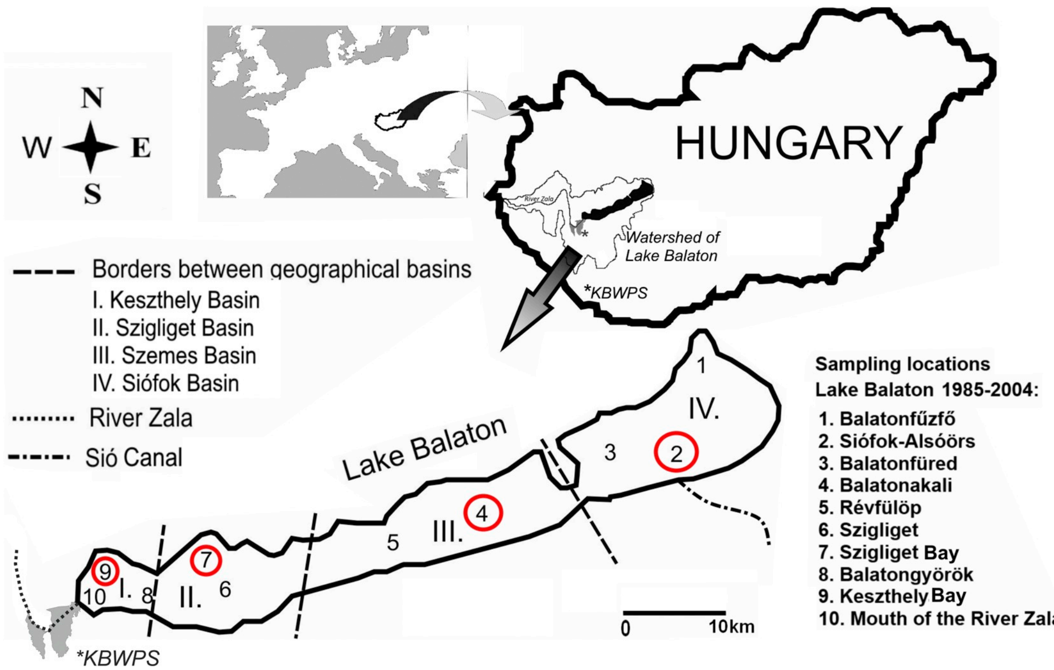

Fig. 1. Lake Balaton, its geographical basins and the 10 sampling sites operated up to 2005 by the responsible water authority. In addition, the Kis-Balaton Water Protection System (KBWPS) and Lake Balaton's watershed is marked on the outline map of Hungary. The sampling sites marked with a red circle (those used in the present work) and site 10 were operating after 2005. The figure is based on that in (Kovács et al., 2012b). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

(high water level, etc.) were the cause of this particular bloom (Istvánovics, 2019), while increasing temperatures are set to become an ever-more important factor in eutrophication worldwide (Ho et al., 2019). From a lake management perspective, long term studies of nutrients and their limiting role in primary production are of undoubtedly great importance, since these account for the effects of loading on the natural succession of phytoplankton communities (Istvánovics et al., 1986).

This research aims to offer a long-term overview and an update on changes in the trophic status of Lake Balaton over the last three decades, from the perspective of the multivariate data assessment of its water quality variables. The specific goals are to explore the spatio-temporal changes in the trophic status of Lake Balaton by (i) exploring whether distinct time periods can be distinguished in the history of the lake, (ii) determining the robust trends in parameters indicating trophic status within these time periods in its different basins, and (iii) assessing the change in the importance of external phosphorus loads on general water quality, including trophic indicators.

2. Materials and methods

2.1. Sampling sites and acquired dataset

In the course of the research 10 water quality parameters were selected for analysis. These were measured bi-weekly/monthly at four sampling sites between 1985 and 2017 (that is, the last year for which overall data for the Lake were available), one from each geographical basin of the lake (Fig. 1), a total of approximately 5600 data. The data were acquired from the Central Transdanubian Water and Environmental Inspectorate, and had been collected as a part of the National Water Quality Monitoring System.

Due to the occasionally insufficient number of samples and/or their values being below the level of detection (LOD), the set of parameters analyzed was restricted to the following: soluble reactive phosphorus (SRP), total phosphorous (TP), chlorophyll-a (Chl-a; mg l^{-1}), electrical conductivity (EC; μS cm^{-1}), ammonium – N (NH_4 – N), dissolved oxygen (DO; biological oxygen demand (BOD), chemical oxygen demand (COD), water temperature ($T_{W;}$ °C) and pH. These parameters were chosen so as to provide continuous temporal coverage over the whole of the investigated time interval as they were consistently measured using the same methods and at the same locations. It should be noted that in 2005 the monitoring was spatially recalibrated (Kovács et al., 2012b). In relation to this point, because of these changes, most forms of N had to be omitted from the evaluation. Total nitrogen was not recorded up to 2004, while in 80% of cases, the values of nitrate-N after the recalibration of the monitoring network in 2005 were below LOD. Thus, neither could have been incorporated into the study. However, data concerning ammonium-nitrogen were available for the complete period, and it is this form which is in any case the preferred N form for algal N uptake in Lake Balaton (Présing et al., 2001), as indeed in other lakes, also (Mitamura et al., 1995).

2.2. Methodology

After preprocessing the data (outlier detection and filtering of typos), its descriptive statistics (mean, median, coefficient of variation etc. following Kovács et al., 2012c) were calculated basin by basin to obtain an overview of the dataset. The next step was to use the available indicator variables to shed light on decadal changes in the trophic status of the various basins of the lake following the standard OECD classification (Vollenweider and Kerekes, 1982), which is the most widely accepted worldwide (Istvánovics, 2009). This classification estimates the trophic status of a water body primarily using information on the concentration of the limiting nutrients (TP) and a proxy for phytoplankton biomass (Chl-a). It had been previously used in the case of Lake Balaton (e.g. Crossetti et al., 2013; Istvánovics et al., 2007) and in numerous studies worldwide (e.g. Cloutier and Sanchez, 2007; Marsden, 1989), thereby ensuring the comparability of the results of the present study on a global scale. The question of whether time intervals with common patterns exist in the water quality parameters (WQP) time series was investigated by using combined cluster and discriminant analysis (CCDA Kovács et al., 2014; Section 2.2.1) on the

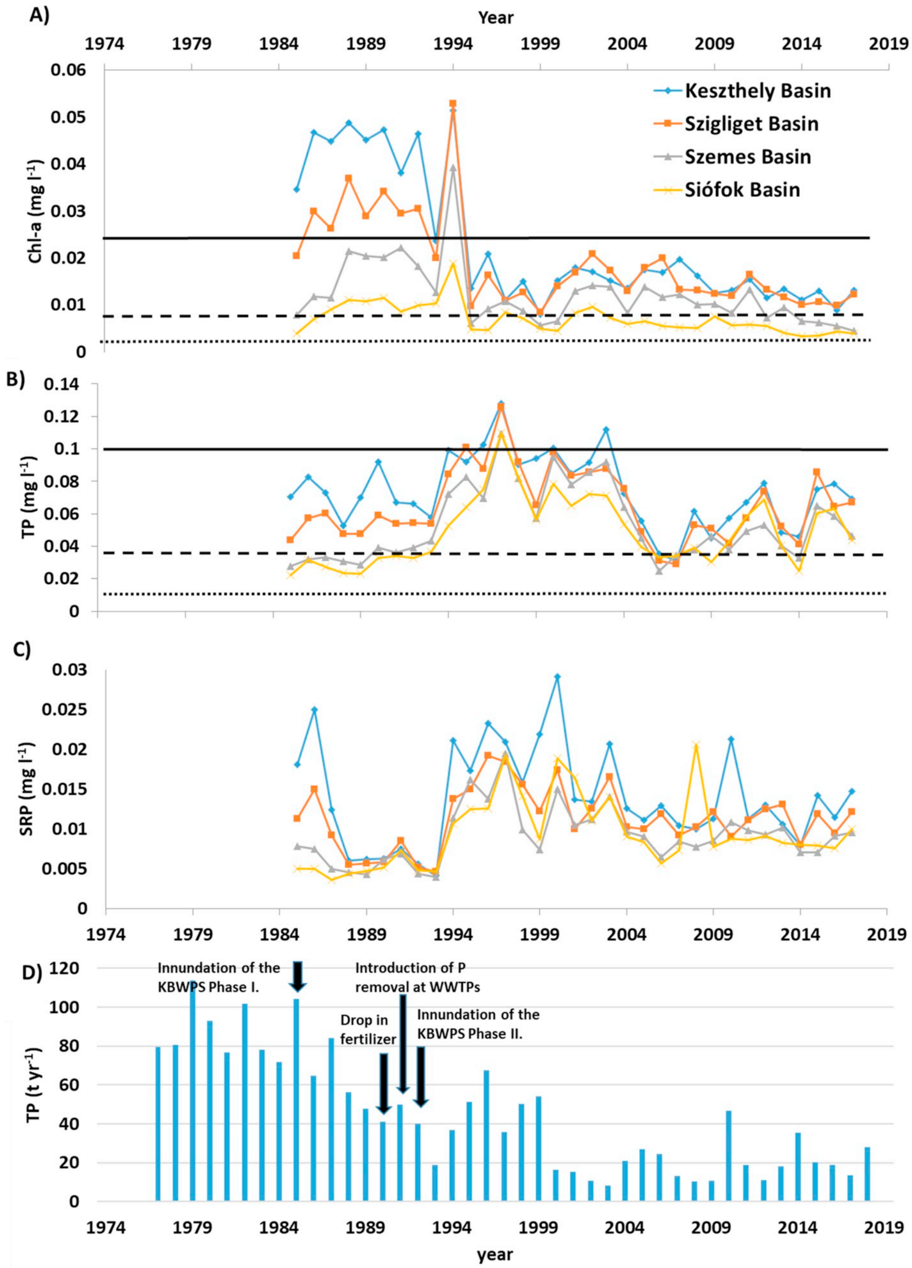

Fig. 2. Annual average concentration of Chl-a A) TP B) and SRP C) for the four investigated basins of Lake Balaton along with the inflow of annual TP loads from the River Zala through the KBWPS to Lake Balaton (sampling location 10 on Fig. 1) D) (redrawn and extended from (Hatvani et al., 2014) C) (1985–2017). In panels A) and B), above the continuous horizontal black line hypertrophic conditions prevail. The dashed line is the threshold for eutrophic conditions, and the dotted for mesotrophic. Below the dotted line oligotrophic conditions prevail, as determined by using the scheme of Vollenweider and Kerekes, (1982).

annual averages of the WQPs. Next, exploratory principal component analysis (PCA; Section 2.2.2) was conducted on the annual averages of the variables in the different basins to explore whether gradually changing common trends might be found as the distance from the lake's main input, the River Zala, increased.

Lastly, the magnitude of change in the concentrations of Chl-a and the P forms was explored for the whole lake basin by basin, using the nonparametric Mann– Kendall test (Kendall, 1975; Mann, 1945) and Sen's slope estimates (Sen, 1968) for the previously obtained differing time intervals using the monthly averaged concentrations of the parameters as the input. In each time interval the annual change in the parameters' values – obtained from the estimated Sen's slope - was given as a percentage of the average concentration of the given parameter in the investigated time interval and basin.

2.2.1. Combined cluster and discriminant analysis

Combined cluster and discriminant analysis (CCDA) is a multivariate data analysis method (Kovács et al., 2014) which aims to find not only similar, but even homogeneous groups in measurement data of known origin (so, in this work, to identify groups of water quality sampling sites). CCDA consists of three main steps: (I) a basic grouping procedure, in this case using hierarchical cluster analysis (HCA), to determine possible groupings; (II) a core cycle in which the goodness of the groupings from Step I and the goodness of random classifications are determined using linear discriminant analysis; and (III) a final evaluation step in which a decision concerning the further iterative investigation of sub-groups is taken. If the ratio of correctly classified cases for a grouping (“ratio”) is higher than at least 95% of the ratios for the random classifications (“q95”), i.e. the difference d = ratio–q95 is positive, then at a 5% level of significance, the given classification is not homogeneous in CCDA. Therefore, the division into sub-groups (Step III) and the iterative investigation of these sub-groups for homogeneity is required.

2.2.2. Exploratory principal component analysis

Exploratory principal component analysis (PCA; Tabachnick and Fidell, 2014) was used to find the variables with the greatest influence on the water quality status of the lake over the investigated time period. PCA decomposes the original dependent variables into principal components that explain the original total variance of the dataset component-wise in a monotonically decreasing order. The correlation coefficients between the original parameters and the principal components (PCs) are the factor loadings, and these explain the weights of the PCs in the original parameters, while the PC time series are referred to as PC scores (Olsen et al., 2012; Tabachnick and Fidell, 2014).

In the present case, the input variables for PCA were the annual averages of complete cases (1985–2017) of the WQPs (Sect. 2.1). The Kaiser-Meyer-Olkin (KMO) test (Cerny and Kaiser, 1977) was employed to determine the measure of sampling adequacy (MSA), providing information which allows the decision of whether PCA can be applied to the dataset. The variables with factor loadings outside the ± 0.7 interval were taken as important, while the PCs were taken into account based on their scree plots as suggested by Cattell, (1966) and their eigenvalues, which had to be above 1 according to Kaiser, (1960).

Principal component time series are commonly related to possible explanatory parameters in space (e.g. Magyar et al., 2013; Olsen et al., 2012) and time (e.g. Çamdevýren et al., 2005; Hatvani et al., 2018; Page et al., 2012). In the case of the latter, when correlating the explanatory parameters with the PC scores, the serial correlation of the data should to be considered, since it limits the number of independent observations, not satisfying the assumptions of conventional statistical methods (Macias-Fauria et al., 2012). Thus, in the present study one thousand Monte-Carlo simulations were performed with frequency (Ebisuzaki) domain time series modelling to obtain the correct significance levels of the correlation coefficients (r).

The analysis described in Section 2.2.1 and 2.2.2 was conducted on the annual averages of the complete cases of data considering the lake as a single water body, as well as the four basins of the lake separately. The former approach was justified by the water body designation criteria of the Water Framework Directive (EC, 2000) given that Lake Balaton is one single water body, while latter was justified by the study of Kovács et al. (2012b) highlighting the separate behavior of the four basins of the lake.

R statistical environment was used (R Core Team, 2019) to calculate the descriptive statistics, the Sen's slope estimates with Mann-Kendall tests using the mannKen () function of the wql package (Jassby and Cloern, 2017), and CCDA using the CCDA package (Kovács et al., 2014). The Wilks' λ statistics and PCA were computed using IBM SPSS 26, the statistical significance of the correlation coefficients under serial correlations were calculated using the Windows version of the software provided in Macias-Fauria et al., (2012), and additional tasks were performed in MS Excel 360.

3. Results

To provide an overall picture of the dataset, its descriptive statistics were produced (Table 1). The first quartile and median of the phosphorous forms, Chl-a, BOD and COD were highest in the Keszthely Basin, decreasing continuously with distance from the inlet of the River Zala (Table 1). Conductivity, on the other hand, showed a continuous increase along the main transect of the lake from W to E with respect to medians (from 630 to 666 μS cm^{-1}); this was also true of its first quartile, too (from 680 to 730 μS cm^{-1}). With respect to pH and water temperature, no noteworthy pattern was observed. In general, the highest degree of variability relative to the average in water quality was observed in the Keszthely Basin. However, TP (CV = 85.3) and chl-a (CV = 117.1) were most variable in the Szemes Basin, and orthophosphate in the Siófok Basin.

3.1. Trophic state of Lake Balaton

The relatively high TP and Chl-a values observed throughout the whole investigated time period (1985–2017) made necessary the thorough exploration of the temporal change in the concentration of the indicator variables of trophic status in the lake (Chl-a (Fig. 2a), TP annual mean values (Fig. 2b)) and annual external TP loads arriving in Lake Balaton (Fig. 2d).

It became clear that with regard to the average Chl-a concentration of the whole lake and the Keszthely Basin, 1994 was a turning-point. The average figures for the concentration of Chl-a prior to 1994 were 0.0264 mg l^{-1} and 0.0426 mg l^{-1}, for the lake and the basin, respectively; these figures then dropped by ~60 and ~ 65% in the lake and in the Keszthely Basin, as well as falling below hypereutrophic levels in all the basins (Fig. 2a). Interestingly, in the Szigliget Basin (Fig. 1), phytoplankton biomass in certain years of the early 2000s (e.g. 2002, 2003 and 2005, 2006) and afterwards in 2011 and 2012, was slightly higher and/or comparable to those in the Keszthely Basin. The lowest values for Chl-a were always characteristic of the Siófok Basin (0.008 mg l^{-1}), the furthest from the main external source of TP (Fig. 2d), the mouth of the River Zala.

In almost all cases, TP and SRP values were highest in Keszthely Basin, with the exception of six of the 33 years investigated in the case of the former (Fig. 2b), and four in the latter (Fig. 2c), in which slightly higher values were observed in the neighboring Szigliget Basin. Up to 1994 an increase in TP concentrations characterizes the system, with a parallel decrease in SRP (Fig. 2b,c). Afterwards, TP peaks in 1997 (~0.13 mg l^{-1}) and SRP in, 2000 (~0.03 mg l^{-1}). In the meanwhile, in the Keszthely Basin after 1997 TP concentrations decreased overall by ~50%, to 0.0670 mg l^{-1}, a drop which was even larger in the other basins (Fig. 2b). A classification based on annual averages in the main qualifies the lake's water as eutrophic, and only in the mid-1990s did this become hypereutrophic, as observable from the data of individual basins as well (Fig. 2).

The annual external TP loads arriving in Lake Balaton via the waters of the River Zala for the most part display a continuous decrease, due to the combination of measures taken to reduce nutrient loads in the region (see Sect. 1). However, in the mid-1990s a sudden increase was observed in TP loads reaching Keszthely Bay, while after 2000 the average loads were about 50% of those in the 1990s, with only a couple of years, the first being 2004 (Fig. 2d), with elevated loads due to hydrometeorological conditions (Hatvani et al., 2014). For example, in the region, in 2010 and 2014 the annual precipitation was the sixth and tenth highest in the period 1901 and 2014, amounting to > 875 mm (Jakuschné Kocsis and Anda, 2018; Kocsis et al., 2017).

It should be noted, that the TP loads arriving to Lake Balaton from the River Zala, through the KBWPS (Fig. 2d) resemble the SRP concentrations in Keszthely Basin (Fig. 2c) much more than any other parameter in any of the basins. It was empirically assessed, that after a continuous decrease from 1985, both SRP in Keszthely Basin and TP loads of the River Zala reach a decadal minimum in 1993 (Fig. 2c,d), unlike TP (Fig. 2b). Afterwards, fluctuating in a similar manner, both show peak in 2010, not characteristic of any other basin, or water quality parameter.

Primary indicator variables of trophic status were used to determine thetrophic state of the lake annually (Table A1). Regarding Chl-a, up to 1994 the Keszthely and Szigliget basins were mostly hypertrophic, while the westernmost basins were in most cases eutrophic and in the mid-1980s mesotrophic (Fig. 2a; Table A1) with the exception of 1994, when all the basins were eutrophic/hypertrophic in terms of Chl-a. After 1994 - which would appear to be a tipping point for Chl-a - all the basins except Siófok were mostly eutrophic. In the Siófok Basin, after 1994 a mesotrophic state obtained, which turned exclusive after 2003

Table 1
Descriptive statistics of the water quality parameters in the different basins of Lake Balaton (1985–2017), where M: mean, MED: median, SD: standard deviation, Q1: first quartile, Q3: third quartile; CV: coefficient of variation.

Statistic		Water T (°C)	pH	Cond (μS cm^{-1})	DO (mg l^{-1})	BOD (mg l^{-1})	COD (mg l^{-1})	NH_4-N (mg l^{-1})	TP (mg l^{-1})	SRP (mg l^{-1})	Chl-a (mg l^{-1})
Keszthely Basin	M	16.06	8.49	628.74	10.52	3.01	29.27	0.046	0.079	0.014	0.026
	MED	17.3	8.49	630	10.3	2.55	27	0.03	0.07	0.011	0.016
	SD	7.53	0.21	65.12	2.05	1.64	8.66	0.039	0.046	0.014	0.029
	Q1	9.75	8.37	580	9.16	1.9	23	0.02	0.05	0.007	0.009
	Q3	22.15	8.6	680	11.7	3.9	33	0.05	0.098	0.016	0.031
	CV%	46.9	2.5	10.3	19.5	54.5	29.5	84.5	58.7	95.4	114.8
Szigliget Basin	M	15.54	8.51	636.95	10.43	2.58	25.79	0.046	0.069	0.012	0.021
	MED	16.8	8.5	634	10.2	2.3	24	0.031	0.058	0.009	0.014
	SD	7.39	0.19	63.1	1.82	1.16	6.38	0.037	0.051	0.011	0.023
	Q1	9.4	8.4	585	9.12	1.7	21	0.02	0.04	0.006	0.009
	Q3	21.6	8.62	690	11.7	3.3	28	0.05	0.082	0.013	0.023
	CV%	47.5	2.2	9.9	17.4	45.1	24.7	80.7	73.5	92.7	112.9
Szemes Basin	M	15.36	8.51	644.55	10.35	2.25	22.61	0.041	0.056	0.009	0.013
	MED	16.7	8.52	638.5	10.1	2	22	0.03	0.042	0.007	0.009
	SD	7.36	0.17	63.15	1.89	1.06	4.9	0.033	0.048	0.008	0.016
	Q1	9	8.4	594	9	1.5	19	0.02	0.03	0.005	0.006
	Q3	21.55	8.62	700	11.54	2.8	25	0.05	0.066	0.012	0.015
	CV%	47.9	2.1	9.7	18.2	47.3	21.6	79.5	85.3	86.3	117.1
Siófok Basin	M	14.93	8.52	670.71	10.3	1.99	20.12	0.04	0.05	0.009	0.008
	MED	16.2	8.53	666	10.1	1.8	19	0.03	0.04	0.007	0.006
	SD	7.26	0.16	68.11	1.85	0.94	4.1	0.029	0.038	0.011	0.007
	Q1	8.85	8.4	620	9	1.3	18	0.02	0.027	0.003	0.004
	Q3	21	8.63	730	11.6	2.4	22	0.05	0.061	0.011	0.009
	CV%	48.6	1.9	10.1	17.9	47.3	20.4	72.9	75.3	113.3	92

in terms of Chl-a mean values (Fig. 2a; Table A1).

With regard to TP, the Keszthely and Szigliget basins were mostly eutrophic through the years analyzed, while the Siófok and Szemes basins were in the beginning mesotrophic, later turning eutrophic. The only exception involving all the basins is 1997, in which all of them were hypereutrophic in terms of TP (Fig. 2b; Table 2).

Overall, a clear pattern is visible, with a spatial divide between the western and eastern basins. The western basins (Keszthely and Szigliget) are mostly hypereutrophic at the beginning of the investigated time period (from 1985 to 1994), while the eastern ones (Szemes and Siófok) are eutrophic/mesotrophic. Moving forward in time, this pattern changes to eutrophic for the western basins and dominantly mesotrophic for the eastern basins, with oligotrophic characteristics (in Chl-a maxima) occurring in the mid-2010s in the westernmost, the Siófok Basin (Fig. 2a; Table A1).

3.2. Similarly behaving time intervals and temporal trends of primary trophic indicators

With the use of CCDA on the tagged annual averages of the water quality variables, three similarly behaving ("optimal") time intervals were determined as the basic grouping (Fig. 3). At the q = 95 level the biggest difference was 41.2% (Fig. 3), and this split the data into three time intervals: 1985–1994, 1995–2003 and 2004–2017, with the year 1993 not being part of any continuous time interval.

The greatest difference between the basic grouping and a random grouping (Fig. 3: curve d) was observed at the division of the data into three groups (Fig. 3), indicating that this is therefore the optimal grouping. These groups were further divided for the sake of verification, to the point at which all the years become separate, thus demonstrating that homogeneity can only be reached if the separate years form temporal groups alone. However, the three time intervals determined (with similarly behaving years) were objectively determined, and a metric assigned to their existence.

In accordance with the results of the CCDA, it was necessary to investigate the trends for Chl-a and TP as the main WQPs indicating the trophic state of the different basins in each individual basin in turn, and in addition SRP in the three time intervals: 1985–1994, 1995–2003 and 2004–2017 (Table 2). Sen's slopes were determined for Chl-a and P forms, and their annual change relative to the period mean concentration (Mp), representing a significant or insignificant long-term change, was derived (Table 2).

In the first time-period (1985–1994), Chl-a and SRP showed a mostly significant decrease in the western Keszthely and Szigliget basins (Fig. 2a,c), while in the other basins, a significant increase (Fig. 2a,b) and stagnant behavior (Fig. 2c) were observed for Chl-a, TP and SRP, respectively (Table 2). It should be noted here that, although there is a decrease in Chl-a in the Keszthely Basin, the average value is almost 1.5 times higher than in the neighboring Szigliget Basin, and more than twice and almost four times higher than in the eastern basins (Table 2).

The greatest change in all investigated periods and basins was the significant ($p < .01$) decrease of SRP by approx. -7.5% yr^{-1} relative to the period (1985–1994) mean (1.11×10^{-2} mg l^{-1}; Table 2) in Keszthely Basin, resulting in a total 8.2×10^{-3}mg l^{-1} drop in SRP.

Between 1995 and 2003, in the western basins the investigated parameters did not change significantly, although SRP decreased. In the eastern basins, however, Chl-a increased significantly ($p < .05$) by ~4% per year compared to the mean (Mp 1995–2003), while SRP decreased insignificantly (Table 2).

In the last investigated period (2004–2017), SRP concentrations did not show any significant change (Fig. 2c), while Chl-a decreased significantly in all basins (Fig. 2a; between approx. -2 to −4% per year), except Keszthely (Table 2). TP showed a minor (2.5%), but significant ($p < .05$) increase in the Keszthely and Siófok basins (Table 2). The observed trends are all in accordance with the data presented in Fig. 2a-c.

Overall, there is a significant ($p < .01$) decrease in SRP in the western basins in the first of the years investigated, and as this trend weakens over the decades, by the 2010s a decrease in biologically available P and Chl-a presents itself in the eastern basins, although to a degree as yet insignificant (Table 2). It should be noted here that in the Szemes Basin, the −2% annual SRP decrease is significant at $\alpha = 0.1$. It should be further noted that while the trends indicate the change in concentrations within the three distinct water quality time periods, the concentrations of Chl-a should also be considered in the light of the overall change as well (Fig. 2a). This shows a large drop in concentrations, for example, in the Keszthely Basin Chl-a mean

Table 2
Changes in concentration of major trophic indicator parameters in the three time intervals, obtained using CCDA on the different basins. The significance of the Sen's slopes determined by Mann-Kendall tests is indicated by three *** or two asterisks ** for $\alpha = 0.01$ or 0.05. In any given time period, Mp stands for the mean concentration value. Relative change to the Mp represents the average annual change of the WQP in percentages relative to the Mp of the given time period.

WQP	Keszthely Basin			Szigliget Basin			Szemes Basin			Siófok Basin			Interval
	Chl-a	TP	SRP	Chl-a	TP	SRP	Chl-a	TP	SRP	Chl-a	TP	SRP	
Period mean (Mp)	3.94×10^{-2}	7.38×10^{-2}	1.11×10^{-2}	2.88×10^{-2}	5.65×10^{-2}	8.27×10^{-3}	1.75×10^{-2}	3.86×10^{-2}	6.09×10^{-3}	1.01×10^{-2}	3.1×10^{-2}	5.40×10^{-3}	1985–1994
Relative change to the Mp	−3.8%***	0.0%	−7.4%***	−0.5%	2.1%	−3.2%**	3.2%***	5.9%***	0.0%	4.8%***	4.1%***	0.0%**	
Period mean (Mp)	1.45×10^{-2}	9.95×10^{-2}	1.89×10^{-2}	1.40×10^{-2}	9.28×10^{-2}	1.56×10^{-2}	9.80×10^{-3}	8.43×10^{-2}	1.30×10^{-2}	6.58×10^{-3}	7.55×10^{-2}	1.40×10^{-2}	1995–2003
Relative change to the Mp	0.9%	0.7%	−2.7%	2.4%	−0.8%	−3.0%	4.0%**	1.3%	−2.8%	4.5%**	0.0%	−1.9%	
Period mean (Mp)	1.40×10^{-2}	5.92×10^{-2}	1.22×10^{-2}	1.33×10^{-2}	5.43×10^{-2}	1.01×10^{-2}	9.08×10^{-3}	4.40×10^{-2}	7.72×10^{-3}	4.93×10^{-3}	4.32×10^{-2}	8.05×10^{-3}	2004–2017
Relative change to the Mp	−1.8%	2.6%***	0.0%	−2.4%**	3.2%	0.0%	−4.4%***	1.8%	−2.0%	−3.8%***	2.4%**	−1.3%	

concentrations decreased by ~65% between the period averages of 1985–1994 and 2004–2017 (Table 2).

3.3. Common patterns in the general water quality of Lake Balaton (1985–2017)

Except for the Szigliget Basin, the eigenvalues of the first 3 PCs reached a value of 1, and their cumulative explanatory power fell between ~70 and ~ 80% (Table 3). In the Szigliget Basin, the cumulative variance explained by the first 2 PCs was ~65%. Nevertheless, the explanatory power of the first PC in all the basins was between ~40 and ~ 50%. It is clear that if just the first two PCs are considered for the sake of comparison, their cumulative explanatory power decreases from 68.3% to 57.5% as we move in an easterly direction. According to the KMO test, the MSA yielded values which fell between what may be considered acceptable (> 0.7) and mediocre (> 0.6) in the different basins, thus demonstrating that PCA can provide reliable results.

While the parameters related to biological activity indicating primary production (Chl-a) and its impact on the indices of saprobity (DO, BOD, COD, NH_4-N) had the highest loadings in the first PCs, the phosphorus forms (TP and SRP) were most important in the second. The latter continuously lost its importance as one moves eastwards, with their average loading decreasing continuously from 0.88 to 0.71, while in terms of the loadings of TP in the Szemes and Siófok basins, these fell beneath the 0.7 threshold (Table 3). As for Chl-a, the other main indicator of trophic conditions, its highest loading was observed in the westernmost basin, in the Keszthely Basin (loading of 0.91 in PC1), while in the easternmost basin, at Siófok, it does not even reach the 0.7 threshold in either PCs (Table 3).

The PC scores and the time series of the external TP loads (Fig. 2c) for 1985–2017 arriving to Lake Balaton in the waters of the River Zala (sampling location 10 in Fig. 1) were correlated. It was found that the TP loads – as an external driving parameter – correlated significantly with the first PCs ($r > 0.65$), which were recognized as indicating primary production, rather than with the PCs mainly driven by inorganic nutrients (e.g. P forms; Table 3). The TP loads explain ~60% of the variance of PC1, and this decreases eastwards to ~40% in the Szemes Basin. In the Siófok Basin, the significant linear relationship between the TP loads of the River Zala reaching the lake through the KBWPS and the first PC strengthens. However, in this particular basin the representativity of parameters related to primary production decreased in the first PC (Table 3).

In theory, another viable approach would have been to explore the common patterns/trends in the different basins for the three time periods separately; unfortunately, the matrices serving as the input for PCA were singular, and the KMO test indicated that this particular sub-setting of the dataset would therefore be unsuitable for PCA, with an MSA of < 0.5.

4. Discussion

The spatiotemporal development of the trophic status of Lake Balaton over three recent decades was primarily determined by the complex interplay of its natural internal and external nutrient loads (discussed in Section 4.1), as well as the measures taken to reduce these (Section 4.2). These together make the case of Lake Balaton a unique international example of the result of drastic external load reduction measures in order to ameliorate oligotrophization, and have the potential for application worldwide (Section 4.3).

4.1. Factors behind observed changes and trends

From a spatial perspective, it has been demonstrated that the nutrient content of the water decreases from west to east, resulting in a lower trophic status in the eastern basins (Fig. 2; Table A1). This is due tomorphological reasons: the (i) increasing size of the watershed of the

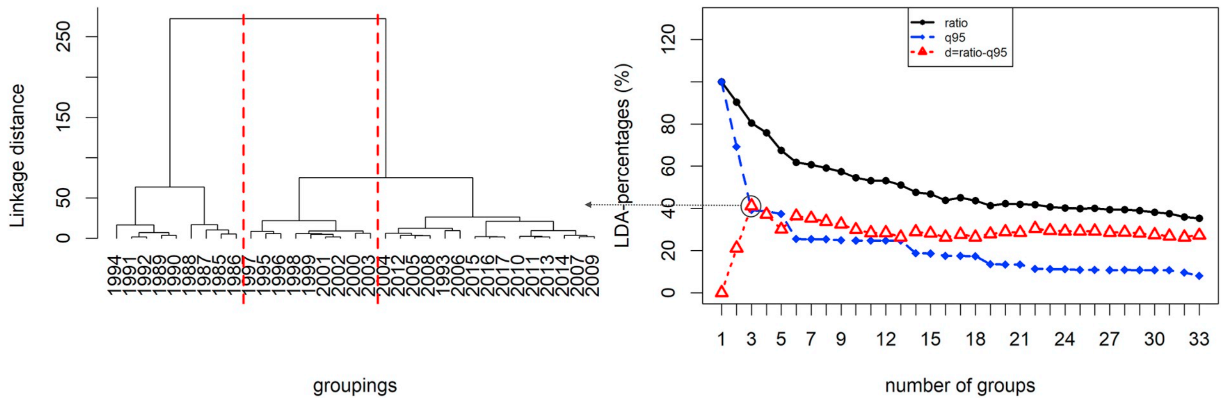

Fig. 3. Dendrogram representing the basic grouping from the initial run of CCDA (left) and the summarized results of CCDA for groupings. Right panel: ratio of correctly classified cases vs. the random classification and the difference values (d).

Table 3
Principal components of the water quality parameters in the different basins of Lake Balaton. Loadings outside the ± 0.7 interval are highlighted in bold. The percentage of original variance explained by the PCs can be found in the penultimate row, and the correlation coefficients of the PC scores and the TP loads of sampling location 10, representing the River Zala arriving to the lake through the KBWPS, in the bottom row. Coefficients of determination significant at $\alpha = 0.01$ or $\alpha = 0.05$ are marked with three ***, or two asterisks ** respectively, taking serial correlation into account.

	Keszthely Basin			Szigliget Basin		Szemes Basin			Siófok Basin		
WQP/PC	PC1	PC2	PC3	PC1	PC2	PC1	PC2	PC3	PC1	PC2	PC3
WT	−0.55	−0.3	0.45	−0.54	−0.51	−0.69	−0.1	0.3	−0.66	−0.36	0.39
pH	−0.3	−0.44	**−0.73**	−0.43	−0.2	−0.38	0.4	0.67	−0.66	0.3	0.34
Cond	**−0.83**	−0.19	0.27	**−0.76**	−0.21	**−0.78**	0.03	0.3	−0.68	0.08	0.26
DO	**0.8**	−0.17	0.04	0.69	0.05	0.43	0.52	−0.08	0.66	0.01	−0.47
BOD	**0.95**	−0.06	0.03	**0.9**	−0.03	0.68	0.47	0.35	**0.76**	0.17	0.28
COD	**0.94**	−0.08	0.03	**0.92**	0.03	0.68	0.49	0.14	0.27	**0.72**	0.37
NH_4-N	**0.76**	−0.03	0.4	**0.81**	−0.06	**0.77**	−0.01	−0.26	**0.8**	0.07	0.08
TP	0.07	**0.85**	−0.25	−0.19	**0.87**	0.57	−0.67	0.4	−0.42	0.66	−0.48
SRP	−0.05	**0.88**	0.08	−0.3	**0.82**	0.53	**−0.75**	0.24	−0.51	**0.71**	−0.25
Chl-a	**0.91**	−0.2	−0.14	**0.82**	−0.13	**0.86**	0.03	0.28	0.61	0.43	0.44
Explained variance	49.30%	19.00%	10.50%	46.50%	18.00%	42.70%	19.10%	11.30%	38.50%	19.00%	12.70%
TP loads (r^2)	0.58**	0.05	0.01	0.48***	0.03	0.42***	0.00	0.18**	0.50***	0.03	0.08

basins, the increasing residence time of the water (Istvanovics, 2002), (ii) decreasing area-specific nutrient loads eastwards (Istvánovics et al., 2007); and (iii) the increasing distance from the mouth of the River Zala, which brings ~50% of the lake's total water and 35–40% of its total nutrient input (Istvánovics et al., 2007) and 90–95% of the nutrient input of the Keszthely Basin (Istvánovics et al., 2004). In the case of the latter, the various interventions intended to reduce its nutrient loads were also a significant factor (see Section 4.2). Taken together, these factors resulted in the decreasing gradient of P and Chl-a content (Tables 1 and A1), influenced to an ever-decreasing extent by the TP input of the River Zala as one heads east on the lake, away from Keszthely, dropping to ~40% (Table 3). Sediment resuspension in Lake Balaton is much higher than in other shallow lakes, moreover, since its carrying capacity is P-determined, its internal loads are of major importance, i.e. the internal P loads can reach the magnitude of external SRP loads in dry years, e.g. early 2000s (Istvánovics et al., 2004), when external loads are smaller than the multidecadal average (Fig. 2d). With the increase in the water's residence time and the previously-discussed eastward gradual decrease in internal and external nutrient loads, the importance of algal biomass in explaining the water quality variability of the lake decreased (Table 3).

From a temporal perspective, although the measures taken (see Section 1) may be considered as the primary factor in the oligotrophization of the lake (Hatvani et al., 2015; Istvánovics et al., 2007), the local hydrometeorological conditions (e.g. precipitation, runoff, temperature, wind) had a measurable effect on its water quality (Hatvani et al., 2014; Istvánovics et al., 2004), as well as, in the form of such phenomena as local temperature, cloud cover, etc., on the water quality of the Kis-Balaton Water Protection System (Hatvani et al., 2017), which serves as a pre-treatment wetland for the loads arriving with the waters of the River Zala (Tátrai et al., 2000). The separation of the three time periods in the history of the lake coincides with major interventions to reduce its loads, as well as larger-scale economic changes (Section 1). However, in certain cases, e.g. the dividing line of 1994/'95, the lake responded to the drop in nutrient loads (Hatvani et al., 2015) with a time-lag; for details, see Section 4.2.

In addition, the reason for the unprecedented behavior of the year 1993 seems likely to be the dry conditions then prevailing (Hatvani et al., 2014), since this was the fifth driest spring of the region in the twentieth century (the spring precipitation amount was only 78.6 mm (Kocsis et al., 2020)). This resulted in decreased external loads, and thus a significant drop in P and Chl-a concentrations throughout the lake (Fig. 2a-c; Table 2). This was also the time of the lowest average SRP concentration (1993 annual avg. 0.0043 mg l^{-1}) in the lake between 1985 and 2017; additionally, this was also the point at which the smallest difference between the basins (max-min SRP in 1993: 0.0007 mg l^{-1}; Fig. 2c) was observable. These conditions were accompanied by a lack of large algae blooms, such as that which occurred in the following year 1994 (Istvánovics et al., 2007).

4.2. The results of the interventions taken to reduce trophic conditions

The significant decreasing trend (Sen's slope on annual averages; $p = 6.83 \times 10^{-5}$) in external TP loads from the watershed of the River Zala (1985–2017; Fig. 2d) was not followed by a direct monotonic decrease in P concentration in the lake as a whole, nor its continuous oligotrophization (Fig. 2b, Table A1, respectively). By way of contrast, SRP mirrored the decrease in external loads to a much greater extent (Fig. 2c,d; Table 2). This may be explained by the following facts:

(i) the lake responds to external load reduction measures (e.g. the inundation of the KBWPS, sewage diversion, P precipitation at WWTPs from the area, see Sect. 1) with a time-lag (Istvanovics, 2002), as in the case of other shallow lakes (Sas, 1990).
(ii) the KBWPS is capable of removing a higher ratio of nutrients from an already elevated input load, while contrariwise, if the loads are reduced, its efficiency also decreases (Clement et al., 1998), leading to a concomitant increase in the importance of internal loads from resuspension-desorption.

The interventions made at the mouth of the River Zala had primarily a local effect, while investments in the development of the waste water infrastructure (Hajnal and Padisák, 2008; Istvanovics, 2002; Istvánovics et al., 2007), and the significant decrease in the use of fertilizers (Sisák, 1993) in the Balaton watershed played an essential role in the decrease of the external nutrient loads in the 30 years covered by this study. The loads dropped by ~75% between 1977 and 1984 and 2004–2018 to ~21 t yr^{-1} in the Keszthely Basin (Fig. 2d), while in the other basins in this figure was already around 50% less as of 2002 (Istvánovics et al., 2007).

The lagged response to the drop in external loads was not only visible in time, but in each basin taken individually, as seen from the Sen's trend estimates (Table 2). While in the Keszthely Basin, the reduction in external loads had an almost immediate effect, in the eastern basins the decreasing trends in P and Chl-a occurred ~10 years later (Table A1). For example, in the Szigliget, Szemes and Siófok basins, only after 2004 did Chl-a start to decrease significantly ($p < .01$), and while reactive phosphorus concentrations stop decreasing in the western basins by the 2000s, at this point these start to show a decreasing trend in the east (Table 2). Findings from the Keszthely Basin indicated a change its trophic conditions from an initial steadily hypertrophic period (1981–1984), to a transient state of hysteresis (1985–1992) (Scheffer, 2013), which concluded in an alternative, less eutrophic state from the mid-1990s (Hatvani, 2014) where most of the eutrophication processes moved upstream to the KBWPS (Hatvani et al., 2014). However, it was from the Keszthely Basin that the unexpected algae bloom of late summer 2019 spread (Istvánovics, 2019), and this demands the further investigation of the specific triggers of this event. The present update on the trophic changes of the lake extends the previous coverage of the thorough analyses on the trophic status of the lake (Istvánovics et al., 2007; Tátrai et al., 2000) by 14 years. It underlines that its trophic state has indeed decreased. Moreover, such updates are crucial, because, at present information on the most recent (2019) algae blooms cannot be immediately provided.

4.3. Global trends in phosphorus load reductions and oligotrophization

Half a century ago, one of the first major reviews dealing with hundreds of studies on all scales was published, and it concluded that the increase in nutrients such as phosphorus and nitrogen originating from external sources are the most likely causes of the eutrophication of lakes (Vollenweider, 1970). At the beginning of the 1970s, lake experiments provided evidence that the reduction of P input is the most effective tool in the reduction of the trophic state and achievement of oligotrophization (Schindler et al., 2016).

There are a number of examples – including that of Lake Balaton – of situations in which efforts to reduce external P loads have resulted in lower TP and Chl-a concentrations and the eventual oligotrophization of a lake's waters (Jeppesen et al., 2005). Recovery has generally been delayed by the internal load, which is in turn dependent on the long-term behavior of sediments (Marsden, 1989; Sas, 1990; Søndergaard et al., 1999). According to these case studies and reviews, in most lakes a new equilibrium was reached after 10–15 years, a period of elapsed time only marginally influenced by the hydraulic retention time of the lakes. With the decrease in TP concentrations, SRP also declined substantially (Jeppesen et al., 2005).

In the case of Lake Balaton, the improvement in water quality was fastest in the Keszthely Basin, which stands in stark contrast to the delayed change in the eastern basins, a difference due to the specific morphometric features of the lake (Istvanovics, 2002). In accordance with the general experience that very large changes in external TP loading were necessary to change the trophic status of a lake (Marsden, 1989), this delayed, but still surprisingly fast recovery was achieved by an external load reduction of around 75% compared to the input when the lake was hypertrophic. As has been observed, "the OECD supports the suggestion that a large reduction in external P loading is necessary to change the trophic status of a lake: a reduction in the annual mean Chl-a concentration across a trophic category requires an approximately 80% reduction in external TP loading" (Vollenweider and Kerekes, 1982). But it is obvious that there is a substantial variation in the load - response relationships of various lakes (Marsden, 1989), and their recovery after nutrient load reduction may be significantly modified by environmental changes such as global warming (Ho et al., 2019), since the effects of global change are likely to run counter to reductions in nutrient loading rather than reinforcing re-oligotrophization (Jeppesen et al., 2005). Also, it is expected that recovery from eutrophication will be more difficult in shallow lakes (Rolighed et al., 2016), and therefore further efforts are needed to arrive at an estimate of the degree of nutrient reduction likely to be required in a future, warmer climate to mitigate eutrophication.

5. Conclusions

The present study provides a 14-year overall update compared to the landmark study of Istvánovics et al. (2007) on the changes and drivers of the trophic status of the largest shallow freshwater lake in Central Europe, Lake Balaton. It highlights the fact that the oligotrophization of the lake took place at a different pace – as indicated by Sen's trend analysis – in the three major time intervals (1985–1994; 1995–2003; 2004–2017) identified in the history of the lake, and what is more, in space along its major axis. At first around the turn of the 1990s the significant decrease in both algal biomass and biologically available phosphorus was observed in the western basins, those in closest proximity to the main water input to the lake, and afterwards spreading east. The stochastic analyses of the linear interrelations of the water quality parameters and the main external P input to the lake, further nuanced this picture. Those showed a continuous decrease in importance of inorganic nutrients (e.g. P forms) driving the general variance of water quality in the lake toward the eastern basins. The overall results indicated that the extent of oligotrophization depended on (i) hydromorphological conditions (ii) the external load reduction measures (e.g. inundation of the lake's pre-reservoir the KBWPS, reduction in fertilizer usage in the watershed, sewage treatment, etc.) of the late 1980s and the 1990s and (iii) local meteorological/basin conditions (e.g. temperature, resuspension of P from the sediment and desorption of SRP).

The findings, in comparison to international case studies highlight the fact that only with the severe reduction of external nutrient loads, and especially in the case of phosphorus, can the oligotrophization of such shallow lakes be achieved. However, due to sediment resuspension, this will occur only with at least a 5–10-year lag in response to the measures taken.

Author contributions

Conceived and designed the study: IGH, JK. Performed the analysis: IGH, VDB, PT, ISzK. Produced the figures: IGH, VDB, PT. Assessed the results: IGH, VDB, AC, JK. Wrote the paper with contributions from all

authors: IGH, AC, VDB. Revised the paper with contributions from all co-authors: IGH and AC. We applied the FLAE approach for the sequence of authors; see https://doi.org/10.1371/journal.pbio.0050018.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Appendix

Table A1
Annual means and maxima of Chl-a and TP (mg l^{-1}) for the four basins of Lake Balaton (1985–2017) with their corresponding trophic states marked according to the OECD classification (Vollenweider and Kerekes, 1982). Red: hypertrophic; orange: eutrophic; blue: mesotrophic; green: oligotrophic.

Year	Keszthely Basin Chl-a mean	Chl-a max	TP	Szigliget Basin Chl-a mean	Chl-a max	TP	Szemes Basin Chl-a mean	Chl-a max	TP	Siofók Basin Chl-a mean	Chl-a max	TP
1985	0.035	0.099	0.071	0.020	0.052	0.044	0.008	0.018	0.028	0.004	0.009	0.022
1986	0.047	0.202	0.083	0.030	0.120	0.057	0.012	0.034	0.032	0.007	0.013	0.031
1987	0.045	0.085	0.073	0.026	0.064	0.060	0.012	0.020	0.033	0.009	0.014	0.027
1988	0.049	0.160	0.053	0.037	0.137	0.048	0.021	0.064	0.031	0.011	0.025	0.023
1989	0.045	0.160	0.070	0.029	0.085	0.048	0.020	0.064	0.029	0.011	0.020	0.023
1990	0.047	0.147	0.092	0.034	0.098	0.059	0.020	0.079	0.039	0.012	0.032	0.033
1991	0.038	0.099	0.067	0.029	0.074	0.054	0.022	0.062	0.036	0.009	0.016	0.034
1992	0.046	0.262	0.066	0.030	0.174	0.054	0.018	0.083	0.039	0.010	0.030	0.033
1993	0.024	0.043	0.058	0.020	0.039	0.054	0.013	0.021	0.043	0.010	0.016	0.036
1994	0.051	0.203	0.099	0.053	0.193	0.084	0.039	0.137	0.072	0.019	0.087	0.053
1995	0.014	0.041	0.092	0.010	0.024	0.101	0.006	0.015	0.083	0.005	0.013	0.064
1996	0.021	0.055	0.103	0.016	0.061	0.088	0.009	0.032	0.070	0.005	0.010	0.075
1997	0.011	0.036	0.128	0.011	0.027	0.126	0.011	0.025	0.109	0.008	0.017	0.110
1998	0.015	0.030	0.091	0.013	0.032	0.092	0.009	0.017	0.082	0.007	0.016	0.082
1999	0.008	0.021	0.094	0.008	0.014	0.066	0.006	0.012	0.057	0.005	0.010	0.056
2000	0.015	0.066	0.100	0.014	0.046	0.098	0.006	0.011	0.095	0.004	0.009	0.078
2001	0.018	0.056	0.085	0.017	0.044	0.084	0.013	0.039	0.078	0.008	0.020	0.065
2002	0.017	0.050	0.091	0.021	0.078	0.086	0.014	0.049	0.086	0.010	0.025	0.072
2003	0.015	0.053	0.112	0.017	0.063	0.088	0.014	0.044	0.092	0.007	0.017	0.071
2004	0.014	0.037	0.073	0.013	0.024	0.076	0.008	0.020	0.064	0.006	0.018	0.054
2005	0.018	0.048	0.056	0.018	0.040	0.049	0.014	0.038	0.045	0.007	0.015	0.040
2006	0.017	0.052	0.035	0.020	0.053	0.031	0.012	0.027	0.025	0.006	0.009	0.033
2007	0.020	0.057	0.031	0.013	0.024	0.029	0.012	0.034	0.034	0.005	0.008	0.034
2008	0.016	0.039	0.062	0.013	0.033	0.053	0.010	0.022	0.038	0.005	0.010	0.039
2009	0.013	0.028	0.045	0.012	0.023	0.051	0.010	0.036	0.046	0.008	0.021	0.030
2010	0.013	0.039	0.057	0.012	0.025	0.042	0.008	0.015	0.038	0.006	0.017	0.043
2011	0.016	0.041	0.067	0.016	0.044	0.057	0.013	0.040	0.049	0.006	0.009	0.058
2012	0.011	0.026	0.079	0.013	0.026	0.074	0.007	0.019	0.053	0.006	0.010	0.069
2013	0.013	0.026	0.048	0.012	0.021	0.052	0.009	0.018	0.040	0.004	0.008	0.041
2014	0.011	0.021	0.046	0.010	0.022	0.041	0.007	0.018	0.033	0.003	0.006	0.025
2015	0.013	0.029	0.075	0.011	0.036	0.086	0.006	0.016	0.065	0.004	0.006	0.060
2016	0.009	0.017	0.079	0.010	0.027	0.065	0.006	0.008	0.059	0.004	0.006	0.063
2017	0.013	0.041	0.069	0.012	0.022	0.067	0.005	0.011	0.047	0.004	0.009	0.044

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