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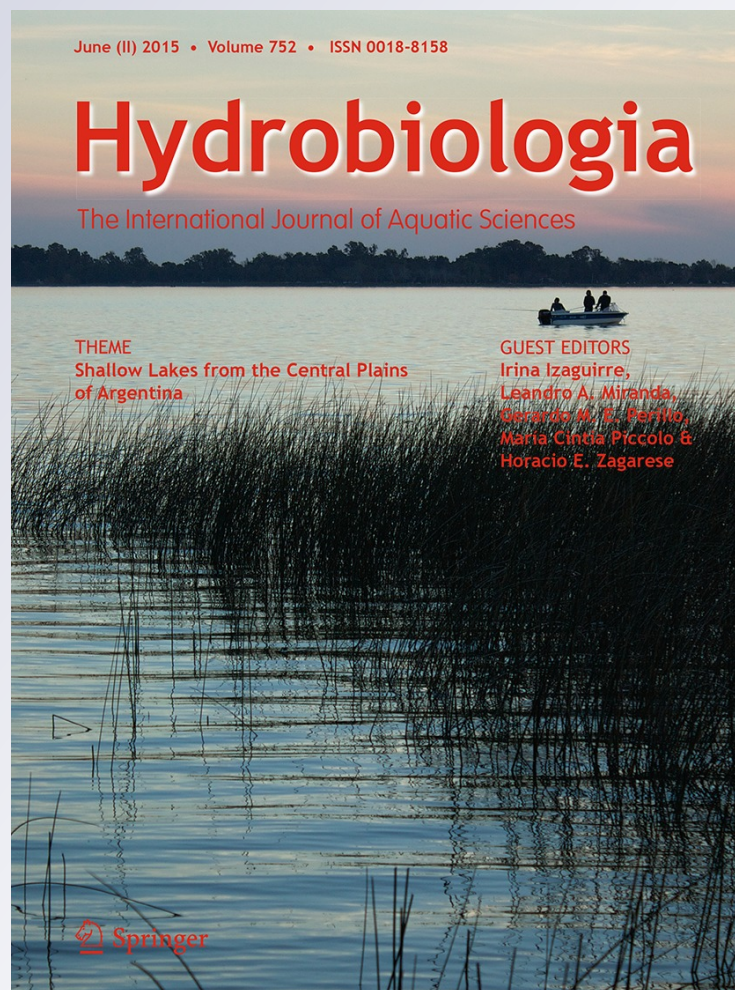
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# Changes in the phytoplankton structure in a Pampean shallow lake in the transition from a clear to a turbid regime

María Laura Sánchez · Leonardo Lagomarsino ·  
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**Abstract** We analysed the changes in phytoplankton and in the main limnological features in a shallow lake during its transition from a clear-vegetated regime to a turbid one from 2005 to 2013. As samplings were discontinuous, data were analysed considering three different sampling periods. At the beginning of the first period, the lake was in a clear-vegetated regime, showing low values of chlorophyll *a*,  $K_{dPAR}$ , total suspended solids and nutrients, and high Secchi depth. Phytoplankton was dominated by nano-phytoplanktonic species. During the second period, some evidences of the shift to a turbid regime were observed (mainly in  $K_{dPAR}$  and total suspended solids). Towards the end of our study, submerged macrophytes

sharply declined; in this period  $K_{dPAR}$  and total suspended solids noticeably increased, whereas a significant reduction in Secchi depth occurred. Concomitantly, phytoplankton abundance augmented in two orders of magnitude, changing to a community with a higher proportion of micro-phytoplankton. Although the causes of the regimen shift could not be unequivocally assessed, the drastic reduction in the hydrometric level of the lake probably provoked a declination in macrophytes, with the consequent increase of nutrients in the water column and the increment in phytoplankton densities, carrying the system towards a turbid regime.

**Keywords** Shift regimes · Phytoplankton · Macrophytes · Shallow lakes · Pampean plain

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## Introduction

Alternative equilibria states have been well documented in natural ecosystems (Holling, 1973; May, 1977; Schröder et al., 2005), predicting that under certain environmental conditions a particular ecosystem can abruptly shift from one state to another, changing their species assemblages (Scheffer, 2001). The well-known “alternative equilibria model” proposed by Scheffer et al. (1993) for aquatic ecosystems postulated that a shallow lake may alternate between turbid and clear states, depending on the interactions

of several factors: macrophytes, phytoplankton, hydrometric levels and nutrient concentrations. An increment in some of the environmental variables could provoke the change from one state to another if the critical threshold is exceeded (Scheffer & Carpenter, 2003). Many investigations have documented that some lakes often exhibit hysteresis, being very difficult to return a lake to its original regime, even diminishing the level of the variable that causes the change (Scheffer et al., 1993; Ibelings et al., 2007). In this manner, often it is necessary to apply alternative measures of restoration such as removal of fishes, iron addition or changes in the hydrometric level (Bakker et al., 2013; Chen et al., 2013; Immers et al., 2013 among others).

Nevertheless, the stable state concept is a simplification of a more complex reality, since natural systems suffer permanent slow changes, thus their stable states are more adequately called regimes, and the sudden changes between these conditions as shift regimes (Scheffer, 2009). In the transition from a clear regime to a turbid one, different changes in the structure of the aquatic communities have been documented, such as in the main autotrophic communities, zooplankton assemblages and fish populations (e.g. Blindow et al., 1993; Scheffer, 2001; Hansel-Welch et al., 2003; Zimmer et al., 2003; Scheffer & van Ness, 2007; Moss, 2010).

The Pampa plain is a region with plenty of permanent shallow lakes, constituting an important wetland. Since the beginning of this century, most of the shallow lakes of the region show a turbid regime, characterized by high phytoplankton biomass, but also there are some clear-vegetated and inorganic-turbid lakes (Quirós et al., 2002; Allende et al., 2009; Izaguirre et al., 2012). An example of a particular lake that experienced a shift in its regime in the region was shown by Cano et al. (2008) and Casco et al. (2009).

Previous studies conducted in freshwater systems of the Pampa region have noticeably shown that clear and turbid lakes exhibit contrasting phytoplankton communities. Nano-phytoplanktonic algae, mainly flagellates, are more abundant in clear lakes, whereas diatoms, chlorococcacean and some bloom-forming cyanobacteria dominate in turbid lakes (e.g. Izaguirre & Vinocur, 1994; Allende et al., 2009). Recently, different phytoplankton functional classifications have been used and proved to be effective in differentiating clear and turbid ecosystems of this region (Izaguirre et al., 2012).

The shift regime would be provoked by several factors. For example, the loss of vegetation by changes in the hydrometric level or increments in nutrient loading may provoke changes in the turbidity of waters (Scheffer, 1998), affecting the structure of the phytoplankton community. Particularly, the different land uses in the catchment area of the lakes are important drivers of the phytoplankton community structure, being agriculture one of the most important (Katsiapi et al., 2012). In this sense, the Pampa region has suffered intensification in their land use over the last decades, mainly due to agriculture and cattle breeding activities that have provoked an increase in the use of agrochemicals with the consequent degradation of the wetland and the acceleration of eutrophication process (Quirós et al., 2006). As a result, those zones with more intense anthropogenic activities have a high number of turbid shallow lakes (Quirós et al., 2002).

Phytoplankton algae have characteristic short life cycles, and thus may quickly reflect any change that occurs in the environment. However, long-term studies of phytoplankton dynamics are necessary to obtain a good understanding and to identify the ecological responses of this community to environmental changes (Reynolds, 1984).

In this investigation, we analyse the changes in the phytoplankton structure associated to the changes in limnological features that occurred in a Pampean shallow lake during the transition from a clear-vegetated regime to a turbid one, from 2005 to 2013. Since samplings were discontinuous over the 7 years, data are analysed considering three different periods. We describe changes in phytoplankton abundance, composition and proportion of the size fractions (nano- and micro-phytoplankton), as well as the variations in environmental variables, and their importance on phytoplankton structure.

## Study site

El Triunfo is a shallow lake located in the Pampa Plain, Buenos Aires Province, Argentina (35°51'S; 57°52'W). The main climatic and geological characteristics of the region have been described in several papers (e.g. Quirós & Drago, 1999; Allende et al., 2009; see also Diovisalvi et al. this special volume).

The area of El Triunfo is 1.5 km<sup>2</sup> and its maximum depth ( $z_{\max}$ ) approximately 2 m. Its perimeter is



5.1 km, and maximum length and width are 1.8 and 1.5 km, respectively (Silvoso et al., 2010). This shallow lake is included in the Salado River Basin, and as other water bodies of this area presents eutrophic conditions.

## Methods

In order to analyse the change in the regime of El Triunfo throughout the time, we included data obtained from previous studies conducted in this shallow lake from 2005 to 2011 (Allende et al., 2009; Sánchez et al., 2010, 2013; Torremorell et al. this special volume). It is important to mention that for this previous data set, there are lacks of information in some years. Since October 2012, the shallow lake has been monthly sampled within the framework of the PAMPA<sup>2</sup>, a long-term network-monitoring project. Thus, the gathered data encompass three periods: the first from November 2005 to December 2007, the second from December 2010 to October 2011 and the third from October 2012 to September 2013.

### Environmental variables

Monthly precipitation data of the region were provided by Servicio Meteorológico Nacional (Argentina).

Several physical and chemical variables were measured in situ: pH, conductivity, dissolved oxygen (DO) and temperature, with portable electronic metres. Downward irradiance vertical profiles were obtained around noon using a USB2000 (Ocean Optics<sup>®</sup>) spectroradiometer attached to an optical fibre and a Teflon diffuser following the methodology described in Sánchez et al. (2013). Vertical diffuse attenuation coefficients for PAR ( $K_{dPAR}$ ) were determined from the slope of the linear regression of the natural logarithm of downward irradiance profiles versus depth. Additionally, we measured Secchi disk depth and turbidity with a HACH 2100P<sup>®</sup> turbidimeter. Depth was measured at the sampling point. Total suspended solids (TSS) were estimated after filtration onto weighted pre-combusted GF/F filters, drying at 105°C to a constant weight (APHA, 2005).

The concentration of the main nutrients was also determined, but as data belong to different studies, some techniques changed through the period. Until 2012, nitrogen was measured as dissolved inorganic

nitrogen (DIN) and was calculated as the sum of nitrites, nitrates and ammonium ions. Nitrate + nitrite (cadmium reduction method) and ammonium (salicylate method) were analysed with a HACH DR/2010 (HACH Company, USA) spectrophotometer using the corresponding kits of HACH<sup>®</sup> reagents. From 2012 to the end of the study period, nitrogen was measured as total organic nitrogen (TON, unfiltered water samples), and total dissolved organic nitrogen (TDON, filtered) was determined by Kjeldahl method (APHA, 2005).

Total phosphorus (TP, from unfiltered water samples) and total dissolved phosphorus (TDP, from GF/F filtered water) were converted into soluble reactive phosphorus (SRP) after an acid digestion (sulphuric acid with persulfate potassium, 30 min at 120°C). In turn, SRP was determined using the molybdate-ascorbic acid method according to the analytical procedures (APHA, 2005). Particulate phosphorus (Ppart) was calculated by subtracting TDP from TP.

### Phytoplankton

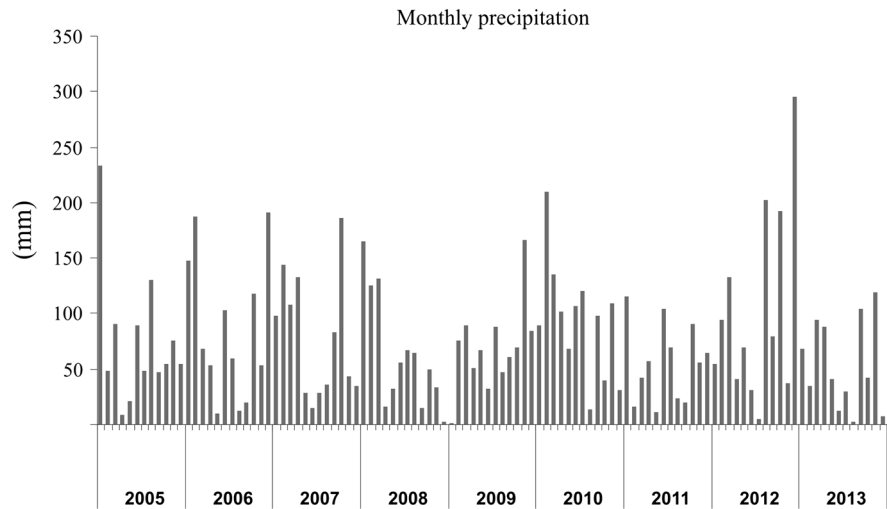
Sub-superficial phytoplankton samples were fixed with acidified lugol 1% and counted under inverted microscope following the methodology described in Utermöhl (1958); the counting error was estimated according to Venrick (1978). Densities of phytoplankton taxa were calculated for each sampling date. We also estimated the variation on the proportion of micro- and nano-plankton throughout the analysed period.

Water samples for chlorophyll *a* (Chl *a*) determinations were filtered through Whatman GF/F filters; Chl *a* was estimated by spectrometry using hot ethanol as solvent following the methodology described in Nusch (1980) and using the equations proposed in Lorenzen (1967) to the pigment calculations.

### Statistical analyses

In order to analyse the changes that occurred in the lake over the study period, we performed regression analysis of several variables (total phytoplankton abundance, TSS, Chl *a* and  $K_{dPAR}$ ) on time, adjusting each one to the best fit model (lineal, polynomic, logarithmic or exponential); the assumptions of normality and homoscedasticity were previously tested. Non-parametric Spearman correlations were used to investigate the relationship between physical and chemical variables. To compare micro- and nano-phytoplankton regression analyses on

**Fig. 1** Monthly precipitation registered in the region from January 2005 to December 2013. Data provided by Servicio Meteorológico Nacional (Argentina)



time of each phytoplankton fraction, a contingency table was performed. We compared data set of the first period (2005–2007) against the third one (2012–2013), which are the most contrasting periods, using the Mann–Whitney rank-sum non-parametric test. In order to compare the inter-annual variations and avoided the seasonal fluctuations, the regressions were performed considering only data of the warm seasons. All of these statistical analyses were carried out using the software Infostat<sup>®</sup> and STATISTICA 7<sup>®</sup>. On the other hand, to analyse the relationships among environmental variables and main taxonomical groups, we performed a multi-variate analysis. First, we ran a detrended correspondence analysis (DCA), which showed a linear response of the biological data; thus, a redundancy analysis (RDA) was applied based on the densities of the principal taxonomical groups and the main environmental variables. Samples lacking some of the environmental variables were eliminated from the analysis. The statistical significance of the first axis and of all the axes was tested by a Monte Carlo permutation test, and the importance of each environmental variable was assessed using forward selection. Analyses were performed using the software CANOCO (Ter Braak, 1986).

## Results

### Environmental variables

Since January 2005 to December 2013, it is observed a succession of dry and wet years with a peak of

precipitation in December 2012 (295 mm) and a minimum value in January 2009 (1.3 mm) (Fig. 1). Regarding the mean annual values of precipitation, it is evident that 2010 and 2012 were the wettest years, and 2011 and 2013 were the driest ones.

Through the present study, depth at the sampling point diminished from about 1.3 m in 2005 to 0.7 m in 2013, evidencing a clear decrease in the water level of the lake towards the end of the study period.

Over the three analysed periods, the shallow lake had well-oxygenated and alkaline waters. Conductivity values were relatively high and showed an increasing trend associated to the diminishing in the hydrometric level (Table 1).

Regarding the optical variables, Secchi disk exhibited a decreasing tendency throughout the three periods, with maximum values of 84 cm at the first period to a minimum of 10 cm at the third one. This pattern was in concordance with changes in TSS that showed an increase over the study periods, with values from 1 mg l<sup>-1</sup> (first period) to 122 mg l<sup>-1</sup> (third period) (Table 1). Chl *a* also showed an increment from 1.04 µg l<sup>-1</sup> (first period) to 543.6 µg l<sup>-1</sup> (third period). TSS and Chl *a* were closely correlated ( $R = 0.93$ ,  $n = 31$ ;  $P < 0.001$ ). A peak of turbid conditions was registered in June 2012, with maximum values of  $K_d_{PAR}$ , Chl *a* and TSS, and lowest of Secchi depth. Exponential regressions were statistically significant for Chl *a* throughout the time ( $P = 0.02$ ;  $R = 0.59$ ) (Fig. 2b) and for TSS ( $P = 0.02$ ;  $R = 0.59$ ) (Fig. 2c).

Concomitantly with the enhancement in the turbid conditions of the lake from 1.1 NTU in the first period

**Table 1** Mean, maximum and minimum values (between brackets) of the main physicochemical variables of El Triunfo throughout the three analysed periods: first period (November

2005–December 2007), second period (December 2010–October 2011) and third period (October 2012–September 2013)

	First period	Second period	Third period
Temperature (°C)	13.3 (25.4–7)	18.4 (29–9)	21.5 (32–10.5)
pH	8.73 (9.45–8.13)	8.81 (9.80–8.33)	9.06 (9.74–8.26)
Conductivity ( $\mu\text{S cm}^{-1}$ )	1,535 (1,870–1,320)	2,827 (3,830–1,890)	2,423 (2,650–1,910)
DO ( $\text{mg l}^{-1}$ )	9.6 (18–6.8)	8.1 (12–5.5)	11 (14.6–8.1)
$K_{d\text{PAR}}$ ( $\text{m}^{-1}$ )	6.6 (11–3.4)	6.8 (12–5.5)	13 (26.3–2.6)
Secchi (cm)	55 (84–43)	29 (59–17)	25 (80–10)
Turbidity (NTU)	1.1 <sup>a</sup>	n.d.	71.1 (169–15.4)
TSS ( $\text{mg l}^{-1}$ )	6.6 (27.5–1)	18.7 (51.6–5.2)	75.3 (17.7–122)
TP ( $\mu\text{g l}^{-1}$ )	n.d.	346 (1,099–20)	872 (1,183–531)
Ppart ( $\mu\text{g l}^{-1}$ )	n.d.	107 (318–2)	172 (538–u.d.)
DIN ( $\mu\text{g l}^{-1}$ )	8.1 (19–u.d.)	7.4 (14–3)	46.4 (84–24)
TON ( $\mu\text{g l}^{-1}$ )	n.d.	8,365 (12,387–6,227)	8,251 (11,054–5,280)

DO dissolved oxygen,  $K_{d\text{PAR}}$  vertical diffuse attenuation coefficients for PAR, TSS total suspended solids, AFDW ash-free dry weigh, TP total phosphorus, Ppart particulate phosphorus, DIN dissolved inorganic nitrogen, TON total organic nitrogen, n.d. no determined, u.d. undetectable

<sup>a</sup> Only one data available

to 169 NTU at the end of third period, an increasing trend of  $K_{d\text{PAR}}$  was registered throughout the time from 6.2 to 26.3  $\text{m}^{-1}$  (Fig. 2a), although the last variable did not fit well with any regression model.

Conductivity, DO,  $K_{d\text{PAR}}$ , TSS and Chl *a* showed significant differences between the first and the third period ( $U_{\text{M-W}} = 80$ ,  $P < 0.0001$ ;  $U_{\text{M-W}} = 113$ ,  $P < 0.05$ ;  $U_{\text{M-W}} = 41$ ,  $P < 0.05$ ;  $U_{\text{M-W}} = 37$ ,  $P < 0.0005$ ;  $U_{\text{M-W}} = 78$ ,  $P < 0.0001$ ; respectively).

Regarding nutrient concentrations, TON ranged from 5280 to 12378  $\mu\text{g N l}^{-1}$  for the second and third periods, because no data of TON were obtained for the first one (Table 1). The dissolved inorganic forms of nitrogen (DIN) contributed less to total nitrogen, ranging from undetectable at the beginning of the first period to 84  $\mu\text{g N l}^{-1}$  at the end of the third one. DIN values were highly correlated with Chl *a* ( $R = 0.59$ ,  $n = 24$ ,  $P < 0.001$ ) and TSS ( $R = 0.77$ ,  $n = 21$ ,  $P < 0.001$ ). TP was measured from December 2010 to the end of the study period, showing an increasing trend over time, with values ranging from 20  $\mu\text{g P l}^{-1}$  (second period) to about 1200  $\mu\text{g P l}^{-1}$  (third period). Considering the different fractions contributing to total phosphorus, Ppart also showed an increasing tendency over the studied periods, with values ranging from 2  $\mu\text{g P l}^{-1}$  to close to 550  $\mu\text{g P l}^{-1}$ .

Changes in TP and Ppart were closely related to changes in TSS values ( $R = 0.51$ ,  $n = 23$ ,  $P < 0.05$  and  $R = 0.85$ ,  $n = 23$ ,  $P < 0.001$ , respectively).

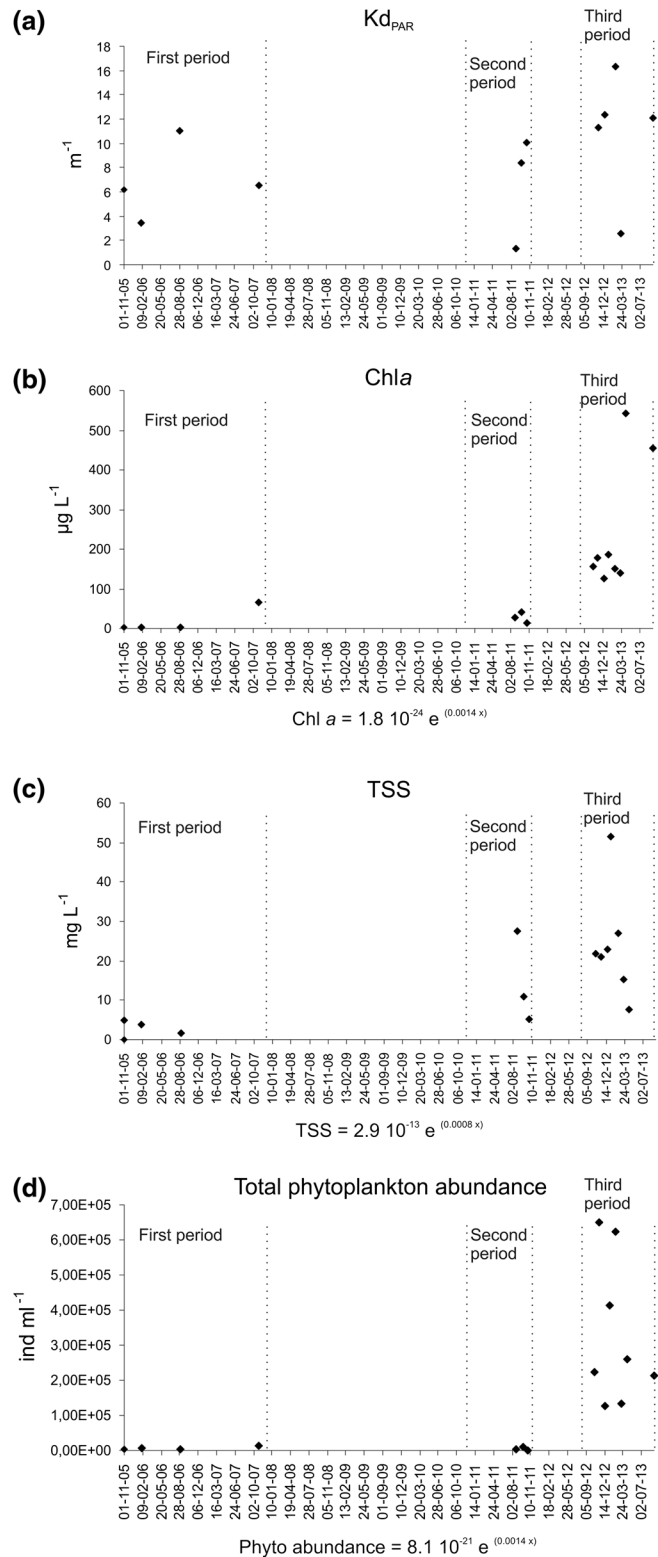
#### Changes in macrophyte abundance

At the beginning of this study (first period analysed), the shallow lake was completely colonized by submerged macrophytes (mainly *Ceratophyllum demersum*). At the end of 2011, the hydrometric level diminished notoriously, but no evidences of macrophyte disappearance were found (personal observation). At the beginning of 2013, the submerged plants diminished drastically (covering less than 50%). The macrophyte abundance continued declining during this year, and at the end our study (September 2013), the shallow lake only exhibited patches of emergent vegetation (*Schoenoplectus californicus*).

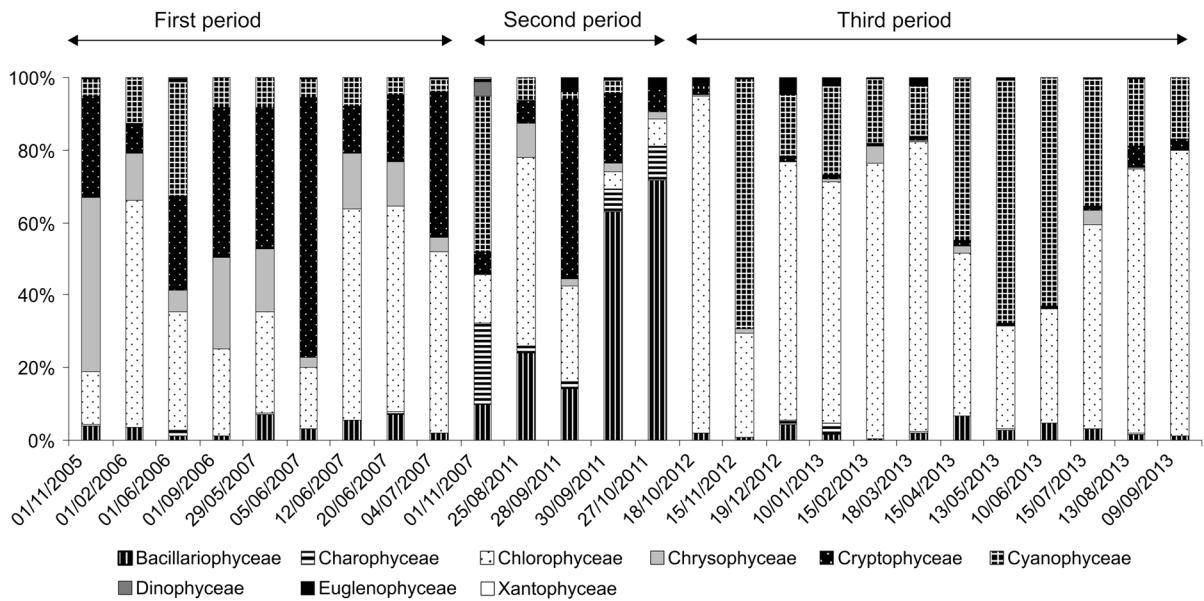
#### Phytoplankton community

Total phytoplankton abundance increased two order of magnitude since the beginning of the first analysed period ( $4 \times 10^3 \text{ ind ml}^{-1}$ ) to the end of the third period ( $2 \times 10^5 \text{ ind ml}^{-1}$ ), observing a significant exponential regression ( $P = 0.03$ ;  $R = 0.56$ ) (Fig. 2d).

**Fig. 2** **a**  $K_{dPAR}$ , **b** Chl *a*, **c** TSS, **d** Total phytoplankton abundance, throughout the analysed period. Equation of significant regression is shown for each variable

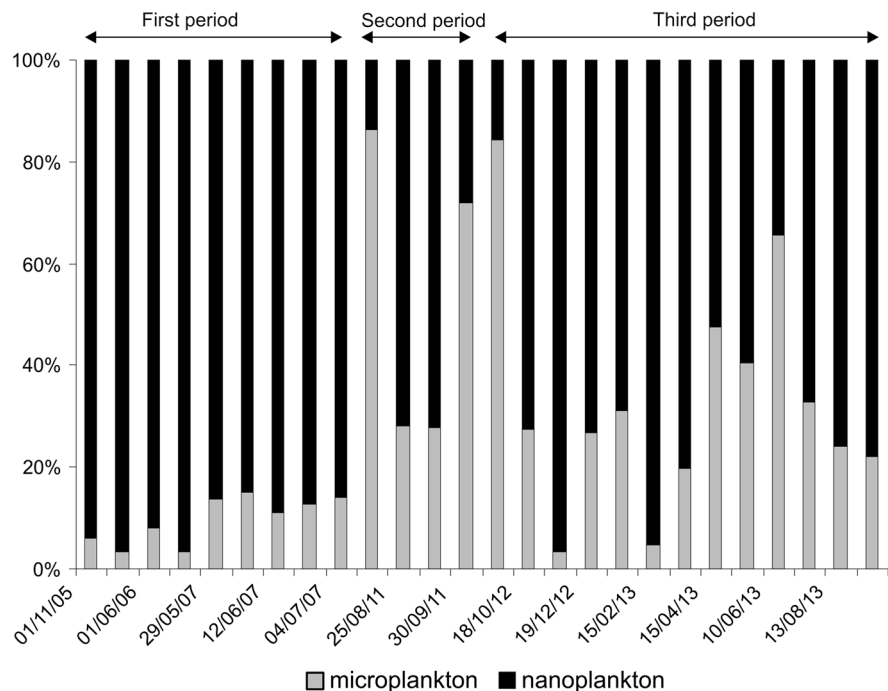






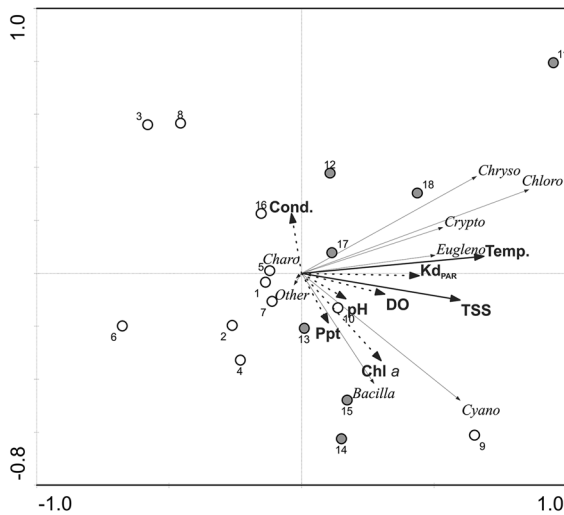
**Fig. 3** Phytoplankton structure variation throughout the analysed period

**Fig. 4** Relative proportions of nano- and micro-phytoplankton to the total phytoplankton abundance



Besides the changes in total abundances, phytoplankton community exhibited important changes in their structure (Fig. 3), shifting from a community dominated by nano-flagellates belonging to chrysophytes (*Ochromonas* sp.), unicellular chlorophytes

(*Monoraphidium circinale*) and pennate diatoms (*Nitzschia* spp.) at the beginning of the first period, to a community dominated by unicellular chlorophytes (*Chlamydomonas* spp., *Monoraphidium* spp., and *Oocystis* spp.) and colonial cyanobacteria (*Aphanocapsa*



**Fig. 5** Triplot corresponding to the redundancy analysis (first and second axis) based on the abundance of the different phytoplanktonic groups and environmental variables. *Solid and dotted arrows* indicate significant and non-significant environmental variables. *Cond* conductivity, *DO* dissolved oxygen, *Kd<sub>PAR</sub>* vertical diffuse attenuation coefficients for PAR, *Chl a* chlorophyll *a*, *Temp.* temperature, *Ppt* precipitation, *TSS* total suspended solids. The names of the phytoplanktonic groups are abbreviated: *Crypto* Cryptophyceae, *Chryso* Chrysophyceae, *Eugleno* Euglenophyceae, *Cyano* Cyanophyceae, *Bacilla* Bacillariophyceae, *Chloro* Chlorophyceae, *Charo* Charophyceae

*delicatissima*, *Chroococcus minor* and *Merismopedia minima*) at the end of the study. Abundances of micro-phytoplankton fit with a linear regression, increasing on time ( $P < 0.05$ ,  $R^2 = 0.26$ ), while nano-phytoplankton did not fit with any regressions. Moreover, the proportion of micro-phytoplankton/nano-phytoplankton significantly increased throughout the time (table contingency,  $P < 0.0001$ ) (Fig. 4).

The results of the RDA analysis (Fig. 5) showed that the first two axes accounted for 71.5% of the variance (axis 1: 55.2%, axis 2: 16.3%). Abiotic factors were significantly correlated with the first axis and with all axes (Monte Carlo's test for the eigenvalues:  $P < 0.05$  and  $P < 0.05$ , respectively). Temperature ( $P < 0.05$ ) and TSS ( $P < 0.05$ ) were statistically significant (solid arrows in Fig. 5). The first axis showed higher correlation with temperature, TSS and  $Kd_{PAR}$  (correlation coefficients: 0.60, 0.52 and 0.39, respectively), and the second axis was mainly correlated with *Chl a* and conductivity (correlation coefficients:  $-0.25$  and  $0.17$ , respectively). The ordination of the samples with respect to the two first axes (Fig. 5) evidenced the

differences in the regime of the lake. Most samples corresponding to the first period (clear regime) were positioned at the left part of the figure (white circles). On the other hand, most samples corresponding to the turbid period were placed at the right side of the graph (grey circles), towards higher values of  $Kd_{PAR}$ , TSS and *Chl a*, and increasing abundances of phytoplankton (Fig. 5).

## Discussion

The pattern of most variables analysed, as well as the results of the multivariate approach, clearly evidenced the shift regime of the shallow lake throughout the analysed period, from a typical clear regime dominated by submerged macrophytes to a turbid one with high phytoplankton abundance. This transition was accompanied by changes in the structure of the phytoplankton community. Regarding nutrient concentration, in spite of the eutrophic conditions of the shallow lake El Triunfo, there were some evidences of nitrogen limitation during the vegetated stage. Macrophytes take nitrogen from the water column diminishing DIN availability (Mjelde & Faafeng, 1997; Villar et al., 1998) and provoke a nutrient limitation to algal growth, situation that was previously reported for this lake (Allende et al., 2009). On the other hand, in the turbid phase, an increase in DIN values was detected, concomitantly with the disappearance of aquatic vegetation and the higher phytoplankton abundances; this increment was accompanied with increases in *Chl a* and TSS. In a study that involved 300 Danish lakes, Søndergaard et al. (2010) found that several macrophytes metrics were negatively correlated with TN, TP and *Chl a*. In particular, Bayley et al. (2007) postulated that clear lakes usually had less than  $18 \mu\text{g Chl } a \text{ l}^{-1}$ , and over this value, lakes are in the turbid state.

In the literature, there are many examples of shallow lakes that present alternations between turbid and clear regimes (e.g. Blindow, 1992; Scheffer, 1998; Bayley et al., 2007; Tátrai et al., 2009). It has been postulated that the changes may be caused by intrinsic processes (such as internal nutrient release during decomposition of great amount of organic material of macrophytes) and external factors (such as catastrophic damage of the vegetation or high loading of phosphorus from the surrounding areas) (Hargeby et al., 2007).

For the shallow lakes of the Pampa Plain, the only documented shift in regimes was observed in another lake of the Salado River Basin (Lake Lacombe), where alternations between clear and turbid regimes were observed in short time periods. In this water body, authors reported three states for only 1 year (one turbid and two clear), with rapid changes in the abundance and composition of the algal communities (Casco et al., 2009; Cano et al., 2008).

Macrophytes enhance water clarity by several mechanisms: compete with phytoplankton by nutrients in the water column, secrete allelopathic substances and provide refuge to zooplankton grazers (Søndergaard & Moss, 1997; Scheffer, 1998). Moreover, the macrophyte bed diminishes water turbulence and provokes an increase in the sedimentation rate of phytoplankton (Søndergaard & Moss, 1997). At the beginning of our study (first period), the bottom of the shallow lake El Triunfo was completely colonized by submerged macrophytes, situation that involved a reduction in waves and nitrogen limitation. These conditions were favourable for nano-phytoplankton, which exhibited a higher proportion than micro-phytoplankton. This structure is in agreement with the descriptions given for several Pampean lakes that were investigated in the 90s (Izaguirre & Vinocur, 1994); in this pioneer survey, a higher contribution of nano-phytoplankton was found in vegetated lakes compared with those registered in systems lacking submerged vegetation. More precisely, nano-flagellates (e.g. chrysophytes) were reported as dominant under this scenario in another clear lake of the same region (Allende et al., 2009). The species belonging to this algal group have own motility that avoid sedimentation (Reynolds, 1984; Søndergaard & Moss, 1997), and also have been reported as mixotrophs (Jones, 2000), both strategies that redound in benefits to them under such conditions.

During the second period of our study, the shallow lake was still in a clear-vegetated regime. Nevertheless, some signs of the transition to a turbid state began to be detected, particularly in  $Kd_{PAR}$  and TSS.

On the other hand, unicellular chlorophytes and colonial cyanobacteria are usually dominant under turbid conditions as was observed in the third period analysed. Huisman et al. (1999) showed that under experimental limiting light conditions, cultures of *Chlorella* sp. and *Microcystis* sp. have resulted in better light competitors than filamentous cyanobacteria and

cenobial chlorophytes, and that this advantage was related with their critical light intensities due to differences in the light absorption spectra. Moreover, cyanobacteria contain phycocyanin that absorbs at 630 nm (Stomp et al., 2004) and confers advantage in turbid eutrophic waters (Scheffer et al., 1997). Even, in some turbid Pampean shallow lakes, an absolute dominance of filamentous cyanobacteria was reported (e.g. Sánchez et al., 2013). Consequently, under this new turbid scenario that we observed in lake El Triunfo, we cannot discard that phytoplankton could evolve towards a community exclusively dominated by cyanobacteria, with the health risk that it implies.

A shift regime in a shallow lake may be triggered by several factors such as the introduction of exotic fishes, increases in nutrient concentrations, storms, changes in hydrometric level or in hydrogeomorphic features (Bachmann et al., 1999; Cristofor et al., 2003; Rodríguez et al., 2003; Van Geest et al., 2007 among others). Although we could not unequivocally determine the factors that lead to the regime shift in El Triunfo, it is likely that it is associated to a decrease in the lake water level, also evidenced by the increase in conductivity values. Contrarily to what was postulated by Van Geest et al. (2007), where low water depths favoured the re-establishment of macrophytes, we observed that the new conditions probably provoke an unusual exposition of the macrophytes to drought with the consequent mortality of these aquatic plants, as was also proposed by Blindow et al. (1993) to explain changes from clear to turbid conditions in a large shallow lake. A great amount of death organic material may be dumped into the water body, provoking an increase in the nutrient concentrations (Carpenter & Lodge, 1986). Additionally, the release of nutrients from sediments in enriched waters favours the phytoplankton growth (Cymbala et al., 2008), with the consequent augmentation of turbidity. The diminishing in macrophytes indirectly promotes the phytoplankton growth since they act as refuge for zooplankton community and releases allelopathic substances (Søndergaard & Moss, 1997; Burks et al., 2006).

However, we cannot be entirely sure which was the trigger process in the case of El Triunfo, since other causes, such as an increase in the agricultural activities in the surrounding areas, may also have occurred during the last years. In this sense, the continuous monitoring of the lake, which is conducting within the framework of a network project, constitutes the best

way to elucidate the causes associated to the changes in the water bodies of the region.

The pristine state of the shallow lakes from the Pampean plain probably was a clear one with abundant aquatic vegetation (Quirós et al., 2006). However, in the last decades with changes in land uses, as the expansion of agriculture frontier, the increase in the amount of agrochemical used in the region accelerated the eutrophication process of the lakes. It is reflected in a higher number of turbid lakes registered in the most human-impacted zones (Quirós et al., 2002).

Management measures are needed to the restoration of clear states in these water bodies. The reduction of nutrient loading is necessary but would be hard and not effective (Moss, 2010). Probably, the improvement of the conditions for the re-establishment of macrophytes could be a successful restoration measure. In this sense, different restoration techniques have been applied in other shallow lakes, for example the biomanipulation of fish populations to provoke the increment on zooplankton densities with the consequent diminishing of phytoplankton biomass. Once the lake shifts to a clear regime, aquatic plants may begin to re-establish. It is also necessary to prevent the loading of herbicides from the catchments that can affect the growth of aquatic plants (Moss, 2010 and cites therein). Moreover, other biotic factors as the existence of propagules, herbivory and competition between macrophytes might be considered to design a good restoration strategy (Bakker et al., 2013). In this sense, a compromise between adequate restoration measures as well as a better planning of the land use in the region could result in a good conservation plan of this important wetland.

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