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Phytoplankton dynamics and structure, and ecological status estimation by the *Q* assemblage index: a comparative analysis in two shallow Mediterranean lakes

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Abstract: The aim of this research was to compare the phytoplankton structure on the basis of environmental variables and test the *Q* assemblage index based on phytoplankton functional groups in two shallow Mediterranean lakes [Lake Taşkısığı (LT) and Lake Little Akgöl (LLA)] in the north of Turkey. Variations in the phytoplankton structure and environmental parameters were analyzed monthly at two stations of each lake between January 2013 and December 2013. We showed that total phosphorus (TP) values were vital in shaping phytoplankton biomass in eutrophic LLA. Nitrate-nitrogen (NO_3-N), TP, z_{mix}/z_{eu}, and macrophyte coverage are important in shaping the differences in dominant phytoplankton functional groups between two lakes. Generally, diversity values were similar between the two lakes and environmental parameters had no effect on diversities to constitute significant differences. The assessment using the *Q* index gave compatible results for determining the ecological status of LT and LLA.

Key words: Biomass, diversity, functional groups, *Q* index, Water Framework Directive

1. Introduction

Shallow lakes in Mediterranean climates, which are generally situated in lowland areas, due to their high evaporation/precipitation ratios, low geographic relief, and dense human population have long water residence times and are becoming more eutrophic (Allan et al., 1980; Borics et al., 2013). Moreover, elevated release of phosphorus from sediments or greater loads from the catchment area and destruction of submerged vegetation may trigger an increase of phytoplankton density and a related decrease in water clarity (Moss et al., 2009; Dokulil and Teubner, 2011). The increased biomass of phytoplankton and frequently occurring toxic algal blooms triggered the reassessment of lake management strategies (Borics et al., 2013). The Water Framework Directive was deigned to assess the ecological quality of surface waters through the analysis of various characteristics of aquatic flora and fauna, and to declare management plans in European countries (EC, 2009). Investigation of the functional traits of phytoplankton of shallow lakes was found to be important to estimate ecological quality and to understand the operation of these systems (Borics et al., 2012).

Many attempts have been made to categorize traits and functions of phytoplankton (Reynolds et al., 2002; Borics et al., 2007; Padisák et al., 2009). At present, 40 phytoplankton functional groups (FGs) have been described, identified by numeric character codes (codons) (Padisák et al., 2009). Padisák et al. (2006) developed an index (*Q* index) using FGs to estimate the ecological status of lakes. The index combines the relative weight of FGs in the total biomass and considers a factor number for each assemblage for each type of water body. It was tested on water bodies significantly differentiated by origin, altitude, salinity, mixing, and stratification in the world (e.g., Crossetti and Bicudo, 2008; Pasztaleniec and Poniewozik, 2010) and in Turkey (e.g., Demir et al., 2014; Çelik and Sevindik, 2015).

Lake morphometry and hydrology are criteria for the composition of lake biota (Murray and Pullar, 1910), and may favor distinct life strategies. Nevertheless, even taking into account the hydromorphology, substantial differences are recognizable when considering the effects of latitude and climate on phytoplankton composition and abundance (Pollingher, 1990). Moreover, lakes that are located in the same geographic region and have similar hydromorphologies could be composed of diverse phytoplankton assemblages as a result of different nutrient content and light availability (Scheffer, 1998; Naselli-Flores, 2000). In addition, Borics et al. (2014) found

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remarkable differences in phytoplankton biomass and FGs based on lake size, average depth, lake bed material, macrophyte coverage, and water regime across shallow lake types. On the basis of these findings, two shallow lakes [eutrophic Lake Taşkısığı (LT) and eutrophic Lake Little Akgöl (LLA)], which are located in the same climatic and geographic region but have different hydromorphology and macrophyte coverage, were investigated to determine which factors shape their phytoplankton structures. Trophic state estimations using chlorophyll-*a*, Secchi depth, or total phosphorus data may be confusing since shallow lakes are exposed to the large mixing effect of wind, and their chlorophyll content may be influenced by dead macro- or epiphytes. Conversely, the *Q* index could be used to determine the seasonal changes of environmental parameters and anthropogenic disturbances (Demir et al., 2014). Therefore, our second aim was to test the applicability of the *Q* assemblage index for these two lakes with different hydromorphology.

2. Materials and methods

2.1. Study areas

2.1.1. Lake Taşkısığı

LT is located at 40°52′18″N, 30°24′14″E, 13 km north of Sakarya, Turkey, as shown in Figure 1. It lies at 12 m above sea level and has a surface area of 0.9 km^2, a length of 1.2 km, a maximum depth of 5 m, and a mean depth of 1.5 m. The lake was formed in the old Sakarya River bed. It is mainly fed by underground water sources located at various places and rainfalls. Shores of the lake are covered with macrophytes (20% mean coverage) (*Phragmites* sp., *Nymphaea alba* L., and *Ceratophyllum demersum* L.). The north shore of the lake is used for recreation. LT is a shallow eutrophic freshwater lake (the mean annual chlorophyll-*a* concentration is 0.005 mg L^{-1}, total phosphorus is 0.036 mg L^{-1}, and Secchi disk depth is 68.91 cm) according to Carlson (1977) and OECD (Vollenweider and Kerekes, 1982) criteria. The influence of some physiochemical parameters on phytoplankton abundance and species composition have been previously reported for LT (Aykulu et al., 1999; Temel and Yardımcı, 2004).

2.1.2. Lake Little Akgöl

LLA is located at 40°52′38″N, 30°25′56″E, 20 km north of Sakarya, Turkey, as shown in Figure 1. It lies at 12.3 m above sea level and has a surface area of 0.16 km^2, a length of 0.58 km, a maximum depth of 1.3 m, and a mean depth of 0.5 m. The lake was formed in the old Sakarya River bed. The sole outlet is located at the northern edge connected with Çark Stream, which was formerly connected with the Sakarya River. Dense macrophyte (40% mean coverage) (*Phragmites* sp., *Nymphaea alba* L., and *Ceratophyllum demersum* L.) development was seen on the shores of the lake. The lake is surrounded by forest. It is not used for recreation or water supply. In 2001, 30 ha of the area was declared a Wildlife Protection Area. LLA is a shallow eutrophic freshwater lake (the mean annual chlorophyll-*a*

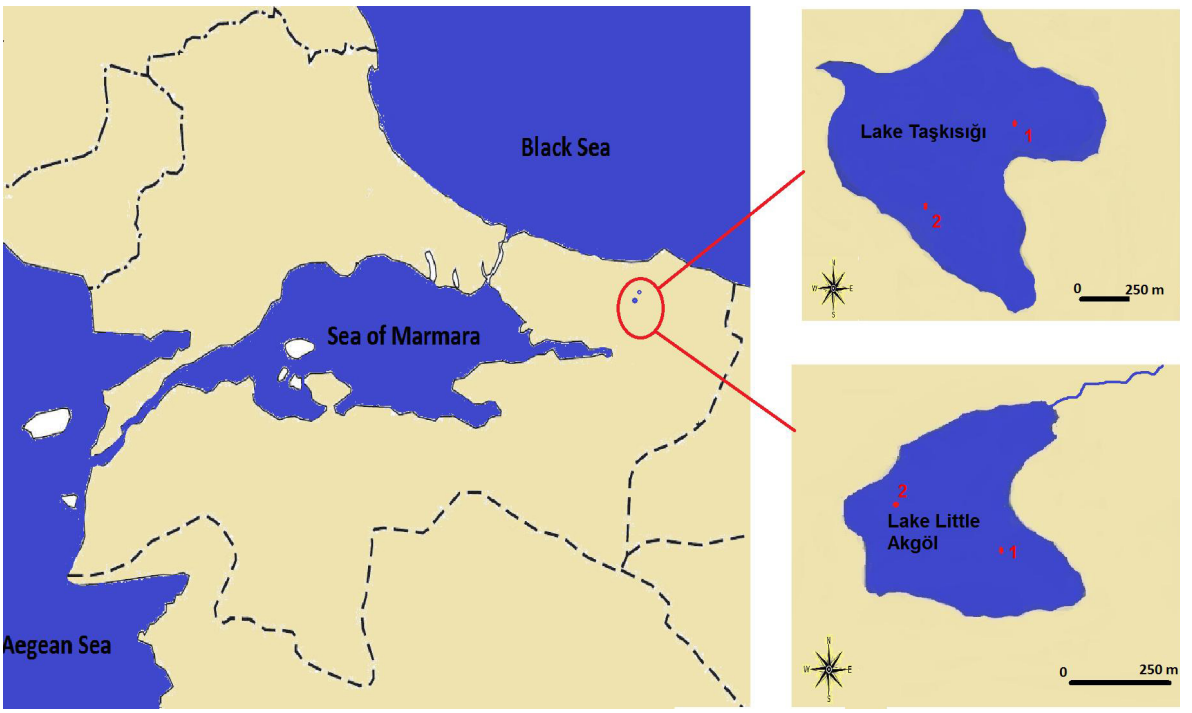

Figure 1. The map of Lake Taşkısığı and Lake Little Akgöl and the locations of sampling stations.

concentration is 0.011 mg L^{-1}, total phosphorus is 0.167 mg L^{-1}, and Secchi disk depth is 55.3 cm) according to Carlson (1977) and OECD (Vollenweider and Kerekes, 1982) criteria.

2.2. Phytoplankton analysis

Sampling was carried out monthly at two monitoring stations in each lake between January 2013 and December 2013. The distance between the sampling stations was 0.5 km in LT and 0.3 km in LLA. Mean values of the two stations were used to determine the phytoplankton structure in each lake. The whole water column was sampled with an integrated sampling tube. Samples were fixed with Lugol's solution. In the laboratory, the samples were first agitated, then poured into 50-mL graduated cylinders and allowed to settle for at least 24 h. At the end of the settling period, 45 mL of water was aspirated from each graduated cylinder and the remaining 5 mL of water was poured into a small glass vial for microscopic analysis (Utermöhl, 1958). Enumeration and identification of algae were performed using a Palmer-Maloney counting cell (volume: 0.1 mL) and a compound microscope equipped with water immersion lenses and a phase-contrast attachment (400× magnification). The final abundance of each species was considered to be the average from three replicates. Algal species were identified according to the most updated literature. Taxonomy of algae was checked according to http://www.algaebase.org. Phytoplankton biomass was calculated from biovolume estimations (Wetzel and Likens, 2000). Biovolume was calculated from the number of cells and cell size measurements by use of appropriate geometric formulas (Sun and Liu, 2003). Species were grouped in FGs according to Reynolds et al. (2002) and Padisák et al. (2009). The *Q* phytoplankton assemblage index was calculated following Padisák et al. (2006). Factor F was determined for each FG taking into account what would be the pristine status of the corresponding natural ecosystems and possible algal assemblage occurring in it. Higher factor F values were assigned to what would be pristine assemblages on the lakes and lower values to assemblages considered not desirable in pristine conditions. Floristic and taxonomic surveys in Turkey, which were presented online at http://turkiyealgleri.omu.edu.tr/, were used to calibrate factor F. Species diversity index (*H*) and evenness (*E*) were computed according to Shannon and Weaver (1963):

$$H' = -\sum p_i \ln(p_i)$$

where p_i is the relative frequency of the biomass of the *i*th species.

2.3. Analysis of environmental variables

Sampling for chemical analyses and measurements of physical variables were carried out in conjunction with phytoplankton collection. Specific electrical conductance (EC), pH, dissolved oxygen (DO), and water temperature (T) were measured at 10 cm below the surface using a YSI ProPlus water quality instrument. Water transparency was measured on each sampling day using a Secchi disk. For the analysis of chemical variables, samples were collected by a tube sampler, integrating the whole water column. Concentrations of nitrate-nitrogen (NO_3-N), nitrite-nitrogen (NO_2-N), total phosphorus (TP), orthophosphate (PO_4-P), soluble silica (Si), and sulfate (SO_4) were determined spectrophotometrically according to Strickland and Parsons (1972) and Technicon Industrial Methods (1977a, 1977b). Chlorophyll-*a* (Chl-*a*) was determined via extraction with 90% methanol spectrophotometrically (Youngman, 1978). The euphotic depth (z_{eu}) was calculated as 2.7 times the Secchi depth (Cole, 1994). The mixing layer depth (z_{mix}) was estimated as the maximum depths (z_{max}) of the lakes. The mixed layer to euphotic zone ratio (z_{mix}/z_{eu}) was used as a measure of light availability in the mixed layer (Jensen et al., 1994). Macrophyte coverage (MC) was monitored every month and recorded as a percentage area (%).

2.4. Data analysis

Mean values of the phytoplankton species richness, diversity, biomass, chlorophyll-*a*, FGs, and chemical and physical parameters of the two stations of each lake were used for statistical analyses. Analysis of variance (ANOVA) was applied to data for determining the statistical differences in species richness, diversity, biomass, chlorophyll-*a*, and *Q* index between the two lakes using SPSS 20.0. Pearson correlations between the physicochemical parameters and the species richness, diversity, biomass, chlorophyll-*a*, and *Q* index were also determined using SPSS 20.0. Redundancy analysis (RDA) was carried out using CANOCO software (Ter Braak and Šmilauer, 2002). In order to determine the relationships between the biomass of the functional groups, sampling periods, and environmental variables, RDA was carried out on the log-normal transformed abundance data. Statistical significance of the environmental predictor variables was assessed by 999 restricted Monte Carlo permutations.

3. Results

3.1. Environmental characteristics

In LT, the percentage of MC increased during spring and reached 30% in summer months. z_{mix}/z_{eu} values were measured as >2.0 during the studied period, except in spring, as shown in Table 1. pH values were low during spring and stayed above 8.6 during the rest of the year. EC and Secchi disk depth values were low during summer compared to other months. Low DO values were measured during spring, and the highest values were measured in winter. NO_3-N values were highest during winter and stayed around 0.3 mg L^{-1} during the studied

Table 1. The mean and standard deviation (SD) of environmental variables measured at the sampling sites in Lake Taşkısığı and Lake Little Akgöl. T: Water temperature, EC: specific electrical conductance, DO: dissolved oxygen, NO_3-N: nitrate-nitrogen, NO_2-N: nitrite-nitrogen, PO_4-P: orthophosphate, TP: total phosphorus, SO_4: sulfate, Si: soluble silica, Chl-*a*: chlorophyll-*a*.

	Lake Taşkısığı		Lake Little Akgöl	
Variable	Station 1	Station 2	Station 1	Station 2
	Mean ± SD	Mean ± SD	Mean ± SD	Mean ± SD
T (°C)	18.33 ± 7.62	18.05 ± 7.47	18.73 ± 8.32	19.1 ± 8.39
pH	8.52 ± 0.97	8.55 ± 0.17	8.42 ± 0.28	8.35 ± 0.29
EC (µS cm^{-1})	534.10 ± 28.61	537.82 ± 28.11	567.9 ± 152.03	547.02 ± 167.81
DO (mg L^{-1})	7.10 ± 4.18	8.91 ± 7.20	5.58 ± 2.15	5.24 ± 1.87
Secchi disk (cm)	69.91 ± 13.31	68.37 ± 16.61	56.25 ± 9.79	53.83 ± 13.78
NO_3-N (mg L^{-1})	0.27 ± 0.18	0.35 ± 0.24	0.17 ± 0.13	0.18 ± 0.22
NO_2-N (mg L^{-1})	0.009 ± 0.005	0.009 ± 0.004	0.017 ± 0.09	0.018 ± 0.02
PO_4-P (mg L^{-1})	0.020 ± 0.033	0.019 ± 0.021	0.15 ± 0.01	0.17 ± 0.14
TP (mg L^{-1})	0.040 ± 0.024	0.034 ± 0.022	0.17 ± 0.08	0.18 ± 0.10
SO_4 (mg L^{-1})	20.55 ± 33.45	19.89 ± 25.69	5.89 ± 7.83	5.68 ± 7.93
Si (mg L^{-1})	8.80 ± 6.07	10.55 ± 9.39	6.86 ± 3.56	6.73 ± 3.41
Chl-*a* (mg L^{-1})	0.002 ± 0.001	0.006 ± 0.005	0.011 ± 0.01	0.011 ± 0.01
z_{mix}/z_{eu}	2.5 ± 0.46	2.6 ± 0.45	0.88 ± 0.29	0.91 ± 0.30

period. NO_2-N values reached the highest values during winter; however, a sudden increase was also observed in the middle of summer. PO_4-P values reached 0.06 mg L^{-1} in spring and stayed around 0.015 mg L^{-1} during the summer months. A second peak was observed (0.077 mg L^{-1}) in the beginning of fall. The highest TP and SO_4 values were also measured in the beginning of fall. Si values reached 36.2 mg L^{-1} in summer and stayed below 20 mg L^{-1} during the rest of the year.

In LLA, the percentage of MC increased during spring and reached 60% in the summer months. As shown in Table 1, z_{mix}/z_{eu} values were generally measured as <1.0 during the studied period; however, spring and summer values were higher (>1.1). pH values were higher (above 8.8) during summer and stayed around 8.3 during the rest of the year. EC values were higher during spring and summer and stayed below 500 µS cm^{-1} during the studied period. DO levels were highest during winter. Secchi disk depth values were low during summer compared to other months. NO_3-N values reached 0.62 mg L^{-1} in spring and were generally low during summer. NO_2-N values were highest during late summer and early fall. PO_4-P values reached 0.37 mg L^{-1} in spring and stayed around 0.018 mg L^{-1} during the summer months. PO_4-P values were generally high during winter months. TP values were also measured highest (0.43 mg L^{-1}) in spring. It slowly decreased during late spring, and a second peak was observed in summer. TP values stayed below 0.15 mg L^{-1} during the rest of the year. SO_4 values were highest during winter. Si values were lower during spring and summer compared to other months.

3.2. Phytoplankton

A total of 130 taxa were grouped in 17 FGs in LT and a total of 105 taxa were grouped in 15 FGs in LLA. As shown in Figure 2, Bacillariophyta and Chlorophyta dominated at least once during the sampling period in both of the lakes; however, Cyanobacteria also dominated during late summer in LT and Cryptophyta during winter months in LLA. As shown in Figure 3, 7 FGs (MP, P, J, B, X2, W2, M) constituted >10% of the total biomass of at least one sample in LT, while 7 FGs (MP, P, J, C, X2, F, X1) constituted >10% of the total biomass of at least one sample in LLA.

In LT, the dominancy of main groups arose during 4 different periods (Period I: December to January, Period II: February to May, Period III: June to August, Period IV: September to November). Codon B (*Cyclotella ocellata* Pantocsek and *Stephanodiscus neoastraea* Håkansson & Hickel, 64.5% of biomass) prevailed among the phytoplankton in Period I, while group J (*Desmodesmus* spp., *Scenedesmus* spp., *Pediastrum* spp., *Coelastrum microporum* Nägeli, 50.2% of biomass) dominated in Period II. *Staurastrum anatinum* Cooke & Wills, *Staurastrum*

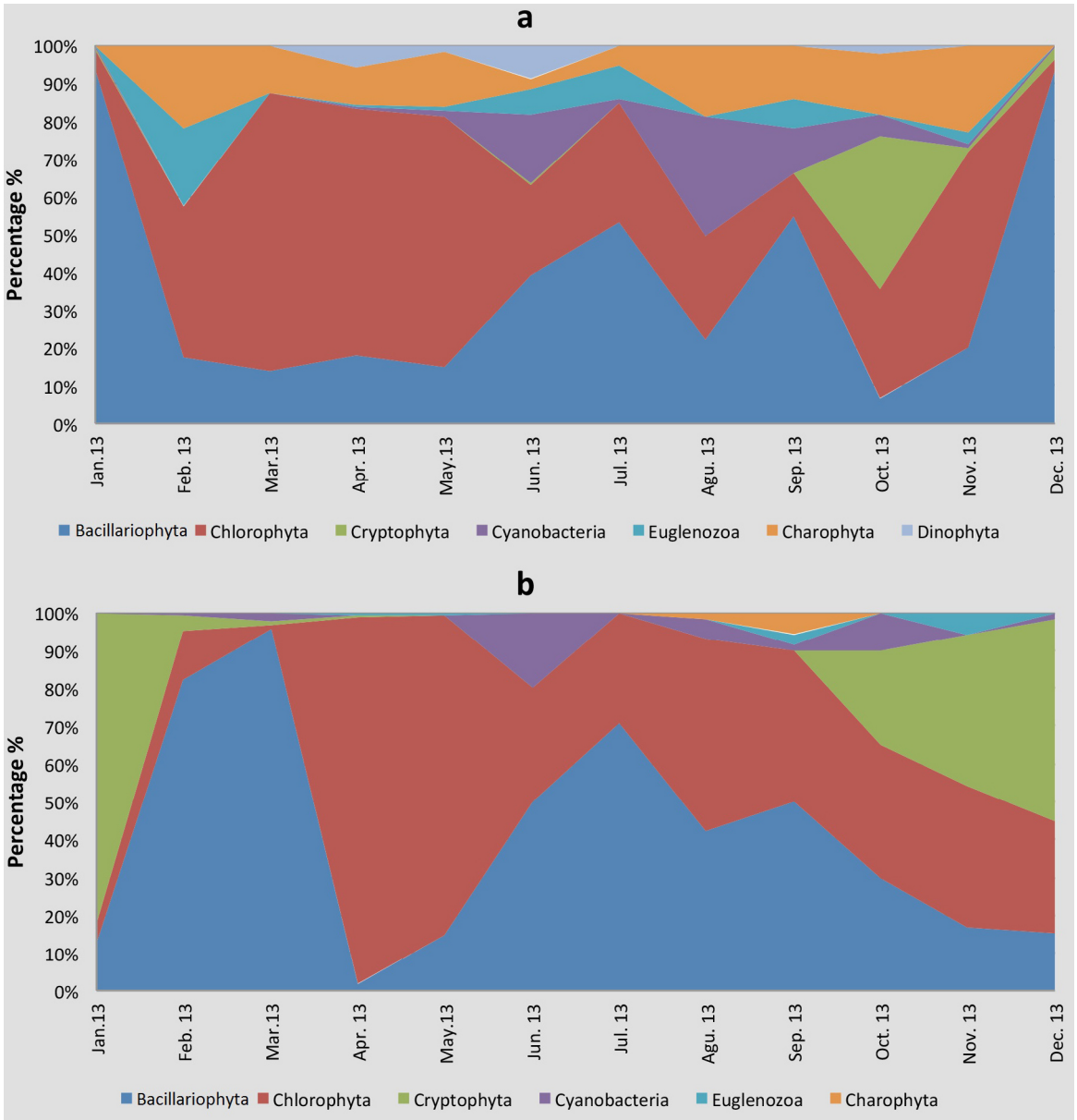

Figure 2. Relative frequency (%) of each phytoplankton taxonomical group according to biomass values a) in Lake Taşkısığı, b) in Lake Little Akgöl.

cingulum (West & G.S.West) G.M.Smith, and *Staurastrum crenulatum* (Nägeli) Delponte, belonging to codon P, were also abundant (14.5% of biomass) during Period II. Moreover, members of codon W2 [*Strombomonas* sp. and *Trachelomonas volvocina* (Ehrenberg) Ehrenberg var. *punctata* Playfair] reached 20% of biomass in the beginning of Period II. During Period III, the biomass of groups P [*Aulacoseira granulata* (Ehrenberg) Simonsen and *Aulacoseira subarctica* (Otto Müller) E.Y.Haworth, 26% of biomass] and J (*Pediastrum* spp., 19% of biomass) were high. Moreover, *Microcytis wesenbergii* (Komarek) Komarek (Group M) reached 14.6% of biomass in the end of Period III. Groups J (*Pediastrum* spp. and *Coelastrum astroideum* De Notaris, 30% of biomass) and P (*S. anatinum*, *S. cingulum*, *S. crenulatum, A. granulata*, and *A. subarctica*, 26.3% of biomass) were abundant during Period IV. However, groups X2 (*Cryptomonas pyrenoidifera* Geitler and *Plagioselmis nannoplanctica* (Skuja) Novarino, Lucas & Morrall) and MP (*Fragilaria tenera* (W.Smith) Lange-Bertalot) were important components of phytoplankton during different months of Period IV.

In LLA, the dominance of the main groups arose during 4 different periods (Period I: November to January, Period II: February to March, Period III: April

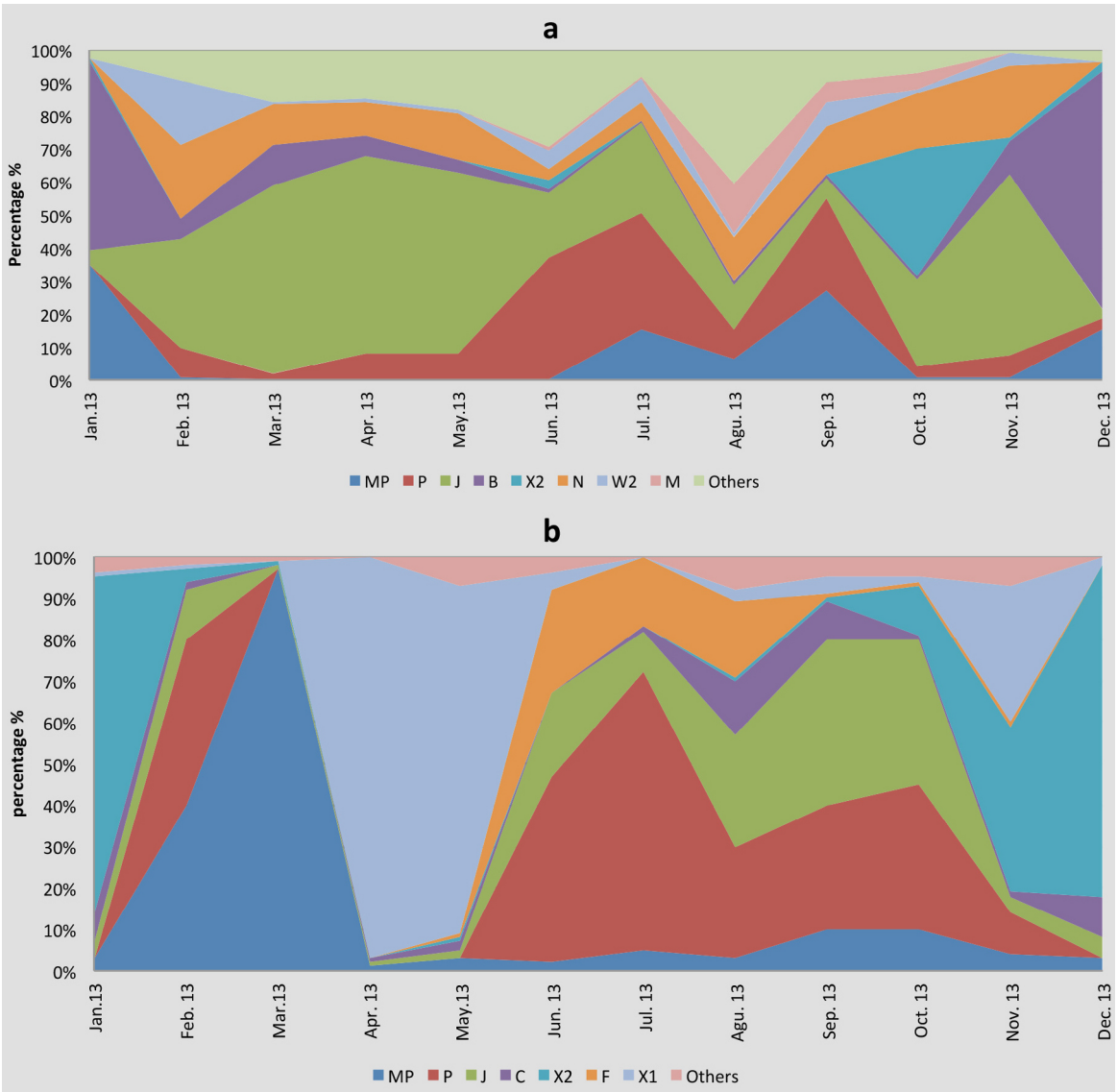

Figure 3. Relative frequency (%) of dominant phytoplankton functional groups according to biomass values a) in Lake Taşkısığı, b) in Lake Little Akgöl.

to May, Period IV: June to October). During Period I, small flagellates belonging to codon X2 (*Cryptomonas* spp., *Chroomonas* spp., and P. *nannoplanctica*, 67% of biomass) dominated the phytoplankton assemblage accompanied by X1 [*Monoraphidium minutum* (Nägeli) Komárková-Legnerová] in the beginning of Period I. Group MP [*Navicula cryptocephala* Kützing var. *veneta* (Kützing) Rabenhorst, *Navicula rhynchocephala* Kützing, and *Nitzschia recta* Hantzsch ex Rabenhorst, 68.5% of biomass] prevailed among the phytoplankton in Period II, while group X1 (*Chlorella* sp., 90% of biomass) took the dominancy in Period III. During Period IV, the biomass of groups P (*A. granulata*, 41% of biomass) and J (*Desmodesmus* spp., *Scenedesmus* spp., *Pediastrum* spp., 27% of biomass) was high. The contribution of the large, colonial mucilaginous species that belonged to codon F [*Oocystis borgei* J.W.Snow and *Golenkiniopsis solitaria* (Korshikov) Korshikov, 20% of biomass] and centric diatoms in codon C (*Cyclotella meneghiniana* Kützing, 10% of biomass) was also important in Period IV.

Species richness ranged between 11 and 41 in the lakes during the studied periods, as shown in Figure 4. The Shannon diversity index generally ranged between 1.0 and 2.5 in LT, while it was between 0.5 and 2.5 in LLA. The lowest diversity and evenness values of LLA were measured in spring. These were mainly due to an increase of *Chlorella* sp. (group X1), which represented 97% of the total phytoplankton biomass. Species richness and

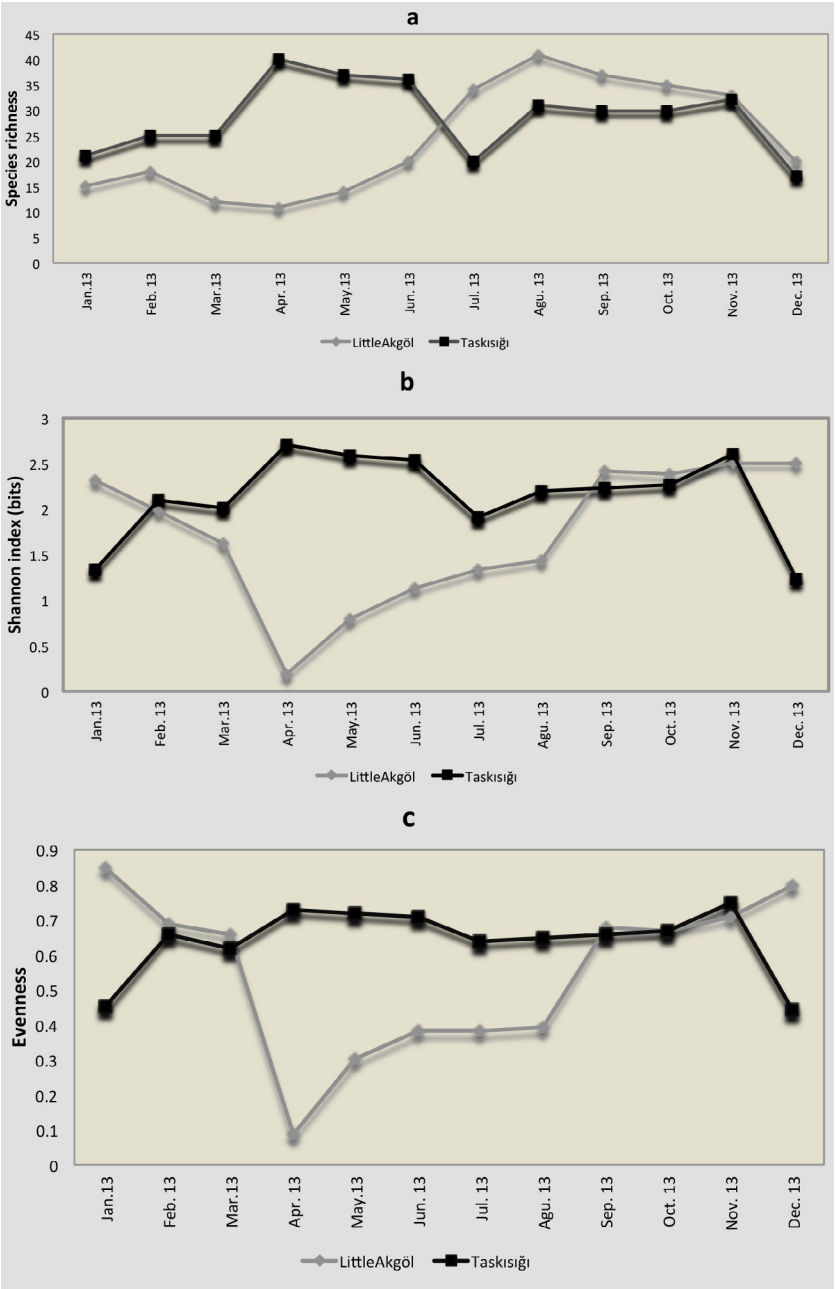

Figure 4. Temporal variations of a) species richness, b) Shannon–Weaver diversity index (*H*), and c) evenness (*E*) in Lake Taşkısığı and Lake Little Akgöl.

Shannon index were not significantly different between the two lakes ($P > 0.05$). Phytoplankton biomass ranged from 1.5 mg L^{-1} to 20 mg L^{-1} in LT, while it ranged from 0.1 mg L^{-1} to 52.4 mg L^{-1} in LLA. Biomass values were highest in spring and decreased during the early summer in LT. A second peak was observed in the beginning of fall. Biomass values of LLA increased during spring and early summer. Maximum values were observed in summer, and then a sharp decrease was seen in late summer. Biomass values were not significantly different between the two lakes ($P > 0.05$). In LT, Chl-*a* values were low during winter and began to increase during spring. The highest values were recorded during summer. In LLA, Chl-*a* values were lower during winter and spring and began to increase during summer. Highest values were recorded during fall. Chl-*a* values were significantly different between the two lakes ($f = 8.73$, $P < 0.05$). As shown in Table 2, phytoplankton diversity, biomass, and Chl-*a* were significantly correlated

Table 2. Pearson correlation coefficients between environmental variables and species richness, Shannon diversity index, and *Q* index in Lake Taşkısığı (LT) and Lake Little Akgöl (LLA). * P<0.05; ** P<0.01. Only significant correlations are shown.

	Richness		Shannon diversity index		Biomass		Chlorophyll-*a*		*Q* index	
	LT	LLA	LT	LLA	LT	LLA	LT	LLA	LT 1	LLA
Temperature	0.45**	–	0.51**	−0.50**	–	–	0.45**	–	−0.49**	−0.45*
Z_{mix}/Z_{eu}	−0.76**	–	−0.68**	–	–	–	–	–	0.47**	–
Specific conductance	–	−0.42*	–	−0.64**	–	–	−0.34*	−0.78**	–	–
Dissolved oxygen	−0.59**	–	−0.75**	0.59**	–	–	–	–	0.87**	–
Secchi disk	0.48**	−0.68**	0.42*	–	–	–	−0.39*	–	−0.39*	–
Nitrate-nitrogen	−0.47**	–	−0.51**	–	–	–	–	–	0.57**	0.74**
Nitrite-nitrogen	−0.72**	0.60**	−0.76**	–	–	–	−0.33*	–	0.56**	–
Orthophosphate	0.35*	−0.55**	–	–	–	–	–	–	–	–
Total phosphorus	–	–	–	−0.63**	–	0.49*	–	−0.43*	–	−0.46*
Sulfate	0.35*	–	–	–	–	–	–	–	–	–
Silica	–	–	–	–	–	–	–	0.58**	–	–
pH	–	–	–	–	−0.34*	–	–	−0.41*	–	–
Macrophyte coverage	0.49**	0.45**	0.52**	−0.38*	–	–	0.44*	–	−0.36*	−0.58**

with most of the environmental factors, such as EC, DO, and nutrients.

Factor F weights of each functional group identified in LT and LLA are given in Table 3. As shown in Figure 5, the *Q* quality index based on FGs generally varied between 2 and 4 (medium to good), and it was higher during winter in both of the lakes. Values were slightly higher in LT than LLA during the studied period ($f = 11.13$, $P < 0.05$). As shown in Table 2, the *Q* quality index was significantly correlated with some parameters, such as z_{mix}/z_{eu}, DO, and nutrients.

To analyze the relationship between phytoplankton distribution and environmental variables, we performed RDA using biomass values of the 10 dominant FGs in both of the lakes. RDA was performed initially on the whole environmental and FG datasets. Forward selection indicated that 6 of the 13 environmental variables made significant contribution to the variance in the FG data. The results of RDA using only these 6 variables are illustrated in Figure 6. The eigenvalues of RDA axis 1 (0.23) and axis 2 (0.17) account for 39.4% of the cumulative variance in the FG data. The FGs–environmental correlations of RDA axes 1 and 2 are high and the first two axes account for 73.3% of the variance in the FGs–environmental relationships. LT was positively correlated with z_{mix}/z_{eu}, NO_3-N, and DO while LLA was positively correlated with PO_4-P, TP, and MC. Group B was positively correlated with DO and NO_3-N; M, W2, J, and P with z_{mix}/z_{eu}; MP with DO; X2 with PO_4-P and DO; X1 and C with TP; and F with MC.

4. Discussion

Although some studies (Topkara and Balık, 2010; Tavşanoğlu et al., 2015) were done about the limnology and biodiversity of the studied lakes, only two studies have been reported about the algal abundance and species composition in LT (Aykulu et al., 1999; Temel and Yardımcı, 2004). In LT, a total of 130 taxa have been identified during our study, while Temel and Yardımcı (2004) recorded 64 taxa during a study conducted between 1998 and 1999. Moreover, seasonal distributions of the main phytoplankton taxonomical groups and dominant species were different between these two studies. PO_4-P and Chl-*a* values did not change and the lake remained eutrophic in both studies.

Several studies were done to describe the relationship between principal macronutrients (TN and TP) and phytoplankton biomass (Vollenweider and Kerekes, 1982; Havens and Nurnberg, 2004; Phillips et al., 2008). In LT, we did not find any relationships between TP, Chl-*a*, and phytoplankton biomass values. Although Chl-*a* values did not show a linear correlation, phytoplankton biomass values were linked linearly to TP values in LLA. For this reason, we can say that there is a linear effect of TP on phytoplankton biomass in eutrophic LLA. A slight increasing tendency in phytoplankton biomass, due to the increasing nutrients, can be expected even in eutrophic conditions (Borics et al., 2013).

Borics et al. (2014) showed that lake depth and lake size are crucial in shaping phytoplankton biomass in

Table 3. Factor F for the phytoplankton functional groups of Lake Taşkısığı (LT) and Lake Little Akgöl (LLA).

Functional group	Factor F LT	Factor F LLA
B	4	3
C	3	3
F	4	3
J	2	2
K	3	3
Lo	2	2
Lm	1	-
M	0	0
MP	4	4
N	-	2
S1	0	-
S2	0	-
P	4	2
T	4	4
W1	1	2
W2	3	3
X1	1	2
X2	3	3

shallow lakes. The sizes of the studied lakes are smaller than 1.0 km^2, so we did not compare the lakes based on lake size. The mean depths of our lakes were 0.5 and 1.5 m, respectively. Lake depth directly affected the z_{mix}/z_{eu} values; consequently, z_{mix}/z_{eu} values were higher in LT than LLA during the studied period. The phytoplankton functional groups are strongly affected by z_{mix}/z_{eu} as indicated in the RDA analysis, and groups W2, M, J, and P occurred when the z_{mix}/z_{eu} values were higher in LT. Algae in these groups prefer shallow, vertically mixed mesoeutrophic systems (Reynolds et al., 2002; Padisák et al., 2009). Moreover, members of group M (*Microcytis wesenbergii*) were generally defined in low light environments in various lakes of the world (Brookes and Ganf, 2001). We found a strong constraint of TP and PO_4-P in shaping phytoplankton functional groups of X1, X2, and C in LLA. Naselli-Flores (2000) found that, in 21 Sicilian reservoirs, as trophic state increases, physical parameters, especially light climate, become more important in promoting the development of a specific phytoplankton assemblage than nutrient availability. In our study, LT is slightly in the lower part of the trophic spectrum than LLA; however, light availability, expressed as z_{mix}/z_{eu}, had important effects on phytoplankton assemblages (W2, M, J, P). On the other hand, nutrient availability, expressed as TP and PO_4-P, exhibited important consequences in shaping phytoplankton assemblage (X1, X2, and C) in more eutrophic LLA.

Studies revealed that there is an inverse relationship between MC and phytoplankton biomass (Dokulil and Teubner, 2011; Borics et al., 2014). It was found that 30% cover of the lake surface area by macrophytes is necessary to shift turbid water to a clear water state in shallow lakes (Jeppesen et al., 1994). Macrophyte coverage reached 30% in LT and 60% in LLA during summer; however, we did not find negative correlations, thereby reducing the effect of MC, neither with biomass nor Chl-*a* values. In the presence of macrophytes, dominance of flagellated algae is expected (Borics et al., 2003; Krasznai et al., 2010). We did not observe the dominancy of flagellated algae in either

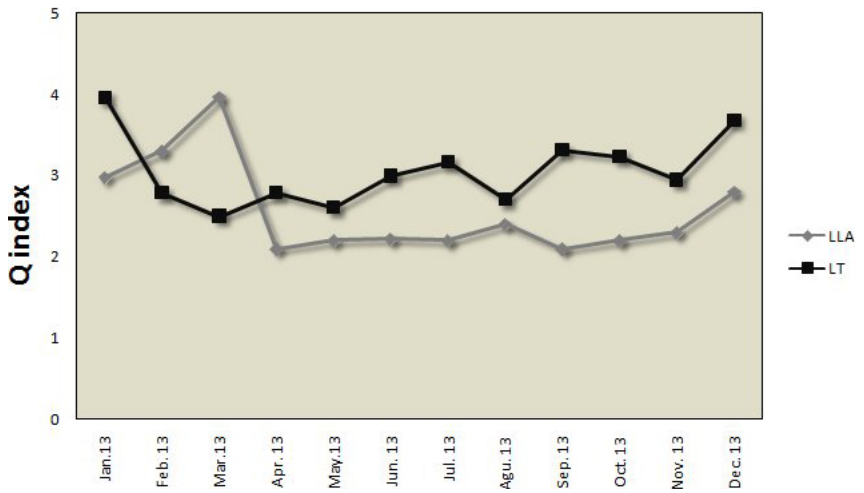

Figure 5. The seasonal differences of the Q index for Lake Taşkısığı (LT) and Lake Little Akgöl (LLA).

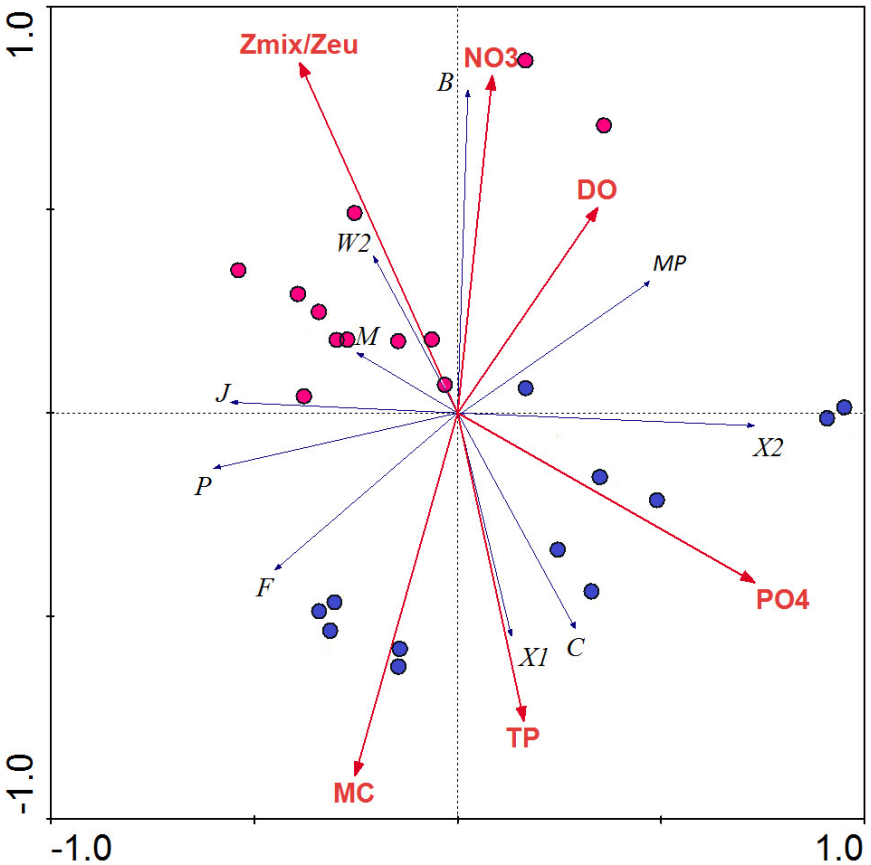

Figure 6. Ordination of the samples corresponding to the different sampling periods and lakes, scores of phytoplankton biomass by functional groups, and environmental variables along the redundancy analysis axes. Environmental variables: MC: macrophyte coverage, z_{mix}/z_{eu}: ratio of mixing and euphotic layers, NO3: nitrate-nitrogen concentration, DO: dissolved oxygen concentration, PO4: orthophosphate concentration, TP: total phosphorus concentration (pink: Lake Taşkısığı, blue: Lake Little Akgöl).

of the lakes; however, group F in LLA was the important component of phytoplankton in the period when the MC values were higher**.** The habitat template described for this functional group is clear, deeply mixed mesoeutrophic lakes (Reynolds et al., 2002; Padisák et al., 2009). Demir et al. (2014) also found summer increase of group F in eutrophic Lake Mogan.

In our study, taxonomical groups belonging to codons J, MP, P, and X2 were important components of phytoplankton in both of the studied lakes. Groups P and J were dominant when the z_{mix}/z_{eu} values were higher during summer in both lakes. Similarly, group MP was found when DO values were higher and group X2 was found when the PO_4-P values were higher during fall and winter in both lakes. The habitat templates for these groups are shallow, mesoeutrophic lakes (Reynolds et al., 2002; Padisák et al., 2009). However, group B, which was dominant when NO_3-N concentrations were higher, and groups W2 and M, which were important components of phytoplankton when z_{mix}/z_{eu} values were higher, were prominent FGs in LT. Moreover, groups C and X1, which were dominant when TP concentrations were higher, and group F, which was an important component of phytoplankton when MC values were higher, were prominent FGs in LLA. For this reason, the differences in $z_{mix}/z_{eu,}$ NO_3-N, TP, and MC values in the two lakes reflected the differences in dominant functional groups.

The diversity (*H*) values of LT and LLA generally ranged between 1 and 2.5, similar to eutrophic Danish lakes (Jeppesen et al., 2000). However, in LLA, *Chlorella* sp. (X1) formed 90% of the biomass during spring when TP values were high, and consequently diversity values decreased to 0.5. Although the trophic state is the most affective factor, diversity is also influenced by lake size, lake depth (Jeppesen et al., 2000), or macrophytes (Declerck et al., 2007). Other nutrients such as NO_2-N, NO_3-N, and PO_4-P; lake depth, which was affected by z_{mix}/z_{eu}; and EC, DO, and macrophyte coverage also influenced the Shannon

diversity and species richness during the studied periods; however, a significant difference was not found in diversity values between the two lakes. This can be expressed by the high number of species with similar habitat template in eutrophic waters (Reynolds, 1998). Therefore, the same 14 FGs were found in both lakes.

The average *Q* index was found as 3.05 for LT and 2.56 for LLA, which reflect good and medium water qualities, respectively. In LT, good ecological quality was observed during Periods I and IV when the highest contributions of B, P, X2, and MP were found. Index values decreased to medium ecological quality during Periods II and III. The contribution of FGs with low factor F weights, such as the groups J and M, were responsible for medium ecological status. In LLA, medium to good ecological qualities were observed during Periods I and II with the highest contribution of X2 and MP. During Periods III and IV, the selection of X1 and C under high TP availability and also P and J provided lower index values (~2.2). It seems that the *Q* index is reliable for assessing the quality of LT and LLA. However, the key factor of the method developed by Padisák et al. (2006) is the exact determination of the factor F corresponding to a given lake. Therefore, misidentification of dominant species or wrong determination of factor F can lead to wrong quality status (Becker et al., 2009).

In conclusion, this study compares phytoplankton structure on the basis of environmental variables in two Mediterranean shallow lakes in the north of Turkey. We showed that TP values were vital in shaping phytoplankton biomass in eutrophic LLA. NO_3-N, TP, z_{mix}/z_{eu}, and MC are important in shaping the differences in dominant phytoplankton functional groups between the two lakes. Generally, diversity values were similar between the two lakes and environmental parameters had no effect on diversities to constitute significant differences. Assessment using the *Q* index gave compatible results for determining the ecological status of LT and LLA.

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