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The phytoplankton functional group concept provides a reliable basis for ecological status estimation in the Çaygören Reservoir (Turkey)

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Abstract: The phytoplankton functional group (FG) concept has been successfully used to determine water quality states in both lake and river systems, but is not yet stressed in regular studies in reservoir environments. The purpose of our study was to estimate the ecological status of the temperate Çaygören Reservoir (2007–2009) using the $Q_{(t)}$ FG composition metric, and to test whether it is able to follow the main structuring of the physical and chemical parameters studied. The main occurring coda were codon C (*Cyclotella meneghiniana* Kützing, *Stephanodiscus neoastraea* Hakansson and Hickel), codon T (*Gloeotila subconstricta* (G.S. West) Printz, *Mougeotia* sp.), codon X2 (*Cryptomonas pyrenoidifera* Geitler, *Plagioselmis nannoplanctica* (H.Skuja) G.Novarino, I.A.N.Lucas and S.Morrall), codon K (*Aphanocapsa holsatica* (Lemmermann) G.Cronberg and J.Komárek, *Aphanothece clathrata* West and G.S.West), and codon S1 (*Planktothrix* sp.). The seasonal succession of phytoplankton displayed the lake-like coda series of Y (winter) → C (spring) → K/S1 (summer) → T (fall) in each year, which demonstrated well the transitional behavior of reservoirs between lake and river ecosystems. The FG compositions showed high relationships with water temperature, ammonia, phosphate, and light availability in this temperate reservoir, where the $Q_{(t)}$ index estimated reliable water quality states between hypertrophic and oligotrophic conditions.

Key words: Ecological status, phytoplankton functional classification, temperate reservoir, water quality

1. Introduction

In northern temperate lakes, phytoplankton succession is largely determined by the dynamics of physical, chemical, and biological factors, the relative importance of which varies with the different periods of the year. The phytoplankton taxa have been identified along different spatio-temporal scales, which highlighted special structural characteristics in the phytoplankton composition (Piiroo et al., 2008, 2010; Borics et al., 2011; Wu et al., 2011).

Besides the traditional taxonomical approach, the functional classification of phytoplankton describing a number of species having similar seasonal sequences (each functional group (FG) includes functionally similar phytoplankton taxa that might be occurring under similar set of environmental conditions) has lately been widely used to explore the relationships between phytoplankton species and environmental variables (Reynolds et al., 2002).

There is an increasing demand for new assessment methods for evaluating the ecological status of lakes and rivers (Borics et al., 2007). Phytoplankton FGs have been widely used as a tool for studying phytoplankton ecology

in lakes, reservoirs, and rivers (Devercelli, 2006; Caputo et al., 2008; Piiroo et al., 2008).

In the phytoplankton functional system, several phytoplankton species may potentially dominate or co-dominate in a given environment. These groups share adaptive features, based on the physiological, morphological, and ecological attributes of the species (Reynolds et al., 2002). The FGs approach uses alpha-numeric codes according to the groups' sensitivities and tolerances (Padisak et al., 2009). The concept assumes that a functionally well-adapted species is likely to tolerate the constraining conditions of factor deficiency more successfully than a less well-adapted species and a habitat typically constrained by physical or chemical variables is more likely to support species with the appropriate adaptations to function there.

Padisak et al. (2009) modified the original functional classification system by reviewing numerous studies dealing with the FG concept and described typical misplacements by authors. It then was applied in numerous studies (Becker et al., 2010; Devercelli, 2010; Piiroo et al., 2010; Wang et al., 2011; Wu et al., 2011; Xiao et al., 2011;

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Abonyi et al., 2012, 2014; Hu and Xiao, 2012; Stanković et al., 2012; Scomparin et al., 2013).

The original idea of the phytoplankton functional group concept (Reynolds et al., 2002) was proposed as a new ecological status estimation method for lake phytoplankton (Q index; Padisak et al., 2006) and then for rivers ($Q_{(r)}$ index; Borics et al., 2007). In the present study, we used the river phytoplankton assemblage index ($Q_{(r)}$) instead of the lake index (Q) because no lake type could be selected as the “reference” to evaluate the ecological status of the reservoir. The $Q_{(r)}$ index relies on the fact that phytoplankton composition is highly related to physical constraints (Devercelli, 2010; Wu et al., 2011; Hu and Xiao, 2012). Besides physical factors, nutrient levels also determine the relevant phytoplankton assemblages, altogether exhibiting quite similar dynamics in rivers and lakes (Reynolds et al., 1994).

Borics et al. (2007) elaborated the lake assemblage method based on the phytoplankton associations for rivers. In this method, all phytoplankton associations were evaluated and scored by a factor number between 0 and 5. As many rivers can be defined as shallow, turbid, mesotrophic ecosystems of short residence time, the groups that prefer this type of environment were given high factor numbers, and the groups that are typical of stable hypertrophic lakes have the lowest values. While the Q index based on lentic phytoplankton functional groups (Reynolds et al., 2002; Padisak et al., 2009) needs a detailed description, there is no need for typology-related reference conditions in rivers (Borics et al., 2007; Mischke et al., 2011). The river assemblage index $Q_{(r)}$ is characterized by a compound factor (F) ranging from 0 to 5, estimated from trophic state, turbulence, and residence

time needed for the occurrence of each algal functional group, and possible risk of presence of the FG (Borics et al., 2007). The explanation for the F factor component is as follows: trophic state (hypertrophic 0 to oligotrophic 5), turbulence (totally standing waters 0 to highly lotic 5), residence time (climax assemblages 0 to benthic codons 5), and expert opinion (varies from high risk 0 to low risk 5). The expert opinion expresses how the occurrence of the given assemblage reflects pollution or toxicity in the ecosystem. The designated values of each element are then summed and provide a water quality state from 0 (low) to 5 (high). Accordingly, the lowest value of $Q_{(r)}$ is characteristic for lentic phytoplankton taxa dominance of stagnant to slow flowing conditions, while the highest value is characteristic of benthic diatoms dominance of highly lotic habitats (Pirsoo et al., 2010).

The purpose of this study was to assess the ecological status of the temperate Çaygören Reservoir through the application of the river phytoplankton assemblage index, $Q_{(r)}$, and to explore the relationships between structuring in the phytoplankton functional groups and the main physicochemical parameters.

2. Materials and methods

2.1. Study area

Çaygören Reservoir is located at 39°17'24"N; 28°19'16"E, 55 km southeast of Balıkesir, Turkey (Figure 1). It lies at 273 m above sea level. It has a maximum depth of 28 m, a length of 4.6 km, and a surface area of 9 km². The Simav Stream feeds the reservoir. The reservoir was built in 1971 and it is used for irrigation and power generation (Devlet Su İşleri, 2005).

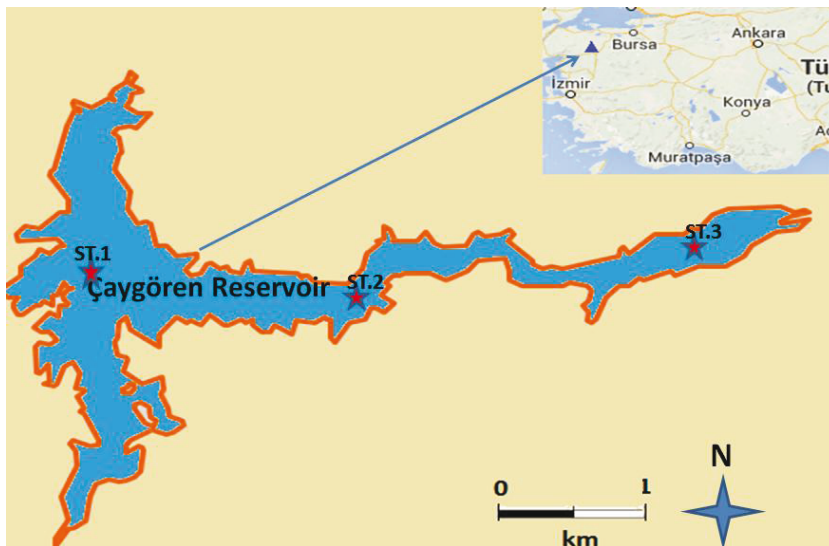


Figure 1. The map of the Çaygören Reservoir and the location of sampling stations.

2.2. Field sampling procedure

Sampling was carried out monthly at three stations between February 2007 and January 2009. The three sampling stations were set up to cover all parts of the reservoir. The first station was set up near the main inlet, the second one in the middle of the reservoir (transition zone between lotic and lentic conditions), and the third one near the dam at the deepest part of the reservoir. Samples were drawn vertically at 5-m intervals using a Kemmerer water sampler.

In situ parameters including pH, oxidation-reduction potential (ORP), and water temperature were measured at 5-m intervals using a YSI multiprobe. Water transparency was measured using a Secchi disk. Water discharge data were obtained from the State Water Works. Samples for phytoplankton were taken from at 1-m depth intervals in the euphotic layer, then one sample at the mid depth, and another at the bottom of the each station. The samples were placed in 250-mL bottles and fixed with Lugol's solution until processed in the laboratory.

2.3. Data processing

Concentrations of soluble reactive phosphorus (SRP), nitrate-nitrogen ($\text{NO}_3\text{-N}$), and ammonium-nitrogen ($\text{NH}_4\text{-N}$) were determined from 1, 5, and 15 m, by filtering a known volume of water through Whatman 934-AH filters, and then analyzed by spectrophotometry according to standard methods (APHA, 1995).

The euphotic zone (Zeu) was calculated as 2.7 times the Secchi disk depth (Cole, 1994). The depth of the mixing layer (Zmix) was estimated from individual temperature profiles. Euphotic zone to mixed layer (zeu:zmix) ratio was used as a measure of light availability in the water column (Jensen et al., 1994).

For phytoplankton data, samples were poured into 50-mL graduated cylinders and were allowed to settle for 24 h. After that, 45 mL of water was aspirated from each graduated cylinder and the remaining 5 mL was poured into a small glass vial for microscopic analysis (APHA, 1995). The samples were examined under an Olympus BX51 compound microscope.

Phytoplankton species were identified according to Huber-Pestalozzi (1983), Bourrelly (1966), Krammer and Lange-Bertalot (1991), Komarek and Anagnostidis (2008), Round et al. (1990), Sims (1996), Komarek et al. (1998), and John et al. (2002). Taxonomical names were updated based on Guiry and Guiry (2013).

Phytoplankton biomass was calculated from biovolume data, assuming a specific gravity of one. Biovolume was calculated from cell numbers and individual cell size measurements, using specific geometrical forms (Sun and Liu, 2003). Species contributing more than 5% to the annual average of the total phytoplankton biomass were grouped into functional groups, using the criteria

of Reynolds et al. (2002). The $Q_{(r)}$ index following Borics et al. (2007) and Padisák et al. (2009) was calculated on the average biomass of the phytoplankton species from the euphotic/mixed layer combined.

Canonical correspondence analysis (CCA) was used to examine the relationships between the environmental variables and phytoplankton functional groups. The significance of environmental variables to explain the variance in functional groups used in the CCA was assessed by 999 restricted Monte Carlo permutations. The CCA and permutation tests were performed using the program CANOCO v.4.5 (ter Braak and Smilauer, 2002). When assumptions of normality were not satisfied, variables were log transformed.

3. Results

The light availability, expressed by the zeu/zmix ratio, was sufficient for phytoplankton growth during the spring, reaching up to 60% in the mixing zone (Figure 2). The maximum inflow ($1300 \text{ m}^3 \text{ s}^{-1}$) to the reservoir was recorded in April 2007 and the minimum inflow ($5.2 \text{ m}^3 \text{ s}^{-1}$) was measured in September 2007 (Figure 3).

Surface water temperature ranged from $4.5 \text{ }^\circ\text{C}$ to $26.6 \text{ }^\circ\text{C}$ at all stations. Maximum surface water temperatures were measured in June and July and minimums were

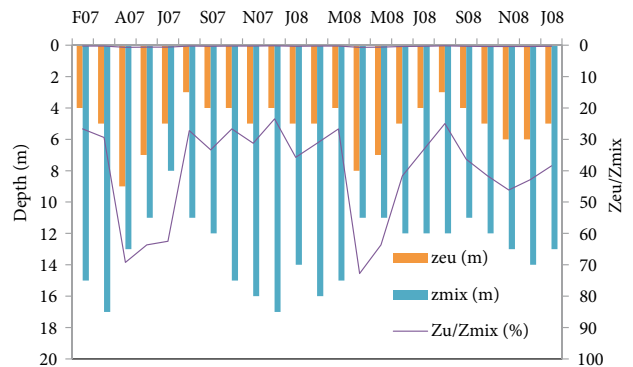


Figure 2. Seasonal variation of the mixing zone (Zmix), euphotic zone (Zeu), and the Zeu/Zmix ratio in the Çaygören Reservoir.

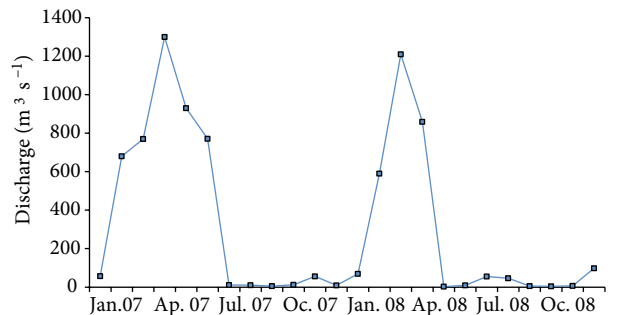


Figure 3. Monthly average of water flow from Simav Stream to Çaygören Reservoir from January 2007 to December 2008.

measured in February. When the reservoir was stratified from May to October, hypolimnetic temperatures varied from 21 °C to 15 °C, metalimnetic temperatures varied from 21 °C to 23 °C, and epilimnetic temperatures varied from 25 °C to 26 °C. Temperature profiles allowed us to identify only one period of total vertical mixing, which occurred during winter (Figure 4).

pH ranged from 7.4 to 11.6 and it stayed around 9 throughout the study period at all stations. A decline to about 7 occurred in pH in November 2008 and another one in January 2009. There were no apparent differences between the depths and the stations in the pH values (Figure 5). In the winter months, ORP decreased from surface to bottom and vice versa in the summer at all stations. ORP values decreased from the first station to the

third station. During the summer time, at the third station (the deepest), ORP values dropped from about 70 mV to 2 mV; at the shallower first and second stations it was about 65 mV at upper depths and about 25 mV at the deeper depths. During the winter, spring, and fall, there were no apparent differences between the depths (Figure 6).

At the first station, soluble reactive phosphorus (SRP) concentrations ranged from 0.005 mg L⁻¹ to 0.032 mg L⁻¹ and were not different between depths. At the second station, SRP concentrations ranged from 0.005 mg L⁻¹ to 0.03 mg L⁻¹ and were different between depths. They ranged from 0.0065 mg L⁻¹ to 0.059 mg L⁻¹ at 15 m. At the third station, SRP concentrations ranged from 0.005 mg L⁻¹ to 0.034 mg L⁻¹ at the surface and from 0.005 mg L⁻¹ to 0.059 mg L⁻¹ at 15 m (Figure 7).

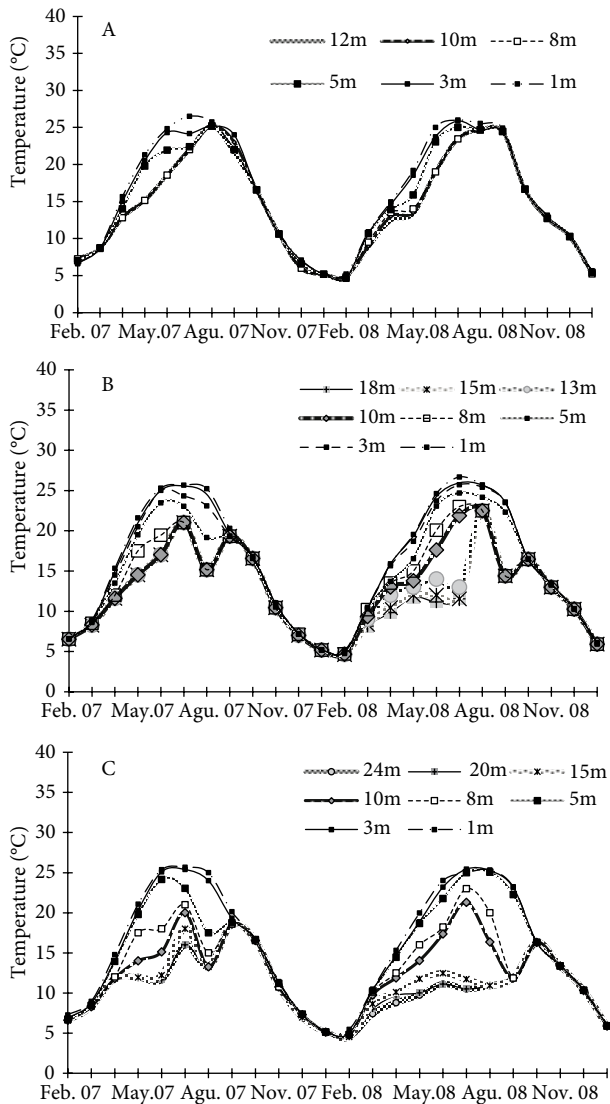


Figure 4. The water temperature (°C) profiles of the Çaygören Reservoir (January 2007–December 2008).

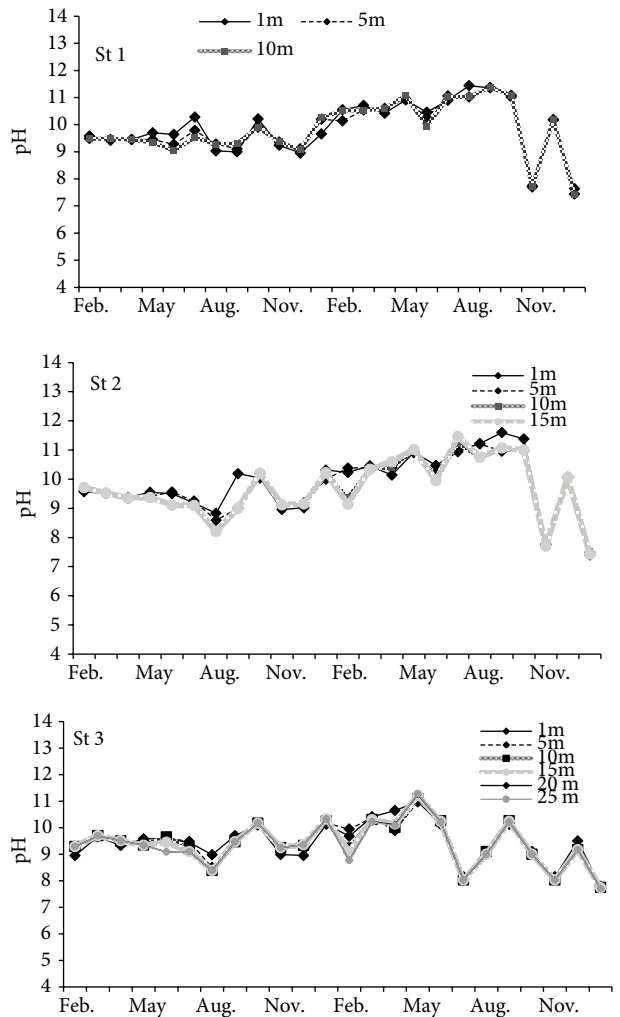


Figure 5. The seasonal variations in the pH of the Çaygören Reservoir (January 2007–December 2008).

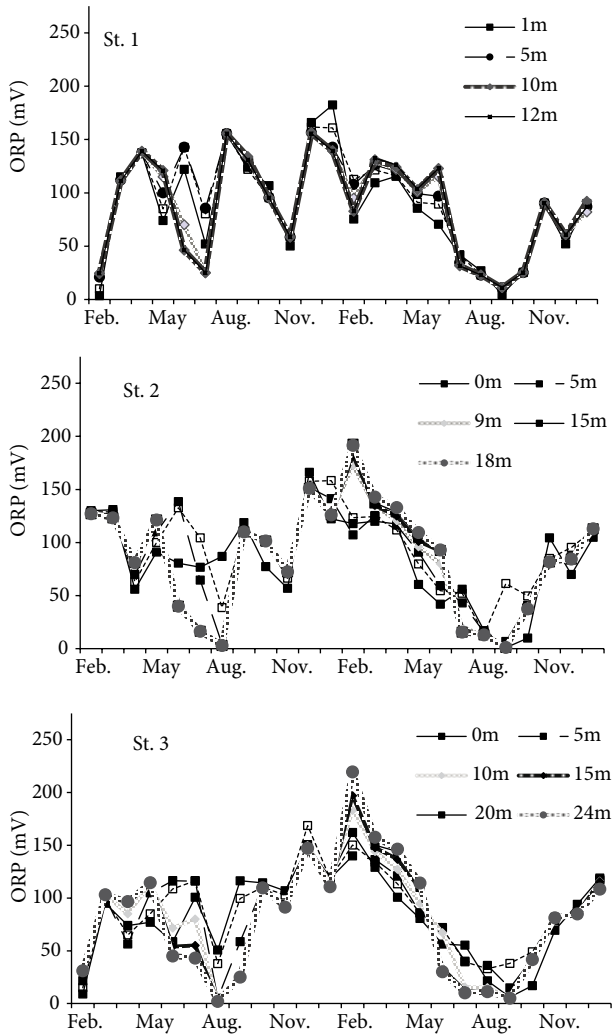


Figure 6. The seasonal variations in oxidation-reduction potential (ORP, mV) of the Çaygören Reservoir (January 2007–December 2008).

At the first station, $\text{NO}_3\text{-N}$ concentrations were different between depths and they ranged from 0.059 mg L^{-1} to 0.25 mg L^{-1} at the surface and from 0.11 mg L^{-1} to 0.26 mg L^{-1} at 15 m. At the second station, $\text{NO}_3\text{-N}$ concentrations ranged from 0.076 mg L^{-1} to 0.26 mg L^{-1} at the surface and from 0.053 mg L^{-1} to 0.25 mg L^{-1} at 15 m. At the third station, $\text{NO}_3\text{-N}$ concentrations ranged from 0.074 mg L^{-1} to 0.25 mg L^{-1} at the surface and from 0.096 mg L^{-1} to 0.3 mg L^{-1} at 15 m (Figure 8).

At the first station, $\text{NH}_4\text{-N}$ concentrations were different between depths and they ranged from 0.0052 mg L^{-1} to 0.013 mg L^{-1} at the surface and from 0.0073 mg L^{-1} to 0.017 mg L^{-1} at 15 m. At the second station, $\text{NH}_4\text{-N}$

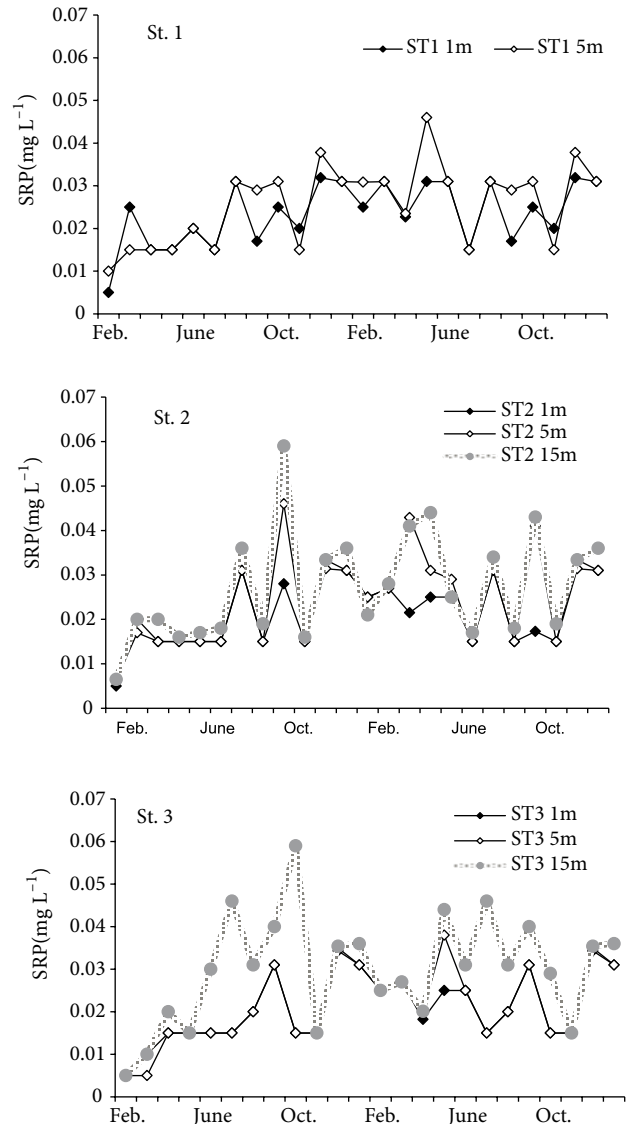


Figure 7. The seasonal variations in soluble reactive phosphorus (SRP, mg L^{-1}) concentrations of the Çaygören Reservoir (January 2007–December 2008).

concentrations ranged from 0.005 mg L^{-1} to 0.017 mg L^{-1} at the surface and from 0.006 mg L^{-1} to 0.017 mg L^{-1} at 15 m. At the third station, $\text{NH}_4\text{-N}$ concentrations ranged from 0.0049 mg L^{-1} to 0.013 mg L^{-1} at the surface and from 0.0064 mg L^{-1} to 0.21 mg L^{-1} at 15 m (Figure 9).

A total of 192 taxa in nine major taxonomic categories were identified. The nine taxa that made up more than 5% of the total phytoplankton biomass were the members of five functional groups (Table 1). Species of Bacillariophyta, Chlorophyta, Streptophyta, Cryptophyta, and Cyanobacteria dominated the phytoplankton at least once in the seasonal cycle. C, K, S1, T, and X2 were the identified functional groups. There were no distinct

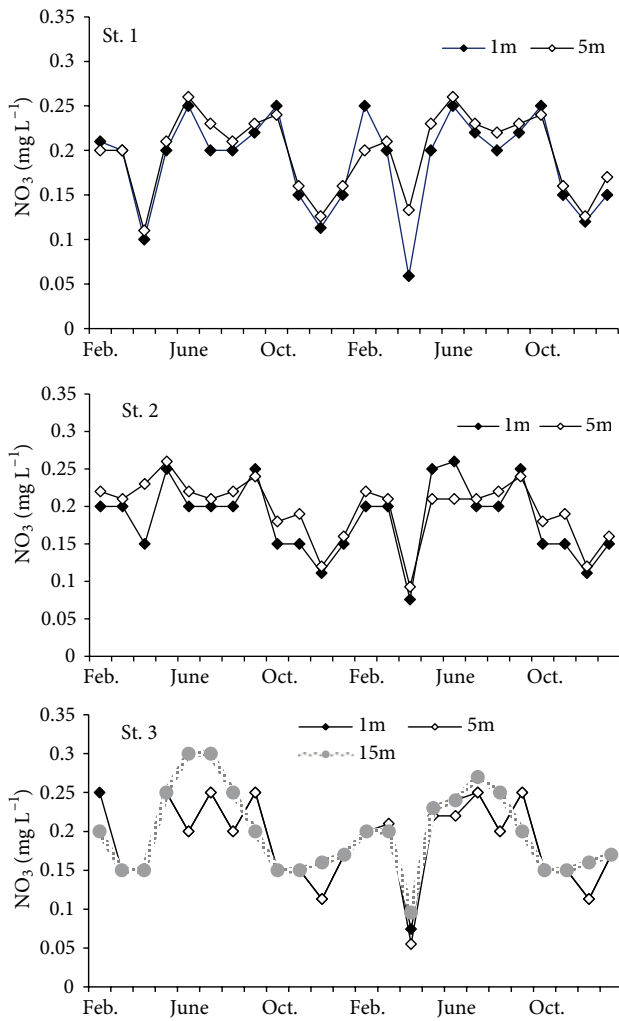


Figure 8. The seasonal variations in nitrate-nitrogen ($\text{NO}_3\text{-N}$, mg L^{-1}) concentrations of the Çaygören Reservoir (January 2007–December 2008).

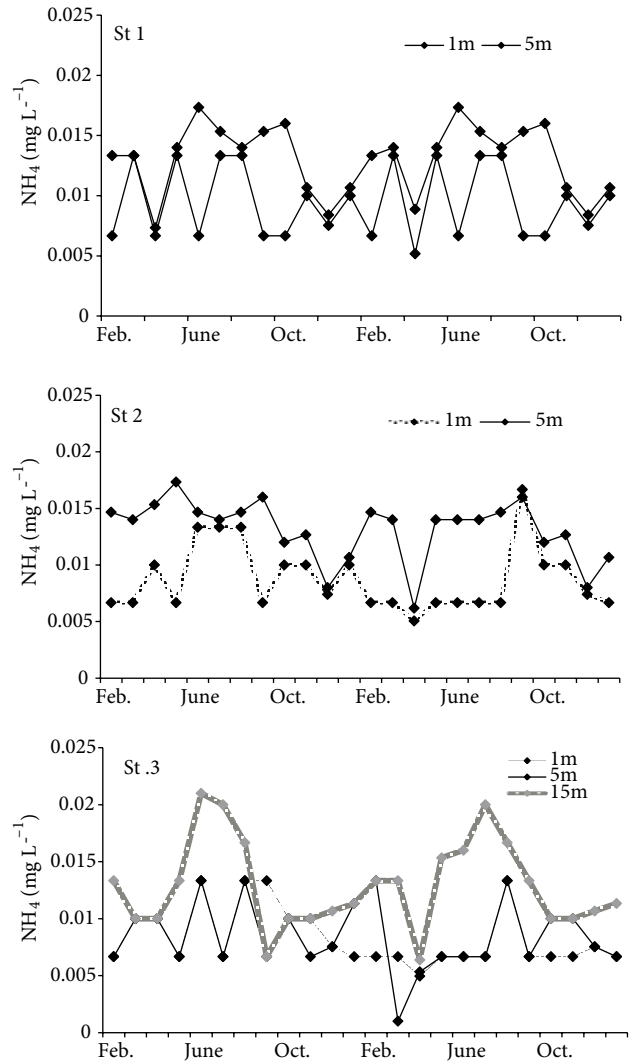


Figure 9. The seasonal variations in ammonium-nitrogen (NH_4 , mg L^{-1}) concentrations of the Çaygören Reservoir (January 2007–December 2008).

Table 1. Main phytoplankton taxa with the corresponding functional groups and the respective F factors in the Çaygören Reservoir between 2007 and 2009.

Functional group	Taxon	F Factor	Description of the functional group
C	<i>Cyclotella meneghiniana</i> , <i>Stephanodiscus neoastraea</i>	4	Mesotrophic small and medium-sized lakes
T	<i>Gloeotila subconstricta</i> , <i>Mougeotia</i> sp.	3	Mixed environments including clear epilimnia of deep lakes
X2	<i>Cryptomonas pyrenoidifera</i> , <i>Plagioselmis nannoplanctica</i>	4	Shallow, meso-eutrophic environments
K	<i>Aphanocapsa holsatica</i> , <i>Aphanothece clathrata</i>	2	Shallow, nutrient-rich water columns
S1	<i>Planktothrix</i> sp.	0	Turbid mixed environments

vertical distribution patterns of the functional groups, except for the **S1** group (*Planktothrix* sp.), which usually preferred deeper depths.

During the winter, the functional group **X2** (*Cryptomonas pyrenoidifera* Geitlerand, Cryptophyta and *Plagioselmis nannoplanctica* (H.Skuja) G.Novarino, I.A.N.Lucas and S.Morrall, Cryptophyta; 12% of biomass) dominated the phytoplankton. In the spring, the group **X2** was replaced by the group **C** (*Cyclotella meneghiniana* Kützing, Bacillariophyta and *Stephanodiscus neoastraea* Hakansson and Hickel, Bacillariophyta; 27% of biomass). In the summer, group **C** was replaced by the groups **S1** (*Planktothrix* sp., Cyanobacteria; 33% of biomass) and **K** (*Aphanocapsa holsatica* (Lemmermann) Cronberg, Cyanobacteria; 12.5% of biomass and *Aphanothece clathrata* West and G.S.West, Cyanobacteria; 30% of biomass). In the fall, the groups **S1** and **K** were replaced by the group **T** (*Gloeotila subconstricta* (G.S.West) Printz, Chlorophyta and *Mougeotia* sp., Streptophyta; 35.6% of biomass).

At the first station, the highest $Q_{(t)}$ index value (5) was calculated in May 2008 and the lowest values (0) were calculated from February to July 2007 and from August to December 2008. At the second station, the highest index value (3.5) was calculated in April 2008 and the lowest values (0) were calculated in February and June 2008. At the third station, the highest index value (5) was calculated in March 2008 and the lowest values (0) were calculated in November and December 2007. The $Q_{(t)}$ index values showed that different ecological states (from hypertrophic to oligotrophic) occurred in the Çaygören Reservoir depending on the time of the year and the station (Figure 10).

In the Çaygören Reservoir, the first and second axes of CCA explained 91.2% of the total variance in the functional groups-environment relationships (eigenvalues, 0.95 and 0.87). The third and fourth axes together explained 9.8%

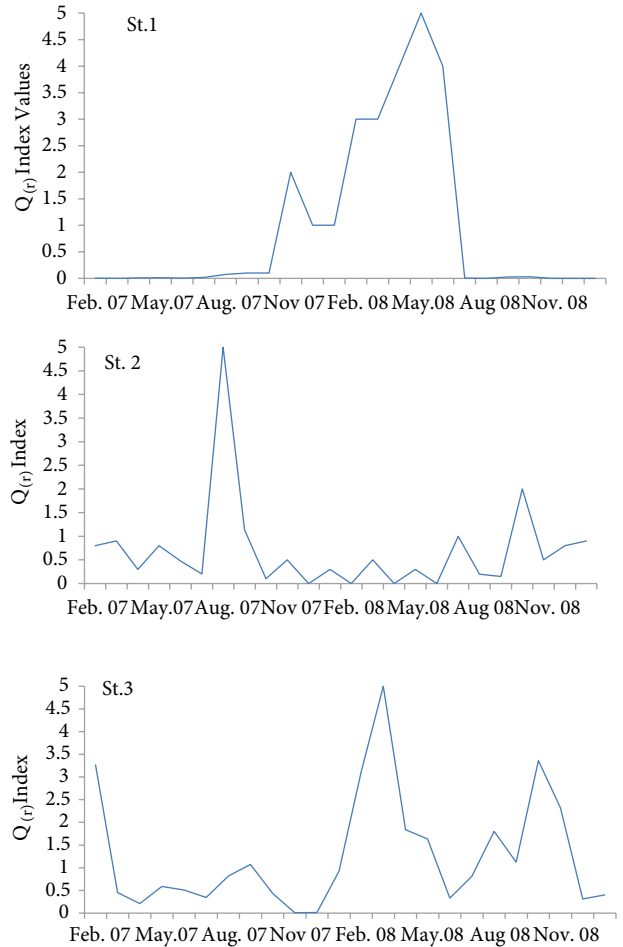


Figure 10. The seasonal differences of the $Q_{(t)}$ index for Çaygören Reservoir (January 2007–December 2008).

of the total variance (eigenvalues, 0.054 and 0.011). Table 2 shows the results of the Monte Carlo permutations for the significance of the individual environmental variables in order of the variance they explain.

Table 2. The results of Monte Carlo tests for the significance of the individual environmental variables (in order of the variance strength they explain).

Variable	Variable number	Variance explained	P	F
Discharge	9	0.32	0.001	3.83
Temperature	8	0.24	0.026	3.50
Secchi disk	7	0.21	0.012	3.42
NH ₄ -N	5	0.19	0.045	2.10
SRP	1	0.18	0.047	1.89
NO ₃ -N	4	0.09	0.37	0.71
ORP	3	0.05	0.46	0.66
pH	2	0.04	0.56	0.43

According to the Monte Carlo permutation results, water discharge, water temperature, $\text{NH}_4\text{-N}$, SRP, and water transparency had significant effects on the functional groups ($P < 0.05$). The first CCA axis was positively related to water temperature, SRP, ORP, $\text{NH}_4\text{-N}$, and pH and negatively related to the water discharge and water transparency (Secchi disk). The second axis was positively related to the SRP and ORP and it was negatively related to $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ (Figure 11).

The distribution of **S1** (*Planktothrix* sp., Cyanobacteria) and **K** (*A. holsatica*, Cyanobacteria and *A. clathrata*, Cyanobacteria) functional groups along the positive side of the first axis reflected their preference for high temperature and low water discharge. The functional group **T** (*G. subconstricta*, Chlorophyta and *Mougeotia* sp., Streptophyta) was also located on the positive side of the first axis and was less related to ammonium and water temperature. The functional group **X2** (*C. pyrenoidifera* and *P. nannoplanctica*, Cryptophyta) was located on the positive side of the second axis and it was related to water transparency, SRP, and ORP. The functional group **C** (*C. meneghiniana* and *S. neoastraea*, Bacillariophyta) was located on the negative side of the first axis and it was associated with high water discharge and low water temperature (Figure 11).

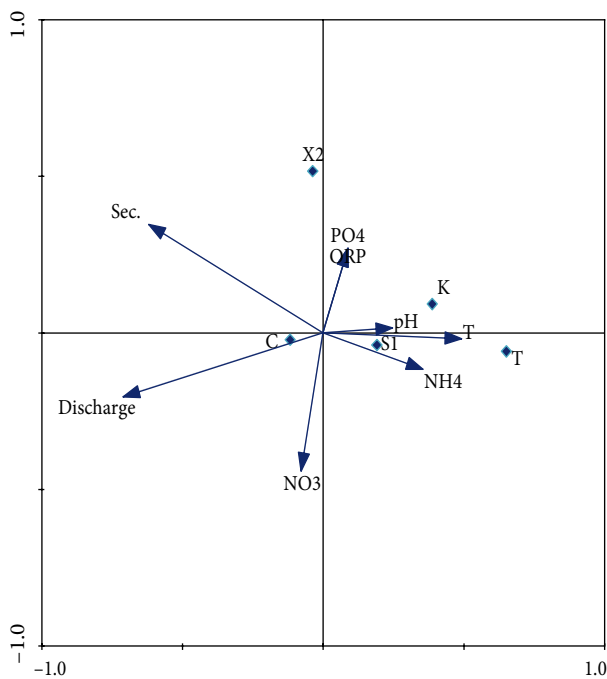


Figure 11. The canonical correspondence analysis (CCA) diagram showing the relationships of phytoplankton functional groups and environmental variables in the Çaygören Reservoir. Functional groups: **C** (*Cyclotella meneghiniana*, *Stephanodiscus neoastraea*), **T** (*Gloeotila subconstricta*, *Mougeotia* sp.), **X2** (*Cryptomonas pyrenoidifera*, *Plagioselmis nannoplanctica*), **K** (*Aphanocapsa holsatica*, *Aphanothece clathrata*), and **S1** (*Planktothrix* sp.).

4. Discussion

Water temperature, pH, and ammonium were related to the same axis of the CCA. This shows that all these parameters depend directly or indirectly on the same thing: hydrology and the consequences of thermal stratification. The relationships between water temperature and ammonium can be explained by the diffusion of $\text{NO}_3\text{-N}$ from the water column at low pH and high temperatures.

Planktothrix sp. was the most abundant cyanobacterium that dominated phytoplankton during the summer in the Çaygören Reservoir. In the CCA diagram, the functional group **S1** (*Planktothrix* sp.) occurred near the water temperature and opposite to the water discharge and water transparency vectors. The members of this functional group prefer turbid, fairly stagnant, or slow flowing environments and high water temperatures (Cuyppers et al., 2011; Wyngaert et al., 2011; Akçaalan et al., 2014)

Although *M. aeruginosa* reached a certain level of abundance (about 2% of the total biomass) in summer during the study, it is interesting to note that it never dominated the community. Ha et al. (1999) stated that the elevated water temperature ($>30\text{ }^\circ\text{C}$) along with low discharge and high irradiance were major factors contributing to the *Microcystis* sp. bloom in the Nakdong River–reservoir system in South Korea. Furthermore, *Microcystis* sp. occurrence depends on N:P ratio and lake hydrology (Xie et al., 2003; Abonyi et al., 2014). In the Çaygören Reservoir, during the dominance of *Planktothrix* sp., although the level of water discharge was low, zeu/zmix ratios were about 30%, meaning that there was probably not sufficient light for the dominance of *M. aeruginosa*, but plenty for the dominance of *Planktothrix* sp. (Reynolds, 2006).

Da Silva et al. (2005) found similar results in a Brazilian subtropical reservoir. They reported that *Planktothrix* sp. occurred near the bottom of the reservoir where there was low light and low disturbance, showing the high adaptability of this species to light decrease and water disturbance. The common occurrence of *Planktothrix* sp. (**S1**), shows a high adaptability to light and water discharge decrease, allowing adequate mixing conditions for *Planktothrix* sp. blooms (Reynolds et al., 2002).

The functional group **K** (*A. holsatica* and *A. clathrata*) was also dominant in the Çaygören Reservoir during the summer. The members of this group include non-N fixing small colonial cyanobacteria. Blomqvist et al. (1994) reported that non-N-fixing Cyanobacteria might dominate phytoplankton under low nitrogen conditions when some ammonium is still present in the water. The members of this group are favored by the conditions where these taxa might profit from the nutrient reserve of gelatinous structures, as well as by bouncing against sedimentation via good floating regulation (Reynolds et al.,

2002). There is a positive relationship between this group and water temperature. Besides water temperature, high ammonium concentrations may have also triggered the high abundance of this group in the Çaygören Reservoir (Komarek and Anagnostidis, 1998).

The functional group C (*C. meneghiniana* and *S. neoastreae*) dominated phytoplankton in the spring. In the CCA diagram, this group occurred near water discharge and opposite to the water temperature vectors. The members of this group are *r*-strategists, fast-growing species selected by high phosphorus concentrations in lakes (Anneville et al., 2002). This is a general spring phytoplankton compositional pattern in both lake and river systems, inside the reservoir; it depends on the flow conditions. In the case of stagnation, the general dominance of centric diatoms varies around reservoirs at a longitudinal scale, and real lentic composition occurs (Abonyi et al., 2012).

The functional group X2 (*C. pyrenoidifera* and *P. nannoplanctica*, Cryptophyta) dominated phytoplankton during the winter season. In the CCA diagram, this group occurred near the SRP vector. Various factors may regulate Cryptophyta seasonality in lakes, but it seems that the key factor in the success of this group is their low light requirement and possibly low zooplankton feeding pressure during higher flow. Hence, low transparency during the winter in the Çaygören Reservoir might have been responsible for the success of Cryptophyta species in this reservoir. Tian et al. (2014) studied phytoplankton of the Jiangdong Reservoir in China, similar to rivers with short hydraulic residence time, and observed that Cryptophyta species were present year-round and the dominated during winter months.

There were no significant differences in physical and chemical variables between the 2 years of the study (ANOVA, $P > 0.05$). Both years were extremely dry and the dry conditions caused extreme low water discharge in summer and fall. This in turn resulted in stagnancy in the reservoir. Since the reservoir is eutrophic on average, the

phytoplankton composition could not be explained only by the trophic state or by only the physical constraints. The patterns of the phytoplankton community were probably more related to the physical constraints, specifically during the summer.

The application of the river assemblage index, $Q_{(r)}$, in the Çaygören Reservoir indicated that the ecological status of the reservoir changed from hypertrophic to oligotrophic with time and space. Good ecological status (oligotrophy) during the spring was due to the contribution of the X2 group. The poor ecological (hypertrophy) status was due to the contribution of the groups S1 and K during the summer. The high abundance of Cyanobacteria (groups S1 and K) during the warm season was highly related to warmer water temperature, low water discharge, and stagnant conditions in the Çaygören Reservoir (Tsujimura and Okubo, 2003).

In conclusion, this study revealed that physical processes like stratification alter reservoir systems between river and lake conditions, where slow flow continuous mixing, low water discharge, and low light conditions may lead to dominance of Cyanobacteria during the warm season, and worsen the ecological status like found in the case of the Çaygören Reservoir. In contrast, in the spring and winter wet seasons, when Cyanobacteria dominance is prohibited, centric diatoms and cryptophytes dominance may occur, improving significantly the ecological status of reservoirs. The $Q_{(r)}$ composition metric was found to be effective in following the ecological status of the Çaygören Reservoir, proving a promising tool for water quality monitoring programs not only in lakes and rivers, but for reservoir systems of the temperate region as well.

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