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Ph.D in Department of Biology

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**UNIVERSITY OF GAZIANTEP
GRADUATE SCHOOL OF
NATURAL & APPLIED SCIENCES**

**DETERMINATION OF ECOLOGICAL STATUS OF
FRESHWATERS AND ECOLOGICAL PREFERENCES OF
ALGAE IN THE WESTERN MEDITERRANEAN BASIN
(TURKEY) USING MULTIVARIATE APPROACHES**

**Ph.D. THESIS
IN
DEPARTMENT OF BIOLOGY**

**BY
ASSANE ANABI TOUDJANI
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of Algae in the Western Mediterranean Basin (Turkey) Using Multivariate
Approaches**

Ph.D. Thesis

in

Department of Biology

University of Gaziantep

Supervisor

Prof. Dr. Abuzer ÇELEKLİ

by

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May 2018



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UNIVERSITY OF GAZİANTEP
GRADUATE SCHOOL OF NATURAL & APPLIED SCIENCES
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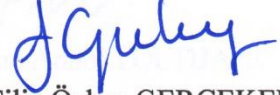
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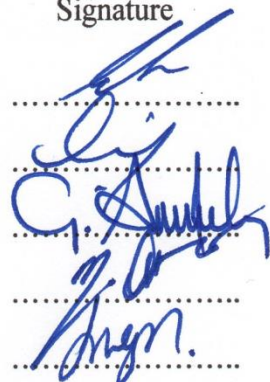
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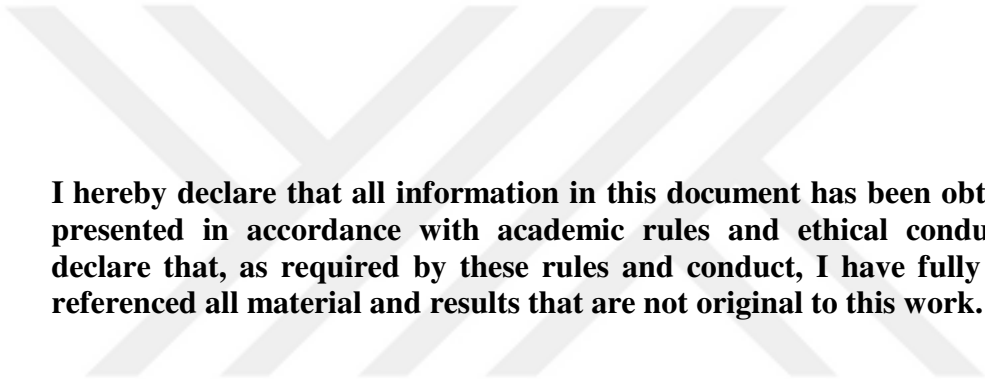
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Assane Anabi TOUDJANI

ABSTRACT

DETERMINATION OF ECOLOGICAL STATUS OF FRESHWATERS AND ECOLOGICAL PREFERENCES OF ALGAE IN THE WESTERN MEDITERRANEAN BASIN (TURKEY) USING MULTIVARIATE APPROACHES

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Several methods have been recently developed for water bioassessment. However, combining these methods to provide a realistic estimation of water quality is rare. The aim of this work was to assess freshwater quality using algae and multivariate approaches. Data were collected from 34 stations in the western Mediterranean basin of Turkey. A total of 206 phytoplankton and 102 diatom taxa were recorded in lentic and lotic systems respectively. Canonical correspondence analysis indicated that environmental variables influenced the distribution of algal taxa. In related to water quality, lakes and reservoirs showed moderate, good and high status according to Phytoplankton Trophic Index (PTI), Mediterranean Phytoplankton Trophic Index (Med-PTI) and the assemblage index (Q). Running waters showed poor to high ecological status based on Trophic Index Turkey (TIT). With regard to the trophic state, the lentic systems varied from oligotrophic to hypertrophic. It appears from this study that combining these metrics could be a useful approach for freshwater bioassessment. However, the use of these indices could lead to wrong interpretation of water quality because of ecoregional changes in algal composition, and there is a need to develop a national phytoplankton based metrics. Besides, TIT should be revised to include all diatom communities from various water types to extend its applicability to a number of aquatic systems.

Keywords: Lake; Running water, Biological indices, Bioassessment; Water quality.

ÖZET

BATI AKDENİZ HAVZASI (TÜRKİYE) ALGLERİN EKOLOJİK İSTEKLERİNİN VE SUCUL ORTAMLARIN EKOLOJİK DURUMLARININ ÇOK YÖNLÜ YAKLAŞIMLAR KULLANARAK BELİRLENMESİ.

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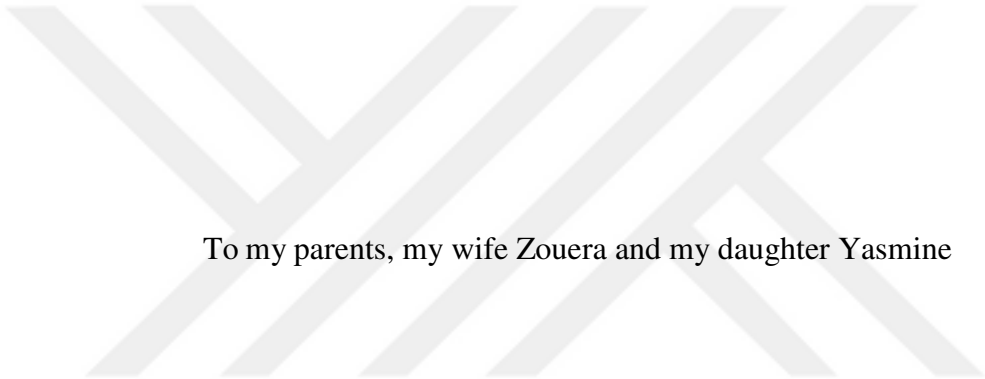
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Suların biyolojik olarak izlenmesi için son zamanlarda bazı yöntemler geliştirilmiştir. Ancak bu yöntemleri birlikte kullanarak su kalitesi hakkında değerlendirmede bulunmak hatalı sonuçlar verebilir. Bu tez algler ve çok değişkenli yaklaşımlar kullanılarak tatlı su kalitesini değerlendirmeyi amaçlamıştır. Örnekler, Türkiye'nin Batı Akdeniz Havzası'ndaki 34 tatlısu istasyonundan toplanmıştır. Lentik bölgelerden toplam 206 fitoplankton taksonu, lotik ekosistemlerden de toplam 102 diyatome taksonu tespit edilmiştir. CCA ile çevresel değişkenlerin alg türlerinin dağılımını etkilediği belirlenmiştir. Göller ve rezervuar sularının ekolojik kalite durumu ise phytoplankton trophic index, Mediterranean phytoplankton trophic index ve fitoplankton Q indeksine göre orta ve iyi ekolojik durumda olmuştur. Akarsu ekosistemlerindeki suların ekolojik kalitesi ise trofik indeks Türkiye (TIT) indeksine göre kötü ekolojik durumdan yüksek ekolojik duruma doğru arttığı belirlenmiştir. Sucul ekosistemlerin trofik durumuna bakıldığında lentik sistemin oligotrofikten hipertrofikliğe doğru değişim göstermiştir. Bu çalışmada bu indekslerin birleştirilmesi tatlı suların biyolojik izlenmesi için yararlı olabileceği görülmüştür. Bununla birlikte bu indekslerin doğrudan kullanılması alg kompozisyonundaki değişimler nedeniyle su kalitesinin hatalı yorumlanmasına neden olabilir. Bu yüzden ulusal fitoplankton bazlı bir ölçüm geliştirmeye ihtiyaç vardır. Ayrıca TIT, tüm diyatome topluluklarını kapsayacak şekilde uygulanabilirliği gözden geçirilmelidir.

Anahtar Kelimeler: Göl; Akar su; Biyolojik indeksler; Biyodeğerlendirme; Su kalitesi.



To my parents, my wife Zouera and my daughter Yasmine

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LIST OF SYMBOLS/ABBREVIATIONS

TP	Total phosphorus
P-PO ₄	Orthophosphate
TN	Total nitrogen
N-NO ₂	Nitrite nitrogen
N-NO ₃	Nitrate nitrogen
N-NH ₄	Ammonium nitrogen
TKN	Total Kjeldahl nitrogen
SD	Secchi depth
TOC	Total organic carbon
DO	Dissolved oxygen
BOD ₅	Biological oxygen demand
PTI	Phytoplankton trophic index
Med-PTI	Mediterranean phytoplankton trophic index
EQR	Ecological quality ratio
TIT	Trophic index Turkey
TI	Trophic index
EPI-D	Eutrophication and/or Pollution Index - Diatom
TSI	Trophic State Index

CHAPTER I

GENERAL INTRODUCTION

1.1 Importance of Water

Water is the most important and abundant compound of the world and makes up approximately 75% of its surface. However, only a very small quantity of the water is freshwater and appropriate for human use (Smol, 2008). Surface waters are important in drinking water supply, agriculture, irrigation; fish farming, recreation, climate and flood regulation and other activities to which the water should be monitored and assessed (Johnson et al., 1997; Holt, 2000; Van Leeuwen, 2000; Sargaonkar and Deshpande, 2003; Petraccia et al., 2006; Boyacioglu, 2007; UN, 2007; Goncharuk, 2008; Mustapha, 2008; Wallis et al., 2011; Cieszynska, et al., 2012; Patil et al., 2012; Ndungu, 2014). Water makes up about 70% of human bodies. More than half of the world's species of plants and animals live in water, and even our terrestrial-derived food is totally dependent on and often largely composed of water (Smol, 2008).

1.2 Water Pollution and Eutrophication

The quality of water is crucial before use, and is defined as its physical, chemical, biological parameters and organoleptic characteristics (Johnson et al., 1997; Sargaonkar and Deshpande, 2003; UN, 2007; Boyacioglu, 2007). The availability of clean water and the good ecological status of surface waters have been endangered by increasing loads of nutrients and chemicals (Malve, 2007). Polluted water may impact irrigated plants resulting in accumulation of salts in the root zone, by leading to loss of permeability of the soil because of excess sodium or calcium percolating, or by containing pathogens or pollutants which are directly toxic to crops or to those consuming them. Contaminants in irrigation water may accumulate in the soil and, after a period of years, render the soil unfit for agriculture. Even when the presence of pesticides or pathogenic organisms in irrigation water does not directly impact plant development, it may eventually impact the quality of the agricultural product

for sale or farmers consumption (Enderlein et al., 1997). Freshwater quality deterioration usually comes from eutrophication, organic pollution, heavy metal contamination, excessive nutrient inputs, acidification or obnoxious fishing practices (Mustapha, 2008). Since the industrial revolution and the expanding of the human population, the production of a huge quantity of harmful products are increasing year by year and causes water deterioration. Anthropogenic activities, increased with industrial revolution have impacted water quality in a significant number of surface watercourses worldwide. Some pollutant products are used largely in agriculture (fertilizers, pesticides, fungicides etc.), in crafts (dye), factories, automotive etc. After use, these products are discharged directly in the water bodies or in the soil contaminating thus the underground and surface water through the intermediary of runoff water. Discharging of effluents and pollutants into watercourses may undesirable consequences on the ecosystems and their biodiversity (Gonzalo and Fernández, 2012). The availability of dyes in surface water ecosystems reduces light penetration, decreases photosynthetic activity and could cause aesthetic issues (Saratale et al., 2011). Also, many dyes or their metabolites in water bodies even at low concentration can cause a variety of diseases and disorders in living organisms such as allergy, skin irritation, and cancer (Çelekli et al., 2009; Salleh et al., 2011; Dotto et al., 2013). Anthropogenic activities have caused a number of negative impacts to the biodiversity, on the management of water resources, and the assessment of the biological integrity in the great proportion of inland aquatic ecosystems (Cao et al., 2007; Allan and Castillo, 2007; Delgado and Pardo, 2014). Environmental deterioration is any disturbance on the environment that is seen to be nuisible for biodiversity, in this way, anthropogenic activities have deteriorated freshwater systems, generating consciousness and the increase in the development of bioassessing methods to evaluate the water quality of aquatic ecosystems (Kelly and Whitton, 1995; Hering et al., 2006, Delgado and Pardo, 2014). Discharging of wastewaters, nutrient-rich runoff, and land use irrigation had a great impact on watercourses over the last century (Billen et al., 2001; Wunsan et al., 2002; Ducharne et al., 2007, Delgado and Pardo, 2014). The increasing presence of nutrients such as nitrogen and phosphorous in fresh watercourses, generally related to eutrophication, is impacting the productivity and functioning of primary producers in aquatic ecosystems (Leira et al., 2009; Delgado and Pardo, 2014). Blooms of algae due to eutrophication (excessive enrichment with nutrients) increasing turbidity following

the loss of living organisms due to oxygen decreasing or acidification are among effects of anthropogenic activities on freshwater aquatic ecosystems. The input of effluents from industries into receiving water may cause negative impacts on the ecosystems. As these waters become more eutrophic, the higher cost of water treatment has encouraged watershed management in North America, Taiwan, South Africa and elsewhere (Horne and Goldman, 1994). Due to this reason, developed countries have continually been assessing and categorising their watercourses, considering characteristic structure of their own rivers and have used this type of indices for revealing the current situation of their water quality level (Solak and Ács, 2011). The importance of managing freshwater ecosystems is becoming more evident every year (Horne and Goldman, 1994).

1.3 European Union Water Framework Directive

The increase of anthropogenic influences in watersheds during the last decades has lead limnologists to develop biomonitoring methods that can quickly evaluate the water quality of aquatic systems (Kelly and Whitton, 1998). In this sense, a great effort has been put forth worldwide to monitor fresh waters using not only physical and chemical variables, but also biological quality elements. Biomonitoring gives a direct measure of ecological integrity by using the species responses to the changes in environmental variables (Karr, 1991; Angermeier and Karr, 1994; Directive, 2000; EC, 2009). In accordance with this idea, the European Union water framework directive (WFD) required member states to protect and assess all their water resources, with the aim of achieving good ecological potential and good surface water quality status by 2015 (EC, 2009). The environmental objectives of the WFD were to i) prevent deterioration, ii) protect, and iii) enhance and restore water resources. Biological quality elements, including phytoplankton, phytobenthos, macrophytes, benthic macroinvertebrates, and fish are employed by the EU Water Framework Directive (WFD) 2000/60/EC as bioindicators for the monitoring of fresh waters (Directive, 2000; EC, 2009) (Figure 1.1.). According to the WFD, physicochemical parameters, as well as the hydromorphological data were used to support the biological quality elements.

In order to calibrate the use of the biomonitoring systems, the results of the biological metrics of each methods shall be transformed as an ecological quality

ratios (EQR) for the purposes of categorisation of ecological status. These ratios shall represent the relationship between the values of the biological parameters obtained for a given surface watercourse and the values for these variables in the reference conditions applicable to that watersource. The ratio shall be calculated as a numerical value between 1 (bad ecological status by) and 0 (high ecological status). Five ecological status classes have been determined and established (Figure 1.2).

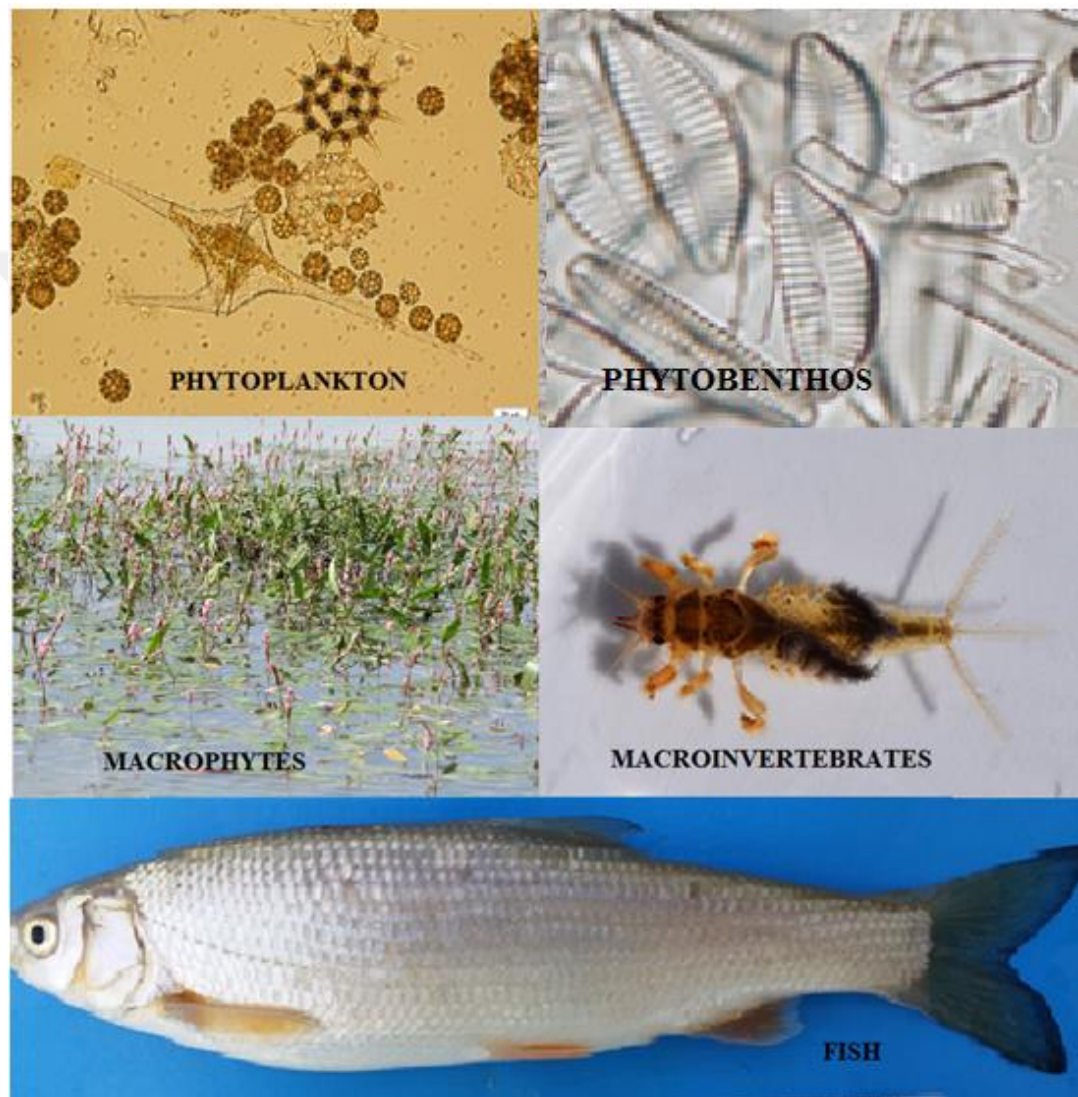


Figure 1.1 Biological quality elements suggested by the WFD

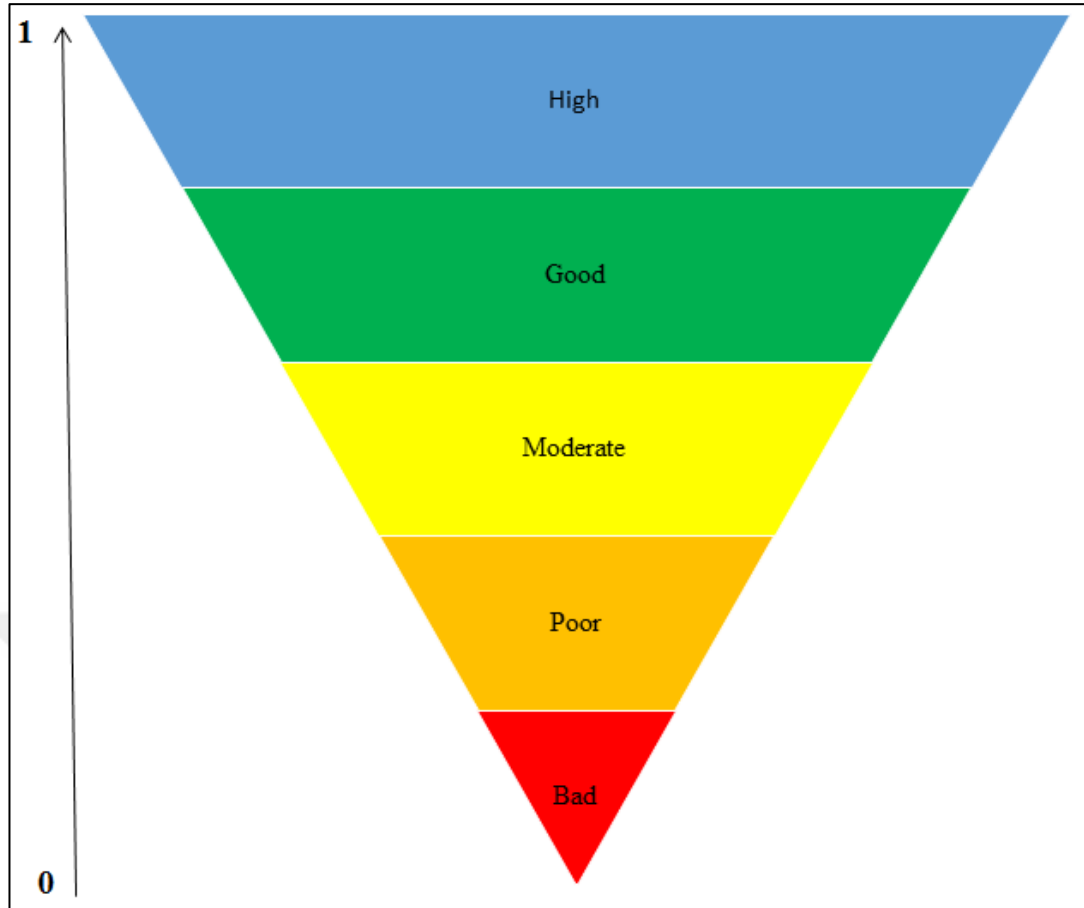


Figure 1.2 Ecological quality ratio (0-1) and ecological status classes

Normative classifications of ecological status classifications for rivers, lakes, transitional waters and coastal waters were then established by the water framework directive. The normative definitions of ecological status classifications (High, good moderate, poor and bad status) are given in Table 1.1.

As a candidate country to the European Union, Turkey adapted the water frame directive for the assessment of its surface water bodies. For this reason, many studies based on the WFD have been carried out in the different regions of Turkey and start to become more and more important. According to Erdoğan (2016) the first project was conducted in Büyük Menderes River basin. Recently, two new projects conducted from 2014 to 2017 by the Ministry of forestry and water affairs for the establishment of an ecological assessment system for water quality and reference monitoring network in Turkey.

Table 1.1 Normative definitions of ecological status classifications. General definition for rivers, lakes, transitional waters and coastal waters. From Annex V of the WFD (Directive, 2000)

Status	General condition
High status	There are no, or only very minor, anthropogenic alterations to the values of the physicochemical and hydromorphological quality elements for the surface water body type from those normally associated with that type under undisturbed conditions. The values of the biological quality elements for the surface water body reflect those normally associated with that type under undisturbed conditions, and show no, or only very minor, evidence of distortion. These are the type specific conditions and communities.
Good status	The values of the biological quality elements for the surface water body type show low levels of distortion resulting from human activity, but deviate only slightly from those normally associated with the surface water body type under undisturbed conditions.
Moderate status	The values of the biological quality elements for the surface water body type deviate moderately from those normally associated with the surface water body type under undisturbed conditions. The values show moderate signs of distortion resulting from human activity and are significantly more disturbed than under conditions of good status.
Poor and bad	Waters achieving a status below moderate shall be classified as poor or bad.

1.4 Phytoplankton and Freshwater Quality

Phytoplankton or microalgae are free living photosynthetic microorganisms in aquatic systems and are primary producer of the limnetic zone. Phytoplankton communities dominate the surface water systems that cover 70% of the world's surface area (Reynolds, 2006). As prokaryotic or eukaryotic, unicellular, colonial or filamentous (Figure 1.3) free-living photosynthetic microorganisms in aquatic systems, phytoplankton play a key role in the organisation and functioning of aquatic ecosystems and are most relevant bioindicators for the monitoring of nutrient enrichment.

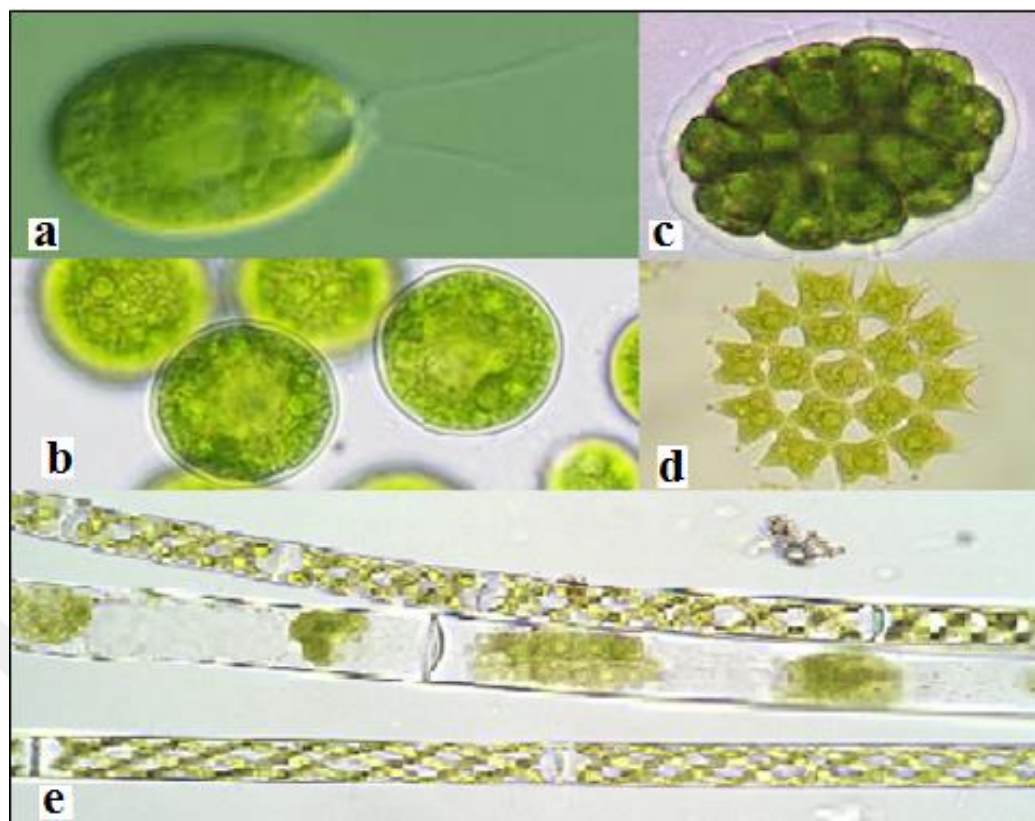


Figure 1.3 Phytoplankton living forms a-b) unicellular, c-d) colonial, and e) filamentous, modified from Anonymous (2015) and SBSAC (2018)

The use of phytoplankton, one of the five biological quality elements as ecological indicator recommended by the WFD to evaluate surface waters is getting more and more important in lakes and reservoirs. Phytoplankton would be used as a good indicator for assessing water quality, due to its sensitivity and rapid responses to variations in its habitat (Padisák et al., 2006). The indicative value of phytoplankton biomass has long been recognized (Naumann, 1919; Nygaard, 1949; Poikane et al., 2011), and a number of metrics and methods have been developed for using phytoplankton in assessments of eutrophication (Carlson, 1977; OECD, 1982; Poikane et al., 2011) and its biovolume (Marchetto et al., 2009). Phytoplankton composition, abundance and its effect on transparency conditions are generally used in the assessment of the ecological status of surface water (Directive, 2000; EC, 2009; Mischke et al., 2012; Phillips et al., 2013). Firstly developed for lakes, phytoplankton functional system is being applied to other categories of reservoirs because of the similarity of the two systems (Reynolds, 1999, 2000). In this approach, an metric using phytoplankton functional groups has been recently proposed by Padisák et al. (2006). Several phytoplankton indices including the algae group index (AGI), Catálan Index, Barbe Index, Brettum index, phytoplankton

trophic index (PTI), Mediterranean phytoplankton trophic index (Med-PTI) using the phytoplankton biovolume and Q assemblage index have been developed for the assessment of ecological status of lakes and reservoirs.

Currently, more than 4000 taxa of marine phytoplankton belonging to Cyanobacteria, Chlorophyta, Cryptophyta, Eustigmatophyta, Chrysophyta, Bacillariophyta, Dinophyta, Raphidophyta, Haptophyta, Euglenophyta, Prasinophyta, Glaucophyta, Anoxyphotobacteria phyla have been identified (Sournia et al. 1991; Tett and Barton, 1995; Reynolds, 2006). In comparison, this number is probably lower in fresh waters (Reynolds, 2006). Phytoplankton are a key bioindicator of the health and functioning of freshwaters in relation to high nutrient level, and for measuring the success of restoration measures following reductions in nutrient inputs (Carvalho et al., 2013). The composition of a phytoplankton assemblage is known to depend not only water quality, physical factors and lake basin size, but also on biological factors such as specific growth and loss rates among the algae, parasitism, predation, and competition (Lepistö et al., 2004, 2006). Phytoplankton community is responsible for the total primary production in surface waters (Reynolds, 1984; Vadeboncoeur et al., 2011; Cellamare et al., 2012), and because of their short life times, respond quickly to variations in their habitat (Schaumburg et al. 2004; Poikane et al., 2011; Cellamare et al., 2012). Phytoplankton communities have developed morphological and physiological adaptive strategies for surviving in different habitats (Margalef, 1978; Reynolds, 1998; 2006; Reynolds et al., 2002; Padisák et al., 2003; Becker et al., 2010; Fetahi et al., 2014), and reflect autecological aspects of preference and tolerance (Fetahi et al. 2014). Thirty nine (39) phytoplankton functional groups which may dominate or co-dominate potentially, and alternately, in a given environment have been identified (Reynolds et al., 2002; Padisak et al., 2009). The phytoplankton functional groups approach applied to aquatic systems has provided key information concerning the distribution of phytoplankton species in the limnetic zone in various regions of the world (Fabbro and Duivenvoorden, 2000; Kruk et al., 2002; Huszar et al., 2003; Leitão et al., 2003; Lopes et al., 2005; Romo and Villena, 2005; Sarmiento et al., 2006; Becker et al., 2009, 2010).

Algae blooms happen when a species of algae reproduces rapidly and reaches high concentrations. Too much phosphorus and nitrogen (found in fertilizers, animal waste, and sewage) lead to perfect conditions for algae blooms. There are several

socioeconomic issues associated with eutrophication due to increases in phytoplankton biomass, especially with high frequency and persistence of toxic cyanobacteria blooms (Carvalho et al., 2013). These include deterioration of drinking water quality, treatment for water supply, recreational activities, and conservation (Carvalho et al., 2013). Phytoplankton blooms principally blue green algae (Cyanobacterial) (Figure 1.4) present main challenges for the management of surface watercourses. Cyanobacterial blooms have adverse effects on freshwater quality and human health, with serious economic and ecological consequences (Hallegraeff, 1993; Mur et al., 1999; Carey et al., 2012). The increased occurrence and persistence of blooms have been associated to human activities, especially nutrient enrichment watercourses regulation and climate change associated with rising levels of atmospheric CO₂ (Anderson et al., 2002; Mooij et al., 2005; Paul, 2008; Beardall et al., 2009; Paerl and Huisman, 2009; Carey et al., 2012; Carey et al., 2012). The surface scum produced by *Microcystis*, the most common cyanobacterium in freshwater blooms is well known to affect water quality conditions by reducing water valuable services (Paerl et al., 2001; Lehman et al., 2005, 2008, 2010; Acuna et al., 2012a, b; Lehman et al., 2013).

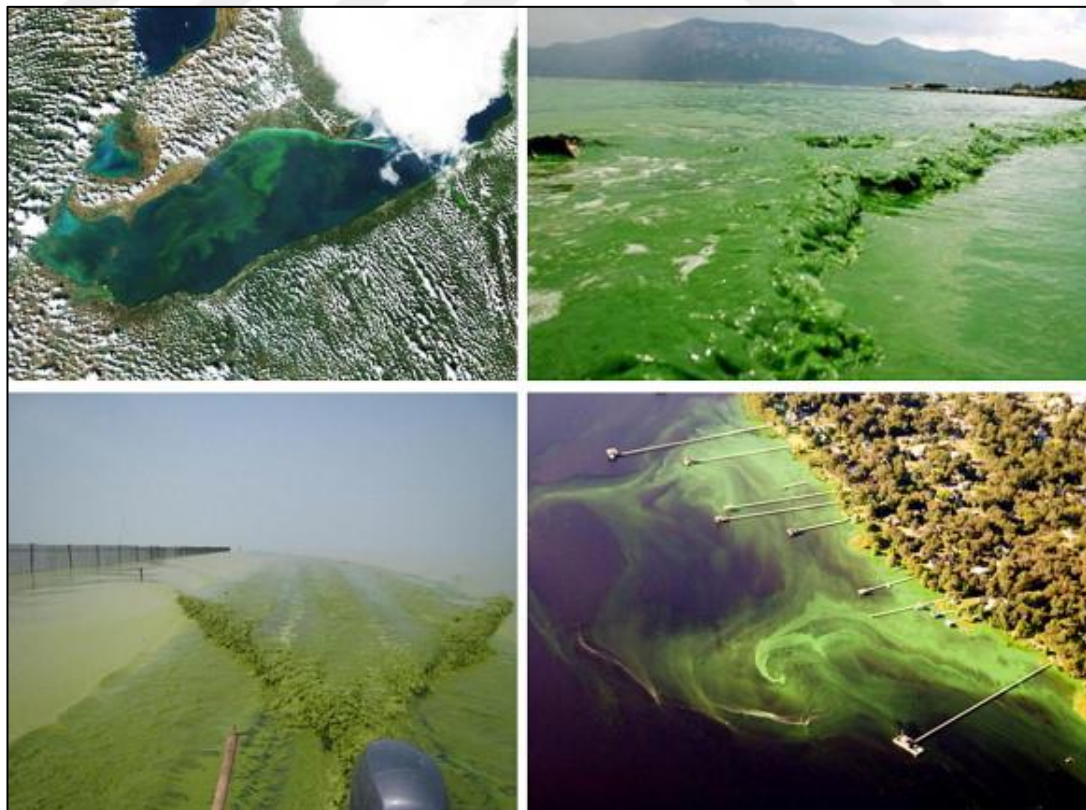


Figure 1.4 Blue green algae blooms in freshwater systems (Paerl and Paul, 2012)

1.5 Phytobenthos in Running Water Monitoring

Benthic diatom have been indicated as good indicators of variations in nutrient levels in running waters (Hering et al., 2006; Johnson et al., 2006a, b, Johnson et al., 2014; Gottschalk, 2014). Phytobenthos communities (especially diatoms) have been widely used in running waters biomonitoring than lake and reservoirs bioassessment compared to phytoplankton and macrophytes (Vadeboncoeur and Steinman, 2002; Ács et al., 2005; Cellamare et al., 2012). Diatoms in running watercourses are a major group of phytobenthos and biodiversity of the ecosystems. They are becoming an useful tool for fresh water quality biomonitoring worldwide in recent decades (e.g., Rott et al., 1999, 2003; Prygiel et al., 1999; Lobo et al., 2004; Potapova et al., 2004; Wang et al., 2005; Herring et al., 2006; Taylor et al., 2007; Porter et al., 2008; Kelly et al., 2009; Rimet, 2012; Rimet et al., 2016). For many reasons, diatoms are largely applied in fresh waters bioassessment especially running water bodies. Several values of their metrics have been calculated for the prediction of water quality (Poikane et al., 2016). Diatom-based metrics usually indicate a highly significant and well response to nutrient gradients (Rott et al., 2003; Kelly et al., 2014, 2016) with higher correlation coefficients than those of other biological elements (Birk et al., 2012). Among biological quality elements, diatom assemblages often considered as important ecological indicators of water bodies due to their rapid and sensitive response to environmental changes (Kelly, 1998; Stoermer and Smol, 1999; Potapova et al., 2004; Rimet et al., 2007; Bona et al., 2007; Delgado and Pardo, 2014). For the assessment of water quality, benthic diatoms have been used since the 1900s (Kolkwitz and Marsson, 1908). In Europe, benthic diatom metrics such as IPS (Cemagref 1982), TI (Rott et al., 1999), EPI-D (Dell'Uomo, 2004), TDI (Kelly et al., 2008) and recently TIT (Çelekli et al., 2016, 2018) developed for running waters are successfully applied for water quality assessment worldwide.

Diatoms are an extremely diverse group of algae with about 100,000 species (Round, 1991; Mann and Droop, 1996). They are an important group of phytobenthos (often 90–95%) and are good bioindicators of water quality in running waters. The advantage in benthic diatoms is that they can be found in every surface water, at any time (Solak and Ács, 2011). They have been widely used in biomonitoring because of their short life cycles and their quickly response to habitat changes (Hering et al., 2006; Bona et al., 2007; Delgado and Pardo, 2014).

In general, diatoms are unicellular algae, but they can form colonies life-forms. Diatom cells are usually referred to as a frustule. The wall is composed of two sections (valves) with different sizes. One valve with its girdle fits over the other girdle with its valve. The outer one is often referred to as the epitheca and the inner one as the hypotheca (Patrick and Reimer, 1966). The frustule is siliceous and brittle and ordinarily etched with lines, rows of dots or puncta (Prescott, 1961). The longitudinal line which appears in the middle of one or both valves is known as raphe and enables diatoms to move (Figure 1.5). There are two groups of Diatoms, the centrics and pennates. The common type of reproduction of diatoms is accomplished by vegetative reproduction in with mitosis. The pigments which are directly or indirectly associated with photosynthesis in diatoms are chlorophyll a and c; carotene α , β , ϵ ; diadinoxanthin; diatoxanthin; fucoxanthin, neofucoxanthin A, and neofucoxanthin B (Patrick and Reimer, 1966). Fucoxanthin and carotene β are responsible of the characteristic brown color of diatoms. An example of living diatom and a cleaned frustule after treatment are given in Figure 1.6.

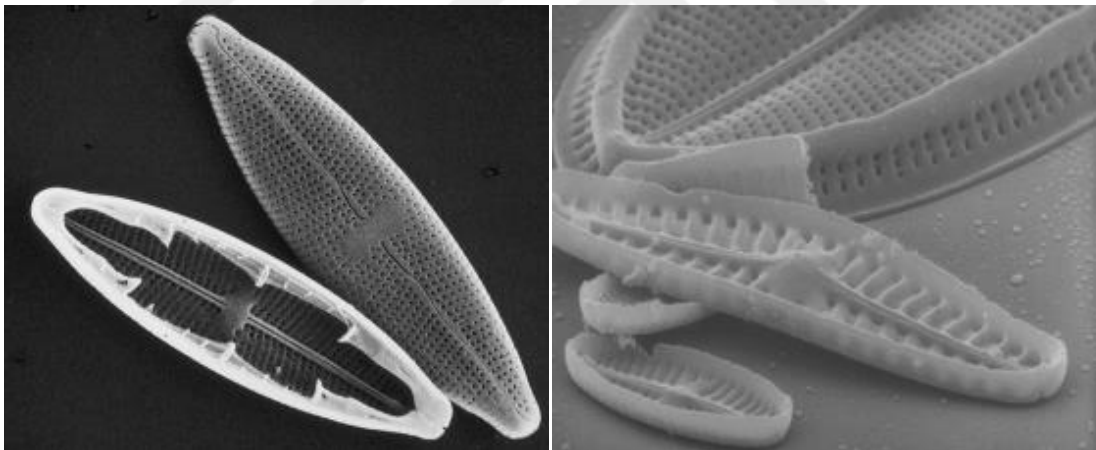


Figure 1.5 Valves of Diatom under electronic microscope (Hürlimann and Niederhauser, 2007)



Figure 1.6 *Cymbella tumida* (left): living material; (right): cleaned frustule after nitric acid treatment (Rimet, 2012)

Diatoms are present in various type of habitats, ranging from fresh and salt water. They live in intertidal zones and on damp soil, rocks, or plants where the spray from waves or falling water reaches them (Patrick and Reimer, 1966). They may present several life-forms during their development. They can be unicellular (Figure 1.7) and free moving at one stage, but attached to a peduncle and immobilized at another stage (Rimet, 2012). Diatoms can be found also in the types of colonies (e.g. chain, ribbon, zig zag Rosette, arbuscular, mucous).

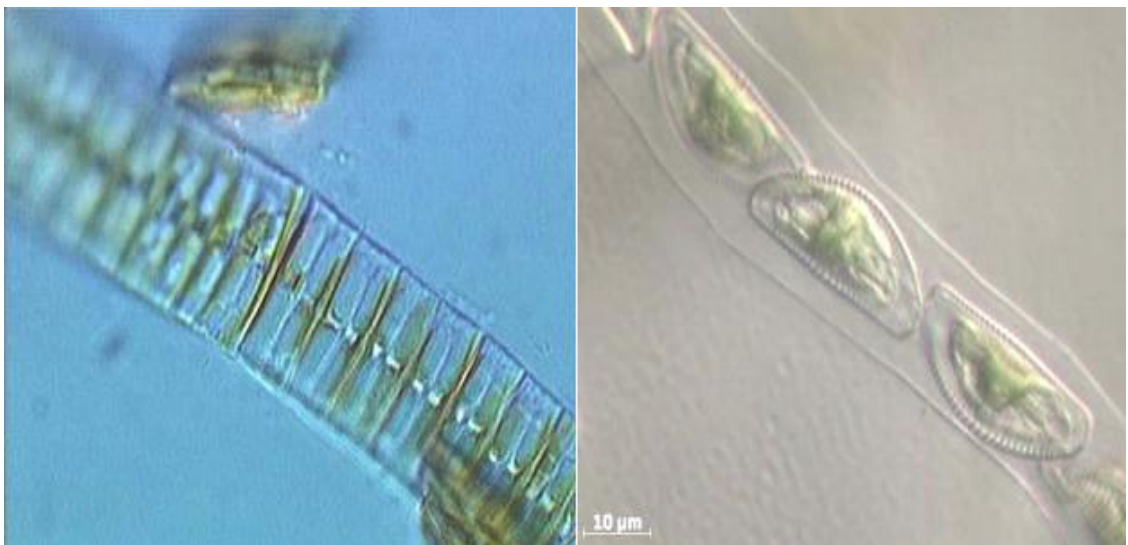


Figure 1.7 Diatom life-forms (Rimet, 2012). (a) ribbon colony with *Fragilaria capucina* var. *vaucheriae*, (b) mucous tubules with *Encyonema minutum*

1.6 Objectives of the Present Study

The monitoring of water quality has a long history especially based on physicochemical data. However, biological assessment of freshwater ecosystems becomes more and more important since the implementation of European Union Water Framework Directive. The principal objective of this study was to evaluate the water quality of fresh waters bodies (lakes, reservoirs, streams and creeks) in the Western Mediterranean basin of Turkey based on phytoplankton and phytobenthos two of the mandatory five biological indicators indicated by the European Union Water Framework Directive using multivariate approaches. The objectives of the study were to:

1. Estimate the phytoplankton assemblages and the structuring factors in Mediterranean lakes and reservoirs.
2. Determine the environmental divers of diatom communities' composition in running water bodies.
3. Evaluate the ecological status of lakes and reservoirs based on phytoplankton metrics
4. Evaluate running water quality based on new diatom metric developed in Turkey
5. Evaluate the trophic states of the lakes and reservoirs

The following hypotheses associated to aforementioned objectives were postulated and tested in the present study:

- First, there is no difference phytoplankton composition between lakes and reservoirs, and no difference in diatom composition between running waters.
- Second, environmental variables, especially pollution parameters (nutrients, organic matter) are the most important structuring factors of phytoplankton and diatom assemblages in lakes and reservoirs and running water bodies
- Third, there is no difference between the ecological status of the lakes and reservoirs on one hand and between running water bodies on the other hand
- Fourth, there is no difference between the trophic states of the lakes and reservoirs.

CHAPTER II

LITERATURE REVIEW

Surface freshwaters (lakes, reservoirs, and rivers) are among the most extensively degraded ecosystems on earth, and most of them have been degraded in multiple ways, (Stephen et al., 2011). Assessing the water quality in freshwater ecosystems should include biological as well as physicochemical parameters. Monitoring of water quality especially freshwater has a long history and based on biological quality elements as well as physicochemical parameters and nutrients (phosphate, nitrate...). In last decades, a significant effort has been done worldwide to monitor water quality using not only physicochemical parameters, but also bioindicators which are extremely sensible to environment changes. In this case, the direct study of the effects of pollution on biota is of great interest (Gonzalo and Fernández, 2012). Due to demographic change and socioeconomic activities, water quality in surface waters is degrading in many parts of the world (Schindler, 2006; Smol, 2008), disturbing thereby many aquatic ecosystem functions and services. In order to understand these disturbances and to provide solutions, the EU water framework directive (WFD) required all member states to protect and assess all fresh water bodies with the aim of achieving good water status at the latest 15 years from the date of entry into force of this Directive, (2000/60/EC). Biological quality elements, including phytoplankton, phytobenthos, macrophytes, benthic invertebrates and fish are recommended by the EU Water Framework Directive (WFD) 2000/60/EC to be used as bioindicators for the monitoring of surface waters (EC, 2009; Directive, 2000).

2.1 Physicochemical Parameters and Ecological Assessment

Assessment of water quality by using physical and chemical variables of ecosystems approach is the empiric way in water ecological status evaluation. Prior studies have been conducted on water quality using physicochemical parameters. (e.g. Renn, 1968; Pratti et al., 1971; Gaudet, 1979; Olajire and Imeokparia, 2001; Tepe et al., 2005; Janse et al., 2008; Mustapha, 2008; Manjare et al., 2010; Patil et al., 2012; Ndungu, 2014). The changes in physicochemical parameters of water provide

valuable information on the quality of the water, the sources of the modifications and their effects on the functions, structures and biodiversity of the watercourses (Mustapha, 2008). The trophic state of the water bodies can be determined by using physicochemical parameters especially phosphate, temperature, nitrogen. However, there is a quite few number of criteria for the assessment of water quality using physicochemical parameters (e.g. Carlson 1977; OECD, 1982).

Rokade and Ganeshwade (2005) studied the effect of pollution on water quality of Salim Ali Lake at Aurangabad, Uttar Pradesh (India) and indicated high fluctuations in the physicochemical parameters demonstrating the intensity of pollution.

Mustapha, 2008, investigated the variations in physicochemical factors for determining the water quality of Oyun Reservoir, Offa, Kwara State, Nigeria. This study concluded that the reservoir had an excellent water quality and high ecological status.

Joshi et al. (2009) found that physicochemical parameters such as pH, electrical conductivity, total dissolved solids, total suspended solids, turbidity, and sodium values were higher than those of the prescribed limit in some water samples of River Ganga.

Cieszynska et al. (2012) applied physicochemical data to assess water quality in the Gdansk Municipality (South Baltic coast) and indicated that water quality of the watercourses in the region depends on their locations.

Sorlini et al. (2013) analyzed the physicochemical parameters in the Logone Valley (Chad-Cameroon) in order to identify the contamination problems. They detected high physicochemical levels both in aquifers and surface waters.

Using physicochemical properties of water to evaluate water quality provides a good impression of the state, function, structure, and sustainability of such watercourses (Mustapha, 2008). However, this method with instantaneous measurements mainly gives restricting knowledge of water conditions. The chemical data at each sampling period indicate the current state of the water quality and ignore the temporal and spatial variations of water quality in aquatic ecosystems (Rocha, 1992). On the other hand, the use of biological monitoring gives a direct information on the ecological

integrity based on the response of organisms to the variations in environmental variables (Karr, 1991; John, 2000; EC, 2009).

In this study, the physicochemical parameters were used as supplementary elements as suggested by the FWD.

2.2 Bioassessment of Water Quality

Biological assessment is an important tool to evaluate biological integrity of watercourses, and consist of the monitoring and other direct investigations of aquatic living organisms such as phytoplankton, photobenthos, macrophytes, macroinvertebrates, fishes, etc. Cranstont et al. (1996) argued that, as a result, changes in the health of a catchment will be reflected in the aquatic living organisms. The biological communities that are under to pollutants react as integrators of the various present and past environmental stresses.

2.2.1 Ecological Assessment Based on Phytoplankton

Early attempts were made in the 20 and 21 centuries to assess water quality focus on study of phytoplankton in monitoring lakes (e.g. Naselli Flores and Barone, 1998, Willén, 2001; Reynolds et al., 2002; Padisák et al., 2003; Lepistö et al., 2006; Naselli Flores and Barone, 2007; Salmaso and Padisák, 2007; Moreno-Ostos et al., 2008; Poikane et al., 2009; Tolotti et al., 2010; Pasztaleniec and Poniewozik, 2010; Poikane et al., 2011; Brucet et al., 2013; Carvalho et al., 2013; Crossetti et al., 2013; Demir et al., 2014; Poikane et al., 2016). A few studies were made in rivers and reservoirs assessment based on phytoplankton (e.g., Negro et al., 2000; Figueredo and Giani, 2001; Moreno-Ostos et al., 2008; Hoyer et al., 2009; Chellappa et al., 2009; Becker et al., 2010; Soylu and Gönülol, 2010; Katsiapi et al., 2011; Abonyi et al., 2012; Molina-Navarro et al., 2012, 2014; Çelekli and Öztürk, 2014). Phytoplankton are one of the five biological quality elements recommended as an ecological indicator by the European Union Water Framework Directive (Directive, 2000) to assess surface waters is getting more and more important in lakes and reservoirs. The normative definitions of ecological status classifications (high, good and moderate) for lakes based on phytoplankton are given in Table 2.1.

Reynolds et al. (2002) established a criteria of categorisation of phytoplankton species to be sensitive to habitat variations (essentially to eutrophication but also to

shorter seasonal fluctuations in stratification and in the accessibility of adequate nutrient supplies) based on functional groups (Table 2.2). They sought to show some of the uses of the functional-group model.

Table 2.1 Normative definitions of ecological status classifications. Definitions for high, good, moderate, poor and bad ecological status in lakes based on phytoplankton. From Annex V of the WFD (Directive, 2000)

Status	Phytoplankton
High status	The taxonomic composition and abundance of phytoplankton correspond totally or nearly totally to undisturbed conditions. The average phytoplankton biomass is consistent with the type-specific physicochemical conditions and is not such as to significantly alter the type specific transparency conditions. Planktonic blooms occur at a frequency and intensity which is consistent with the type specific physicochemical conditions.
Good status	There are slight changes in the composition and abundance of planktonic taxa compared to the typespecific communities. Such changes do not indicate any accelerated growth of algae resulting in undesirable disturbance to the balance of organisms present in the water body or to the physico-chemical quality of the water or sediment. A slight increase in the frequency and intensity of the type specific planktonic blooms may occur.
Moderate status	The composition and abundance of planktonic taxa differ moderately from the type specific communities. Biomass is moderately disturbed and may be such as to produce a significant undesirable disturbance in the condition of other biological quality elements and the physico-chemical quality of the water or sediment. A moderate increase in the frequency and intensity of planktonic blooms may occur. Persistent blooms may occur during summer months.
Poor status	Waters showing evidence of major alterations to the values of the biological quality elements for the surface water body type and in which the relevant biological communities deviate substantially from those normally associated with the surface water body type under undisturbed conditions, shall be classified as poor.
Bad status	Waters showing evidence of severe alterations to the values of the biological quality elements for the surface water body type and in which large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions are absent, shall be classified as bad.

Tablo 2.2 Functional groups of phytoplankton (Reynolds et al., 2002)

Codon	Habitat
A	Clear, often well-mixed, base poor, lakes
B	Vertically mixed, mesotrophic small-medium lakes
C	Mixed, eutrophic small- medium lakes
D	Shallow, enriched turbid waters, including rivers
N	mesotrophic epilimnia
P	eutrophic epilimnia
T	deep, well-mixed epilimnia
S1	turbid mixed layers
S2	shallow, turbid mixed layers
SN	warm mixed layers
Z	clear, mixed layers
X3	shallow, clear, mixed layers
X2	shallow, clear mixed layers in meso-eutrophic lakes
X1	shallow mixed layers in enriched conditions
Y	usually, small, enriched lakes
E	usually small, oligotrophic, base poor lakes or heterotrophic ponds
F	Clear epilimnia
G	Short, nutrient- rich water columns
J	shallow, enriched lakes ponds and rivers
K	short, nutrient-rich columns
H1	dinitrogen-fixing Nostocaleans
H2	dinitrogen-fixing Nostocaleans of larger mesotrophic lakes
U	summer epilimnia
L_O	summer epilimnia in mesotrophic lakes
L_M	summer epilimnia in eutrophic lakes
M	dielly mixed layers of small eutrophic, low latitude lakes
R	metalimnia of mesotrophic stratified lakes
V	metalimnia of eutrophic stratified lakes
W1	small organic ponds
W2	shallow mesotrophic lakes
Q	small humic lakes

In last decades, numerous phytoplankton metrics have been developed for the assessment of lakes and reservoirs:

Brettum (1989) in Norway, developed the Brettum index using the trophic preferences of species, modified by Dokulil and Teubner, 2006 in Austria.

Catalan et al. (2003) developed Catalán Index. The index is based on the percentage of biovolume of the different groups of phytoplankton.

Philippe et al. 2003 developed the ITP or Barbe Index in France. This index, using the proportion biovolume of given phytoplankton taxa, has been developed to infer the water quality of lakes from the composition of their phytoplankton assemblages.

Riedmüller et al. (2006) developed the Phytoplankton Trophic Lake Index (PTSI) based on type-specific indicator species, their trophic and weighting values in Germany.

Padisák et al. (2006) developed an assemblage index (Q) on basis of recent developments in phytoplankton ecology to evaluate the water quality of different lake types. The developed index ranged between 5 and 0 ($5 \geq Q \geq 0$), and can provide five (5) classes of qualification. They did not conclude for the superiority of the assemblage index as a monitoring tool of ecological status assessment within the WFD. However, they indicated that the proposed metric could has some remarkable characteristics that meet the idea of requirements of habitat based establishments. This index seems to be a useful tool for lakes assessment and has been tested in many counties.

Salmaso et al. (2006) used phytoplankton as an bioindicator of the water quality of the Deep Lakes South of the Alpsproposed and developed the Phytoplankton Trophic Index (PTI) in Italy. They concluded that the method applied to develop this index, and the subsequent result may be applied to study other lake types, providing a robust tool to evaluate the degree of specificity of the trophic indicator scores assigned to the single phytoplankton orders and species.

Marchetto et al. (2009) developed a phytoplankton based index called Mediterranean phytoplankton trophic index (Med-PTI) for estimating phytoplankton response to variations in nutrient levels in deep Mediterranean reservoirs. They indicated that the

developed metric can be considered a useful tool for assessing the response of the phytoplankton to variations in nutrient concentration in deep reservoirs in agreement with the WFD guidelines.

Phillips et al. (2013) developed a composition metric, the plankton trophic index (PTI) to evaluate the status of lakes. They suggested that the taxonomic composition of the phytoplankton can be quantified using trophic optima derived from a pan-regional dataset and provided that country-specific reference values are used to account for climatic and biogeographical differences and can be used to compare and assess the status of lakes across Europe.

Hutorowicz and Pasztaleniec (2014) developed Phytoplankton Metrics for Polish Lakes (PMPL) in Poland. The PMPL includes abundance parameters of phytoplankton: the metrics: “chlorophyll a”, “total biomass” while the taxonomic composition is partly evaluated by the metric “biomass of cyanobacteria”

de Hoyos et al. (2014) New Mediterranean Assessment System for Reservoir's Phytoplankton (NMA SRP) in Cyprus and Portugal based on Chlorophyll-a ($\mu\text{g/L}$), Biovolume (mm^3/L), IGA (Index Des Grups Algals) and Cyanobacterial biovolume (mm^3/L) to assess reservoirs.

Phillips et al. (2014) developed Danish lake phytoplankton Index (DLPI) based on both abundance and composition metrics.

Aforementioned methods are useful tools for assessing water quality. However, combining these methods to provide a realistic estimation of water quality is rare. In this work, we proposed to evaluate the water quality by combining three of these methods and multivariable approaches to reduce the influence of climate change and natural environmental variability on the credibility of phytoplankton applications.

Using phytoplankton for water quality assessment is a new topic in Turkey but start to become more and more important after 2000. During the last two decades, a number of studies using phytoplankton have been carried out in the different regions of Turkey.

Soylu and Gönülo (2010) studied the functional categorisation and composition of Phytoplankton in Liman Lake from a data collected between January 2002 and

December 2003 were they indicated an alternate between R-Strategists and C-Strategists groups in the succession of phytoplankton.

Demir et al. (2014) tested the Q assemblage index based on phytoplankton functional classification in Lake Mogan and provided a good water state estimation. They identified 76 phytoplankton species belonging to 12 functional groups. They argued that the Q index was useful to follow the principal seasonal variation of the physicochemical parameters and indicated a moderate ecological status for Lake Mogan.

Çelekli and Öztürk (2014) determined the water quality and ecological preferences of phytoplankton using a multivariate approach in Alleben Reservoir. They identified nine (9) functional groups (A, D, E, J, L_O, L_M, MP, P, and Y) as descriptors and a medium ecological status for the reservoir based on Q index and PTI, and argued that these indices should be suitable phytoplankton metrics to evaluate the water quality of the reservoir.

Çelekli et al. (2016) modified phytoplankton trophic index (PTI) for assessing many lentic ecosystems from 8 basins of Turkey (Lower Euphrates, Ceyhan, the West Mediterranean, the North Aegean, Sakarya, the West Black Sea, the East Black Sea, and Aras).

2.2.2 Ecological Assessment Based on Benthic Diatoms

Benthic diatoms are widely used as a representative for the entire phytobenthos community for many reasons in lakes and rivers environmental assessment. Diatoms in streams and rivers represent a major part of phytobenthos (often 90–95%) and aquatic biodiversity, and they have become a useful tool in water quality biomonitoring. Consisting of inert and easily preserved frustules, diatoms identification and analysis is less complicated than other phytobenthos algae. Besides, there is huge knowledge about diatom identification for a long time, resulting in a number of key books (e.g. Krammer and Lange-Bertalot, 1986-1991; Krammer 2000, 2002, 2003; Lange-Bertalot, 2001; Lange-Bertalot and Metzeltin, 1996) to facilitate the counting based on diatoms shape, constitution and characteristics. Similarity in water nutrient enrichment state deduction has been indicated using benthic diatoms as well as the entire phytobenthos communities

(Kelly et al., 2008). Phytobenthos are one of the five (5) bioindicators recommended as an ecological indicator by the European Union Water Framework Directive (Directive, 2000) to assess surface waters are widely used in evaluating running water quality. The normative definitions of ecological status classifications (high, good and moderate) for rivers based on phytobenthos are given in Table 2.3.

Table 2.3 Normative definitions of ecological status classifications. Definitions for high, good, moderate, poor and bad ecological status in rivers based on phytobenthos. From Annex V of the WFD (Directive, 2000)

Status	Phytobenthos
High status	The taxonomic composition corresponds totally or nearly totally to undisturbed conditions. There are no detectable changes in the average phytobenthic abundance.
Good status	There are slight changes in the composition and abundance phytobenthic taxa compared to the type-specific communities. Such changes do not indicate any accelerated growth of phytobenthos or higher forms of plant life resulting in undesirable disturbances to the balance of organisms present in the water body or to the physico-chemical quality of the water or sediment. The phytobenthic community is not adversely affected by bacterial tufts and coats present due to anthropogenic activity
Moderate status	The composition of phytobenthic taxa differs moderately from the type-specific community and is significantly more distorted than at good status. Moderate changes in the average phytobenthic abundance are evident. The phytobenthic community may be interfered with and, in some areas, displaced by bacterial tufts and coats present as a result of anthropogenic activities.
Poor status	Waters showing evidence of major alterations to the values of the biological quality elements for the surface water body type and in which the relevant biological communities deviate substantially from those normally associated with the surface water body type under undisturbed conditions, shall be classified as poor.
Bad status	Waters showing evidence of severe alterations to the values of the biological quality elements for the surface water body type and in which large portions of the relevant biological communities normally associated with the surface water body type under undisturbed conditions are absent, shall be classified as bad.

Several diatom based indices have been carried out in freshwater ecological status and states evaluation based on phytobenthos (e.g. Van Dam et al., 1994; Kelly et al., 1995; Rott et al., 1997; Kelly et al., 1998; Rott and Pipp, 1999; Finlay et al., 2002; Rott et al., 2003; Potapova et al., 2004; Ács et al., 2005; Hering et al., 2006; Bona et al., 2007; Della Bella et al., 2007; Solak and Ács, 2011; Cellamare et al., 2012; Beltrami et al., 2012; Schneider et al., 2013; Bennion et al., 2014, Delgado and Pardo, 2014, Wang et al., 2014). Due to this reason, several diatom indices were developed for the bioassessment of running water quality. Wang et al. (2014) listed the diatom indices widely used in river health assessment which are among the following metrics.

Percent Community Similarity of Diatoms (PSc) (Whittaker et al., 1958): This index is used to compare, monitor and examine sites, or average community of a category of monitor or reference sites with a test station. Percent community similarity scores varied from 0 (absence of similarity) to 100%.

Simple Autecological Index (SAI) (Lowe, 1974): This index is used to characterize different diatom species along an environmental (stressor) gradient and deduce environmental variations and impact on the phytobenthos community.

Descy Index (DI) (Descy, 1979): The first diatom index developed by Descy in 1979.

Pollution Tolerance Index for Diatoms (PTI) (Lange-Bertalot, 1979): This index is used to classify groups of diatoms based on their tolerance to increased pollutants, with species associated to a score from one (1) for most tolerant species to three (3) for relatively sensitive taxa.

Specific Pollution Sensitivity Index (SPI) (Coste, 1982): Bioassessment of water quality via calculating the proportion of specific pollution sensitive species.

Diatom Assemblage Index for Organic Pollution (DAI_{po}) (Watanabe et al., 1986): Calculates the relative abundance of eusaprobic and eurysaprobic species to assess the degree of polluted water.

Sládeček Index (SLA) (Sládeček, 2006): Bioassessment of water quality via calculating the proportion of saprobic diatom species.

Index of Leclercq and Maquet (ILM) (Leclercq and Maquet, 1987): This metric is used to indicate organic pollution and assess water quality. Derived from Sládeček's method, Leclercq and Maquet proposed other values for the “saprobic valencies” (s) and for the “indicator values” (v).

Generic Diatom Index (GDI) (Rumeau and Coste, 1988; Wu, 1999): This index is the ratio of abundance of *Achnanthes*, *Cocconeis* and *Cymbella*, to that of *Cyclotella*, *Melosira* and *Nitzschia* to measure diatom assemblage changes.

Percent Motile Diatoms (PMD) (Bahls, 1993): This is a siltation index, calculated as the relative percentage of *Navicula* + *Nitzschia* + *Surirella*. The three (3) genera are able to move towards the surface of water if they are covered by silt; their percentage in a given water body provides an idea about the level and frequency of siltation.

Trophic Diatom Index (TDI) (Kelly and Whitton, 1995, Kelly et al., 2008): This index was developed to evaluate the water quality of running waters (especially rivers) based on diatom composition.

Biological Diatom Index (BDI) ([Lenoir and Coste, 1996 ; Prygiel et al., 2002 ; Coste et al., 2009): This index was developed using a list of 209 key species indicating various pollution sensitivities. The pollution sensitivity, or “ecological profile”, is calculated using the species occurrence probability scores along a seven (7) quality classes scale.

Rott saprobic index (ROT) (Rott et al, 1997): This index is an improved saprobic index based on the Sládeček Index; some saprobic values and indicator values have been changed.

Percent live diatoms (PDI) (Hill, 1997): This index is used to show the state of the diatom community. According to this metric, low score could be the consequence of high sedimentation with high algal biomass on substrates.

Percent Aberrant Diatom (PAD) (Mc Farland et al., 1997): PAD indicates the proportion of diatom assemblages in a sampling station that shown anomalies in striae patterns or frustule shapes. The index has been positively correlated with heavy metal presence in running water.

Shannon Diversity (for diatoms) (Barbour et al., 1999): this system consists of a function related to the number of taxa in a given water sample as well as the repartition evenness of individuals of those taxa.

Percent Sensitive Diatoms (PSD) (Barbour et al., 1999): This index is the total of the relative abundances of all intolerant taxa. The metric is especially applicable in smaller-order streams where algae may be naturally low.

Trophien index (TI) (Rott et al., 1999); This index has been developed to estimate the trophic status of running water in Austria. $TI = (\text{Sum of (Indicator Taxa Abundance * Indicator trophic value * Indicator weighting score)}) / \text{Sum of (Indicator Taxa Abundance * Indicator weighting score)}$.

Diatom Model Affinity (DMA) (Passy and Bode, 2004): the index gives an evaluation of water quality by the determination of percent similarity to a model community that can be viewed as a reference standard. The high similarity to the model indicates assemblages that are minimally disturbed, while lower similarity suggests water quality problems.

Pampean Diatom Index (PDI&IDP) (Gómez and Licursi, 2001): IDP integrated both organic pollution and eutrophication, and has been applied for the biomonitoring of water quality in rivers and streams in the Pampean plain.

Eutrophication Pollution Index using diatoms (EPI-D) (Dell'Uomo, 1999, 2004): This diatom-based metric was calculated based on the sensitivity of diatom taxa to organic pollution, mineralization of water, and chloride.

Relative Abundance of Diatom Species (RAD) (Griffith et al, 2005): High taxa number is supposed to show high biotic integrity because many taxa are adapted to the situations present in their environments. Species number is predicted to be low with high pollution conditions because several species are sensitive.

Eastern Canadian Diatom Index (IDEC) (Lavoie et al, 2006; Grenier et al, 2010): This index is based on correspondence analysis (CA) to develop a chemistry-free index, where the position of the sites along the gradient of maximum variance is strictly determined by diatom assemblage structure and therefore is dependent on measured environmental variables.

Diatom Index for Australian Rivers (DIAR) (Chessman et al., 2007): This index is used as a broad-scale indicator of human influences on Australian rivers, especially the effects of agricultural and urban land use, and also for impact studies at local scales.

Trophic index Turkey (TIT) (Çelekli et al., 2016, 2018): Recently developed, the index was calculated based on species sensitivity to nutrient contents of the waters. The trophic weight and indicator values which served to calculate this index were calculated according to the repartition of diatom taxa in different TP classes.

As it can be clearly seen, except the newly developed trophic index Turkey (TIT), there is no diatom based metrics for freshwater quality assessment in Turkish. However, several previous studies (Gürbüz and Kıvrak, 2002; Akbulut et al., 2010; Tokatlı and Dayıoğlu, 2011; Solak, 2011; Solak and Acs, 2011; Solak et al., 2012) were carried out in Turkey using foreign diatom metrics which could lead to lack of important knowledge concerning the quality of watercourses. However, it has been known that the direct application of foreign diatom based system could be a source a wrong interpretation of water quality because of the difference in diatom assemblage between various regions (Pan et al., 1996). In this thesis, we assessed running waters in the western Mediterranean basin of Turkey by combining trophic index Turkey (TIT), a national newly developed diatom-based metric, another Mediterranean called eutrophication and pollution index based on diatom (EPI-D) and the trophic index (TI).

CHAPTER III

MATERIAL AND METHODS

3.1 Description of the Study Area

The studied water bodies are located in the West of the Mediterranean region (in Turkish: Akdeniz Bölgesi) of Turkey. The Mediterranean basin of Turkey is limited by the Aegean basin to the west, the Central Anatolia basin to the north, the Eastern Anatolia basin to the northeast, the Southeastern Anatolia basin to the east, Syria to the southeast, and the Mediterranean Sea to the south (Figure 3.1). This region is approximately 110,000 km² and makes up 14% of the land of Turkey (Gündogan et al., 2009).

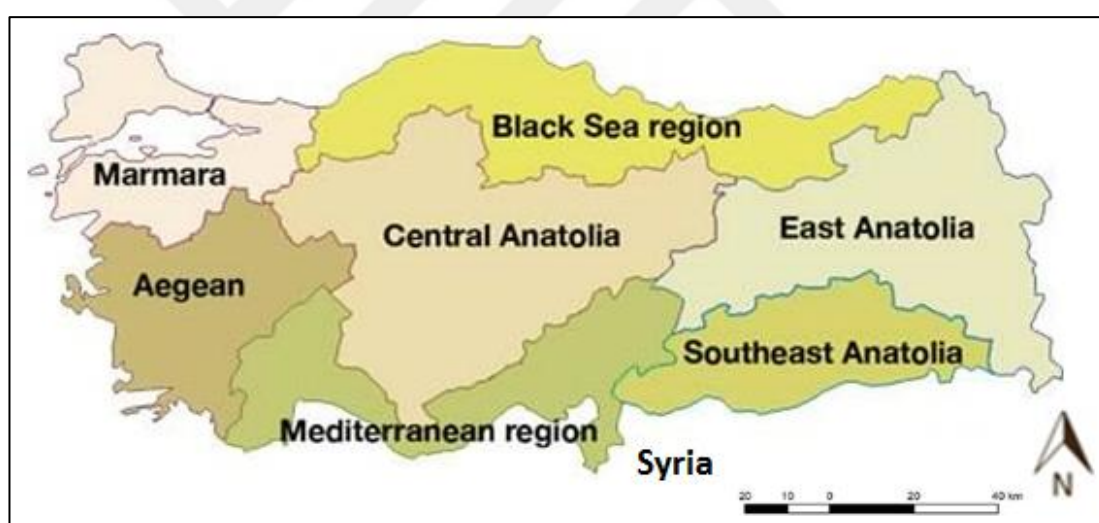


Figure 3.1 Location of the study area (Mediterranean basin) on the map of Turkey, modified from Nations Online Projects (2018)

The climate of the basin is different from that the climate of the Mediterranean because the basin carries a parade between the Aegean, Central Anatolia and Mediterranean Regions. In the settlements with the Aegean and Mediterranean borders, the Mediterranean climate is dominant. The characteristics of the Mediterranean and continental climates are evident in the Denizli province basin settlements as they feature a gateway between the Aegean, Central Anatolia and the

Mediterranean Regions. The Mediterranean-Aegean transition climate can be seen in the Burdur province. In the continental climate, winters are cold and snowy and summers are hot and dry; In the Mediterranean climate, winters are warm and rainy, summers are hot and cold. In the basin, a large part of the annual total precipitation is in fall and the winter months. Snow in fall and frosts are rarely seen on the coast. The Mediterranean Basin of Turkey predominant vegetation cover is depending on the Mediterranean climate characteristics. Also, the Mediterranean region of Turkey is a mountain chains region (e.g. Toros mountains, Amonos mountains, ect.). The coastal plains of the region are formed in lower river watersheds. West Mediterranean basin consisted of many aquatic ecosystems (running waters, reservoirs and lakes) which play an important role in the socioeconomic activities, especially agriculture, irrigation and fish farming in the basin. The important water bodies of the Western Mediterranean Basin are shown in Table 3.1 and Figure 3.2 (Anonymous, 2016).

Table 3.1 Main watercourses of the Western Mediterranean Basin

Running water	Reservoirs	Lakes
Tersakan Creek	Elmalı Çayboğazı Dam	Lake Gölhisar Gölü
Sarıçay	Finike Alakır Dam	Lake Koca Gölü
Namnam Creek	Eşen 1 HES	Lake Avlan
Kargıcak Deresi	Akköprü Dam ve HES	Lake Köycegiz
Dalaman River	Belkaya Dam	Lake Yazır
Seki Creek	Çavdır Dam	
Çayıçi Creek	Kozağacı Dam	
Kocadere/Kızılöz Creek	Yapraklı Dam	
Kocadere Upper part	Geyik Dam	
Çavdır Creek	Akgedik Dam	
Boğluca Creek	Mumcular Dam	
Akçay Creek	Marmaris Dam	
Alakır Creek	Elmalı Çayboğazı	
Koca Creek/Kanlı Creek	Çavdır Dam	
Eşen Creek	Toptaş Dam	
	Yapraklı Dam	
	Osmankalfalar Dam	
	Geyik Dam	

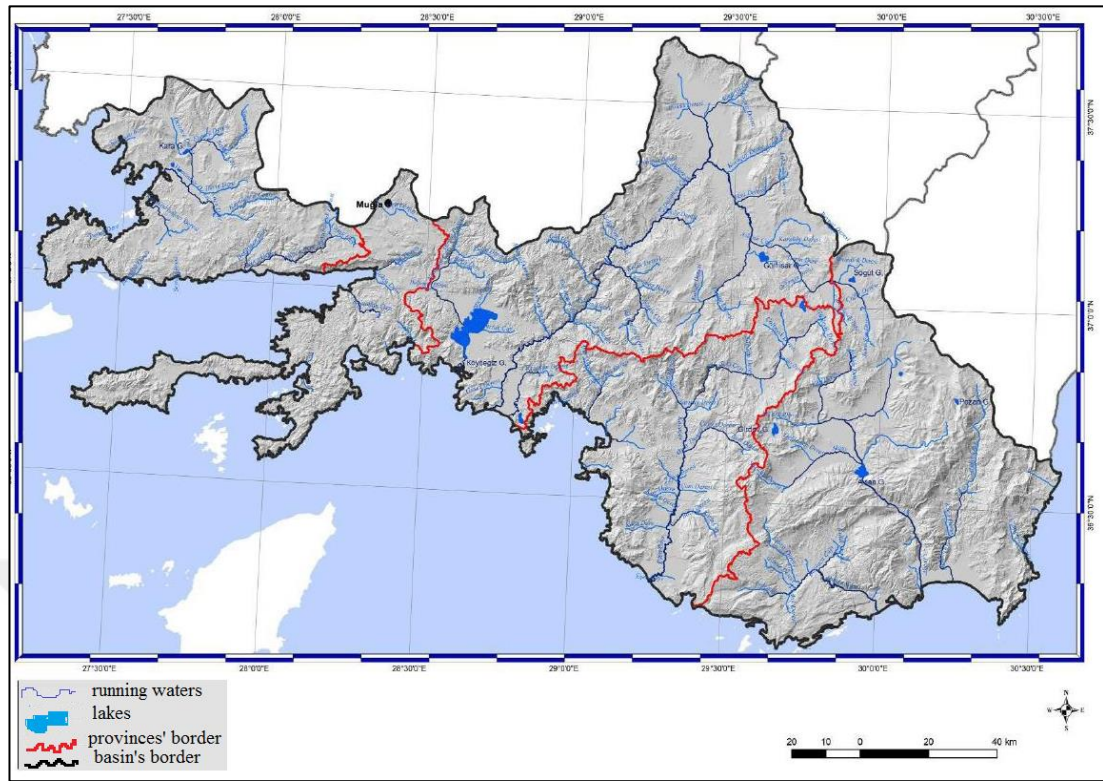


Figure 3.2 Map of lakes and running waters bodies in the western Mediterranean basin of Turkey modified from Anonymous (2016)

3.2 Data sampling and analysis

Data were collected from thirty four (34) freshwater systems including twenty five (25) running waters (streams and creeks) and nine (9) lakes and reservoirs in the western Mediterranean basin of Turkey. The watercourses were seasonally sampled for water physicochemical and biological (algal) analyses from summer 2014 to summer 2015 following standard methods (Kelly et al., 1998; EC, 2003; Gevrey et al., 2004). For phytoplankton identification, hydrobios plankton net was used to collect phytoplankton, while 250 ml water samples were directly taken from just beneath of the surface water to counting of phytoplankton taxa. On the other hand, epilithic diatoms were sampled in riffle areas from stones, by scraping the upper surface of the stones with a toothbrush. Physicochemical factors such as water pH, Temperature (T), Conductivity, Dissolved oxygen (DO), total dissolved solids (TDS), Total Suspended solids (TSS), Biological oxygen demand (BOD₅), Chemical oxygen demand (COD), Salinity, Alkalinity, were directly measured *in situ* from an YSI professional plus oxygen–temperature meter, while the water clarity of the lentic systems (lakes and reservoirs) was measured using a 20-cm Secchi disk.

Geographical data were read from a geographical positioning system (GPS). The chemistry of the watercourses (e.g., TN, N-NH₄, N-NO₃, N-NO₂, TP, and P-PO₄) was analyzed from the taken water sample by an accredited laboratory of DOKAY-ÇED (Ankara).

The phytoplankton and epilithic diatom species were identified under a light microscope (Olympus BX53) attached DP73 model digital camera with imaging software (Olympus CellSens Vers. 1.6) with the assistance of identification tools (Algaebase, Ettl, 1983; Komárek and Fott, 1983; Popovsky and Pfiester, 1990; Krammer and Lange-Bertalot, 1991a, b; Komárek and Anagnostidis, 1998; Krammer and Lange-Bertalot, 1999a, b; John et al., 2002; Wehr and Sheath, 2003; ect.), while phytoplankton taxa were counted using an inverted microscope (Olympus CKX41).

Multivariate and statistical analyses including canonical correspondence analysis (CCA), weighted averaging (WA) regression and Pearson's correlation test were carried out to understand the interactions between environmental parameters and algae species as well as among physicochemical variables.

3.3 Evaluation of water quality

The water quality of the sampled watercourses was evaluated using diatom and phytoplankton based metrics. We assessed first the ecological status of running waters (streams and creeks) following three diatom indices: Trophic index Turkey (Çelekli et al., 2016, 2018), and Eutrophication and Pollution Index Diatom-based (Dell'Uomo, 2004), and trophic index developed by Rott et al. (1999) while the lentic ecosystems (lakes and reservoirs) were assessed based on phytoplankton trophic index (Philips et al., 2013), Mediterranean phytoplankton trophic index (Marchetto et al., 2009), and the phytoplankton assemblage index (Padisák et al., 2006). In the second time, we assessed the trophic status of lakes and reservoirs were assessed following the trophic state index (Carlson, 1977) and the OECD system (OECD, 1982).

CHAPTER IV

ECOLOGICAL PREFERENCES OF BENTIC DIATOM (PHYTOBENTHOS)

ABSTRACT

Running waters bioassessment is one of the challenges for the improvement of water quality in many counties and is getting more and more application since the implementation of the European Union water framework directive which required bioindicators including phytoplankton, phytobenthos, macrophytes, macroinvertebrates and fish to be used for water quality assessment. As an important part of phytobenthos, epilithic diatoms are widely used to in to estimate running water quality level. The main objective of this chapter was to characterize the composition and evaluate the ecological preferences of the diatom assemblages in twenty 25 running water stations in the Western Mediterranean basin of Turkey. The interactions between species and environmental variables was evaluated using canonical correspondence analysis (CCA). A total of 102 epilithic diatom taxa belonging to 22 genera were found. The most dominant genus was *Navicula* (29 species) followed by *Cymbella* (14 species). CCA indicated that the distribution of diatom composition was influenced by physicochemical factors. The first axes gave the species-environment correlations at 89.4% and cumulative percentage variance of species data at 9.4 %. *Clevamphora ovalis*, *Cocconeis placentula*, *Diatoma vulgare* and *Ulnaria ulna* were found to be abundant in polytrophic conditions, in contrast to *Cymbella excisa*, *Fragilaria capucina*, *Gomphonema angustum* and *Gomphonema truncatum* were distributed in eutrophic and polytrophic conditions.

4.1 Introduction

Freshwater ecosystems play a key importance for humans, they provide us a number of benefits in addition to direct use such as drinking water supply, irrigation, fish farming, recreation. Wastewater treatment, flood control, water supply are some of the greatest functions of natural freshwater ecosystems. During the last century, natural continental aquatic systems water quality has been dramatically degraded due to demographic change, industrial revolution, and climate change resulting in severe water pollution and rarity in many regions of the world (Schindler 2006, Smol 2008). Anthropogenic activities have resulted a wide of alterations to aquatic biodiversity in the major part of surface water ecosystems (Allan and Castillo, 2007, Delgado and Pardo, 2014), and on the management of water resources requiring the development of suitable and accurate tools to control the biological integrity of aquatic ecosystems (Cao et al., 2007; Delgado and Pardo, 2014). Habitat deterioration is any variation or perturbation to the environment that became undesirable for living organisms, in this order anthropogenic influences have degraded water catchments, resulting consciousness and the increase in the scientific research in developing biomonitoring methods to evaluate the status of freshwaters (Kelly and Whitton, 1995; Hering et al., 2006; Delgado and Pardo, 2014). Discharge of wastewater, nutrient enriched runoff, and land erosion through agriculture and irrigation practices had a adverse effect on water resources during the last centuries (Billen et al., 2001; Wunsan et al., 2002; Ducharme et al., 2007; Delgado and Pardo, 2014; Toudjani et al. 2017). The occurrence of high amount of nutrients (nitrate, phosphate etc.) associated with anthropogenic activities or natural sources in aquatic ecosystems especially freshwaters could considerably affect the functioning of aquatic primary producers (phytobenthos, phytoplankton, and macrophytes).

The increasing availability of dyes in watercourses decreases light penetration which can reduces primary productivity and causes aesthetic issues (Saratale et al., 2011). Also, several dyes or their metabolites in water resources even at low content can be source of a number of diseases and disorders for living organisms such as allergy, skin irritation, and cancer (Çelekli et al., 2009).

In last decades, a great effort has been done in many regions of the world to monitor water quality using chemical and physical parameters, as well as biological

indicators. In fact, one of the adverse consequences of pollutants is their impact on the biodiversity. Due to this reality, many countries considering the characteristic structure of their water bodies have developed monitoring programs and continually been assessing and categorizing their water resources and used biological indicators for revealing the current situation of their water quality level (Solak and Ács, 2011).

Aquatic organisms are suitable tools to evaluate and monitor anthropogenic pressures on surface watercourses (Solak et al., 2012; Çelekli and Öztürk, 2014; Demir et al., 2014). Aquatic organisms such as hytoplankton, phytobenthos, and macrophytes are largely employed as bioindicators in aquatic ecosystems. The occurrence, absence or proportion of species or composition in aforementioned groups indicates the characteristics of the habitat where they are living (Cellamare et al., 2012). Phytoplankton and phytobenthos, due to their short life periods, respond quickly to variations in the environment (Schaumburg et al., 2004; Cellamare et al., 2012). Phytobenthos species have relatively been used in few studies related to lake and reservoirs than phytoplankton and macrophytes were (e.g. Ács et al., 2005; Sgro et al., 2006; Cellamare et al., 2012; Novais et al., 2012; Bennion et al., 2014; Fidlerová and Hlúbíková 2016; Poikane et al., 2016).

Most researches using phytobenthos (hereafter related specifically to diatoms) were carried out in monitoring rivers and stream (e.g. Round, 1991; Vadeboncoeur and Steinman 2002; Rott et al., 2003; Potapova and Charles, 2003; Ács et al. 2004; Bellinger et al., 2006; Tokatlı and Dayioğlu; 2011; Della Bella et al., 2012; O'Driscoll et al., 2012; Leelahakriengkrai and Peerapornpisal, 2014; Delgado and Pardo, 2014). Diatoms in streams and rivers are major part of phytobenthos and aquatic biodiversity. They form a great proportion of the benthic organisms (often 90–95%) and they have become a suitable indicator for water quality biomonitoring. The advantage in attached diatoms is that they are found in every surface water at any period (Bona et al., 2007; Solak and Ács, 2011; Delgado and Pardo, 2014). Besides, there is huge knowledge on the limnoecology of diatoms and their optimal environmental conditions and tolerance range to water chemistry (Van Dam et al., 1994; Potapova et al., 2004; Schneider et al., 2013; Delgado and Pardo, 2014). Diatoms are often applied for the control of nutrient enrichment due to their high sensitivity and specific capacity to respond spontaneously to a particular

environmental change (Hering et al., 2006; Delgado and Pardo, 2014). Benthic diatoms assimilate dissolved nutrients and use the solar energy for primary productivity, making them key organisms at the base of the grazer's food web, constituting food resources for invertebrates, and fish (Finlay et al., 2002). Periphyton assemblage and distribution are impacted at local scales by environmental parameters such as water nutrients, light, temperature, flow, and type of substrate (Stevenson, 1996; Dodds and Biggs, 2002).

The aims of the chapter were to i) investigate the diatom assemblages and ii) understand the impact of physicochemical variables on the ecological preferences of diatom species in running water ecosystems located in the western Mediterranean basin of Turkey by use of multivariate analyses.

4.2 Materials and Methods

4.2.1 Study Area

The study was conducted in four (4) provinces (Muğla, Burdur, Denizli, and Antalya) of the western Mediterranean basin of Turkey. The western Mediterranean region consists of several running watercourses including Tersakan Deresi, Sarıçay stream, Namnam stream, Kargıcak creek, Dalaman stream, Seki stream, Çayıçi creek, Kocadere/Kızılöz creek, Çavdır stream, Boğluca stream, Akçay stream, Alakır stream, Koca stream, Eşen stream, Karabeyyurdu creek, Delin creek, Kanlı stream. According to Anonymous (2016), Dalaman Stream is the largest running watercourse in this region. The water collection area of the Dalaman Stream is around 3,500 km². Despite the fact that the area for water collection is not very large, the water is often too crowded because the upstream is surrounded by high mountains with abundant rainfall. The minimum amount of water carried by the Dalaman Stream at the moment is seen in August. The Eşen Stream in the Muğla Province is the second largest running water of the Western Mediterranean Basin. A total of 25 stations (Figure 4.1) from aforementioned running watercourses were sampled at seasonal intervals during four seasons (summer and fall 2014, spring and summer 2015). Geographical characteristics and typology of the sampling station are sorted in Table 4.1. Photographs of some studied stations are presented in Figure 4.2.

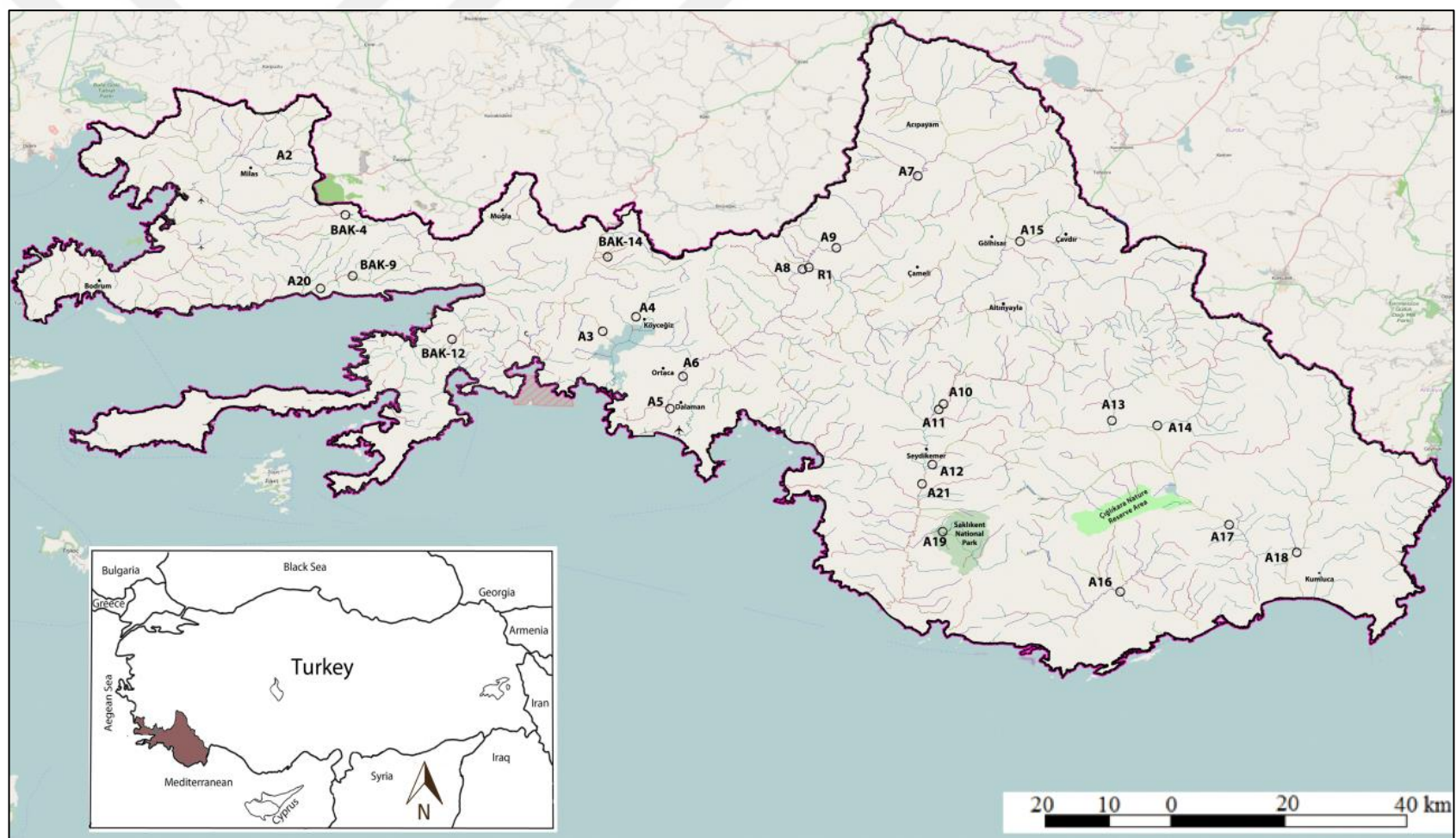


Figure 4.1 Location of sampling stations. For stations' codes see Table 4.1

Table 4.1 Geographical and hydro-morphologic data of the stations

Provinces	Stations	Code	Typology	Latitude (N)	Longitude (E)	Altitude (m)
Muğla	Sarıçay creek	A2	A1R1E2Y2D1J2	37° 22' 50.16"	27° 53' 23.64"	449
Muğla	Namnam creek	A3	A2R1E1Y2D1J2	36° 56' 28.14"	28° 36' 33.659"	12
Muğla	Kargıcak creek	A4	A1R2E2Y2D1J2	36° 45' 42.048"	28° 45' 57.887"	51
Muğla	Dalaman stream	A5	A2R1E1Y2D2J1	36° 50' 3.624"	28° 47' 43.728"	38
Muğla	Dalaman stream	A6	A2R1E1Y2D2J2	37° 18' 19.08"	29° 20' 52.296"	822
Denizli	Dalaman stream	A7	A2R2E2Y2D2J2	37° 5' 20.796"	29° 5' 17.303"	602
Denizli	Dalaman stream	A8	A2R2E1Y2D2J2	37° 8' 16.368"	29° 9' 22.356"	711
Denizli	Dalaman stream	A9	A2R2E1Y2D2J1	36° 46' 17.508"	29° 24' 18.467"	305
Muğla	Seki stream	A10	A2R2E2Y2D1J1	36° 45' 31.644"	29° 23' 47.507"	212
Muğla	Seki stream	A11	A2R2E1Y2D1J1	36° 37' 47.316"	29° 22' 56.027"	153
Muğla	Çayiçi creek	A12	A2R3E2Y2D1J1	37° 9' 3.024"	29° 35' 16.548"	959
Antalya	Kızılöz creek	A13	A2R3E1Y2D1J1	36° 19' 53.759"	29° 49' 12.805"	148
Antalya	Kızılöz creek	A14	A1R2E1Y2D1J1	36° 29' 23.928"	30° 4' 24.78"	370
Burdur	Çavdır stream	A15	A2R2E1Y2D1J2	36° 25' 37.436"	30° 13' 52.356"	101
Antalya	Boğluca stream	A16	A2R1E1Y2D1J1	36° 28' 26.256"	29° 24' 12.419"	129
Antalya	Akçay stream	A17	A2R3E2Y2D2J1	37° 2' 33.18"	27° 56' 49.595"	23
Antalya	Alakır stream	A18	A1R1E1Y2D1J1	36° 35' 8.484"	29° 21' 25.74"	85
Antalya	Eşen stream	A19	A2R2E2Y2D1J2	37° 22' 50.16"	27° 53' 23.64"	449
Muğla	Kanlı stream	A20	A1R1E1Y2D1J1	36° 56' 28.14"	28° 36' 33.659"	12
Muğla	Eşen stream	A21	A2R1E1Y2D2J2	36° 45' 42.048"	28° 45' 57.887"	51
Muğla	Karabeyyurdu creek	BAK 12	R1D2A1J2	36° 55' 23.484"	28° 15' 25.775"	194
Muğla	Delin creek	BAK14	A2R1E1Y2D1J2	37° 5' 20.58"	28° 37' 41.808"	393
Muğla	Kaya creek	BAK4	A2R1E1Y2D1J1	37° 12' 49.968"	28° 0' 19.331"	515

Table 4.1 Continue

Provinces	Stations	Code	Typology	Latitude (N)	Longitude (E)	Altitude (m)
Muğla	Kocabük creek	BAK9	A1R1E1Y2D1J1	37° 4' 18.12"	28° 1' 17.867"	111
Denizli	R1creek	R1	R2D2A2J2	37° 5' 37.032"	29° 5' 31.199"	633

Typology codes (A, flow; R, altitude; J, geology; E, slope; Y, precipitation and D, drainage)

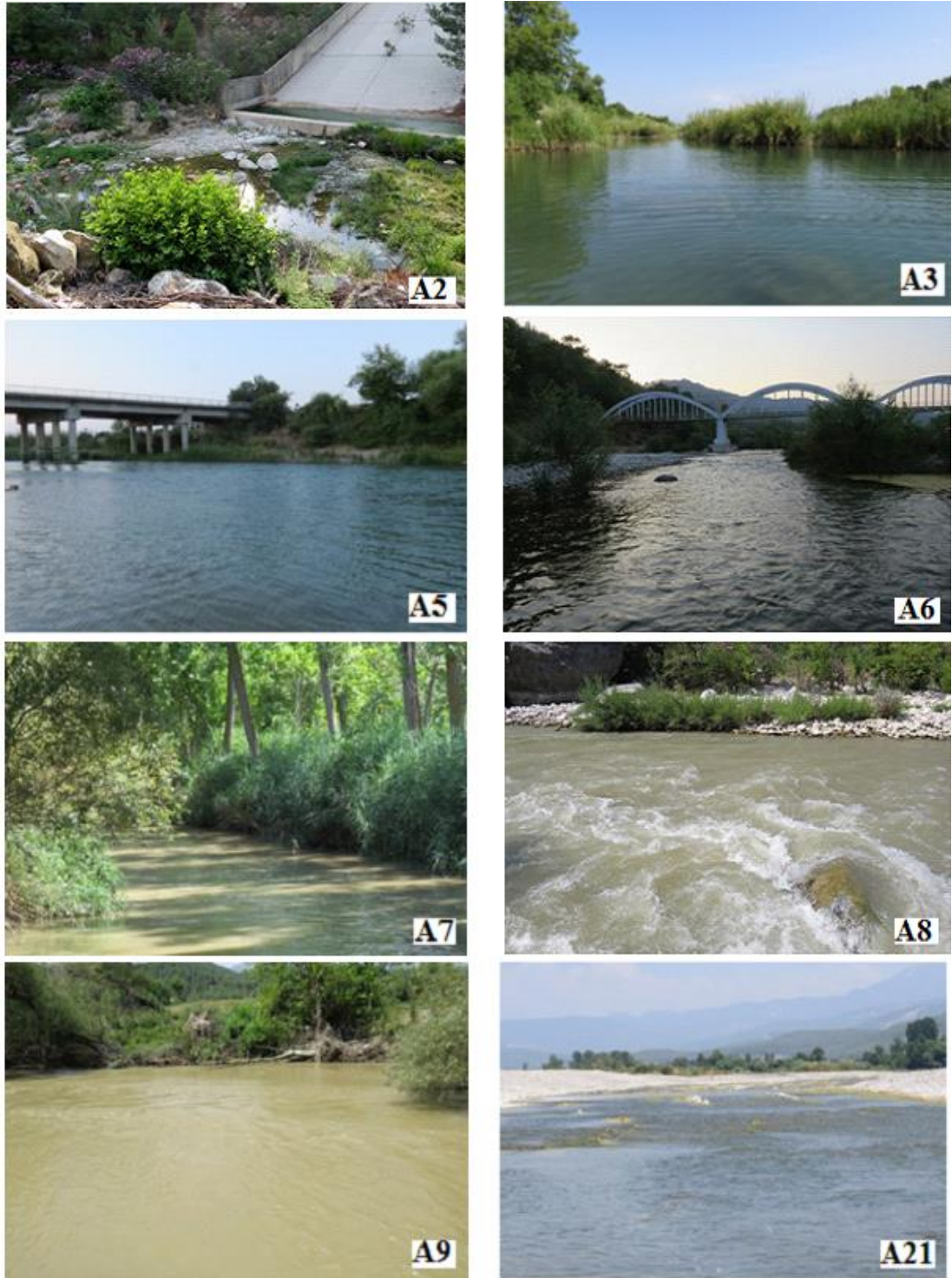


Figure 4.2 Several sampled stations; Sarıçay creek (A2), Namnam creek (A3), Kargıcak creek (A4), Dalaman stream (A5, A6, A7, A8, and A9), Eşen stream (A21)

4.2.2 Field Sampling and Laboratory Techniques

4.2.2.1 Field Sampling

Physico-chemical parameters such as water pH, Temperature (T), Conductivity, Dissolved oxygen (DO), Total Suspended solids (TSS), Biological oxygen demand (BOD₅), Chemical oxygen demand (COD), Salinity, Alkalinity, were directly taken *in situ* using an YSI professional plus oxygen–temperature meter from just beneath of the surface in the stations. Geographical characteristics such as elevation, latitude, and longitude were read from a geographical positioning system (GPS) during different seasons of the study time.

The phytobenthos (epilithic diatom) attached to sediment particles were sampled following to standard sampling methods for lotic (running waters) systems (Kelly et al., 1998; EC, 2003; Gevrey et al., 2004). In clear each epilithic samples were collected from stones, which are not moved under normal hydrological conditions, by scratching the upper surface of the stones by the use of toothbrush (Figure 4.3a). Immediately after collection, periphytic samples were fixed with Lugol's glycerol solution. Collected water samples were then conserved in coolers with ice packs (Figure 4.3b), during the transfer to the laboratory and conserved before analysis (Figure 4.3c).



Figure 4.3 a) sampling; d) transfer in cooler with ice packs; and c) conservation in the laboratory of epilithic diatoms

4.2.2.2 Laboratory Lechniques

4.2.2.2.a Chemical variables measurement

Total organic carbon (TOC), ammonium nitrogen (N-NH₄), nitrite nitrogen (N-NO₂), nitrate nitrogen (N-NO₃), total Kjeldahl nitrogen (TKN), orthophosphate (P-PO₄), total phosphorus (TP), and total nitrogen (TN) were analysed using standard methods (APHA, 1989, 2012).

4.2.2.2.b Diatoms cleaning prossess

Prior to permanent slides preparation, all the cell contents and organic matter were removed from samples. During this process, the hot acid (permanganate) method of cleaning was used according to the European committee for standardization (CEN, 2014). In clear, the sample was homogenized by shaking, 10 ml of the suspension was transferred into beakers and 2 ml of potassium permanganate (KMnO₄) solution was added. 7 ml of diluted hydrochloric acid (HCl) was then added after 24 hours and heated on a hot plate at about 90 ± 5 °C. Distilled water was added at least three times, or until all traces of hydrochloric acid had been removed. The beaker was then removed from the heat and the content (cleaned diatom suspension) was transferred to a centrifuge tube topped up with distilled water then centrifuged at 6000 rpm for 3 minutes. Centrifugation was repeated three to five times (Figure 4.4).

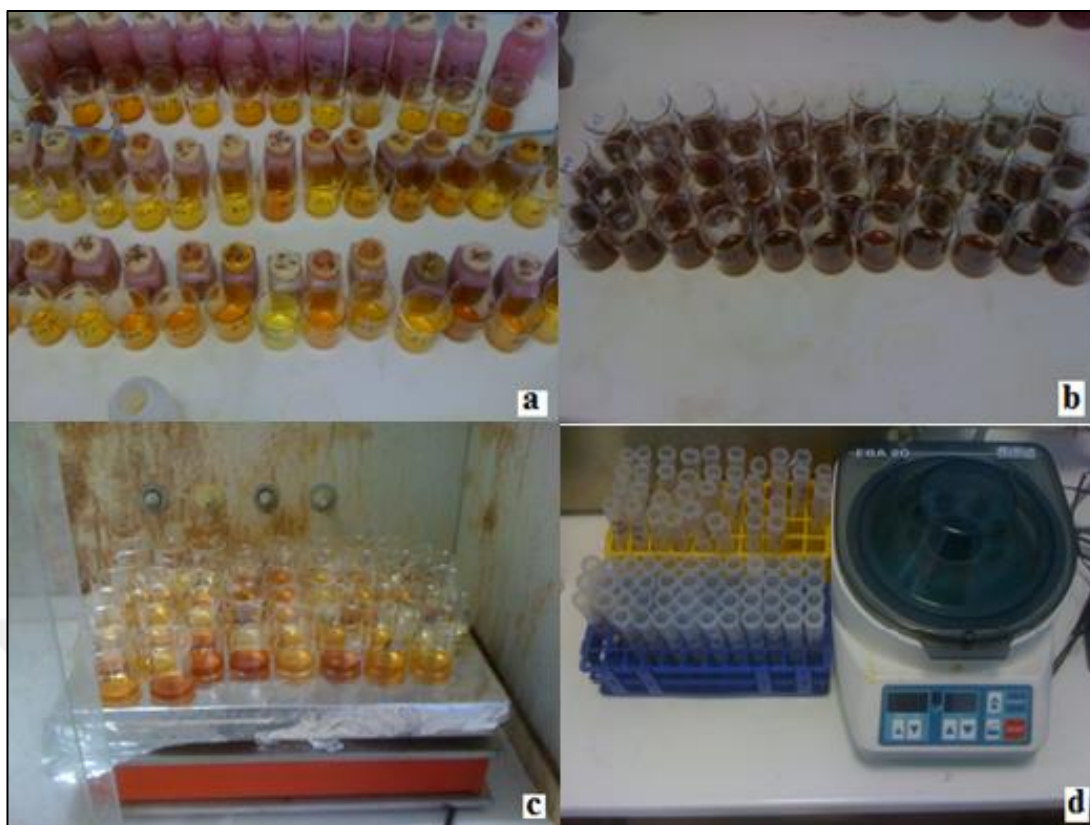


Figure 4.4 a) Transfer of 10 ml to beaker, b) After addition of KMnO_4 and HCl , c) heating and d) centrifugation process

4.2.2.2.c Preparation of permanent slides

The cleaned diatom suspension was diluted to a suitable concentration and permanent slide mounts (Figure 4.5) were then prepared according to standard methods (CEN, 2003, 2014) for each sample for the enumeration of diatom species.

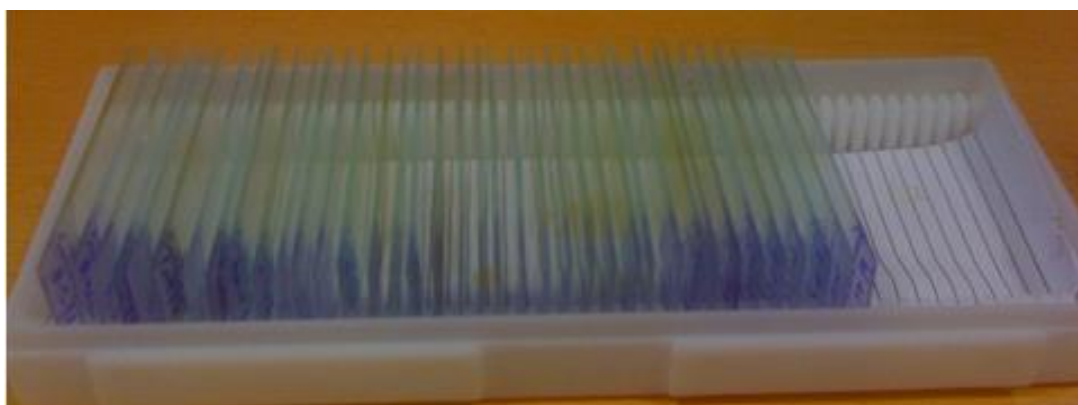


Figure 4.5 Prepared permanent slides

4.2.2.2.d Microscopic examination and identification

Diatoms identification and analyses were carried out in the hydrobiology's laboratory of the Biology department of Gaziantep University (Turkey) according to the standard methods (CEN, 2004, 2014). The prepared permanent slides were later examined at 1000x magnification using oil immersion microscopy to estimate taxonomic composition and relative abundance of with Olympus light microscope (Olympus BX53) attached the DP73 model digital camera with imaging software (Olympus CellSens Vers. 1.6) (Figure 4.6). The phytobenthos samples were identified and counted at species level according to primary identification diatom checklists and books (e.g., Krammer and Lange-Bertalot, 1991a, b, 1999a, b, Lange-Bertalot, 2001; Krammer, 2000, 2002). Relative percentages and biovolume of benthic algal were then calculated and used for statistical analyses and interpretation relevant to the assessment of water quality.



Figure 4.6 Diatom examination and identification under Olympus light microscope (Olympus BX53)

4.2.3 Statistical Analyses

In order to estimate the relationship between the diatom species and physicochemical parameters a direct gradient analysis technique, canonical correspondence analysis (CCA) (ter Braak and Šmilauer, 2002) was used. Prior to this, detrended correspondence analysis (DCA), which is a unimodal (Leps and Šmilauer, 2003) was used at a gradient length more than 3.0. Environmental variables were measured in the study area at the 25 stations for analysis. Except pH, these variables were transformed using the function $\log(x+1)$, to decrease skewness (ter Braak and Šmilauer, 1998). Statistical significance of environmental variables was estimated using the Pearson correlation test of SPSS statistical program. The transformed environmental variables and diatom abundances were analyzed using canonical correspondence analysis (CCA) with CANOCO for Windows 4.5 package (ter Braak and Šmilauer, 2002) to relate diatom assemblage structure to all predictor environmental variables and to explore the relationships among and between species and the environmental variables. A total of 8 environmental variables and species having a proportion higher than 1% were used for the CCA ordination. The significance of environmental factors to explain the variance of species data in CCA was examined following the forward selection of Monte Carlo permutation test. A weighted average (WA) regression was done to evaluate diatom taxa optima (u_k) and tolerance (t_k) values for each of the environmental variables.

The hypothesis of no difference in diatom composition (similarities in species composition) between the running watercourses was tested by means of vegetation tables (species-by-sites matrix) used by Schaumburg et al. (2004), while the Monte Carlo permutation test was used to test the hypothesis on relationship between the diatom taxa and the environmental factors. On the other hand, the hypothesis of no difference between the ecological status of running water bodies, the variance analysis (one-way ANOVA) was employed at 95% confidence interval using the program SPSS (IBM statistics version 23).

4.3 Results

4.3.1 Physicochemical variables

Physico-chemical variables of the sites are summarized in (Table 4.2). The highest mean concentrations of TN ($2208.8 \mu\text{g L}^{-1}$) and N-NO₃ ($1503.3 \mu\text{g L}^{-1}$) were found in Çavdır Stream and Dalaman stream (A8) which were also located at higher altitudes 936 m and 821 m, respectively. With regard to phosphorous, the highest TP value ($2263.3 \mu\text{g L}^{-1}$) was recorded in Boğluca stream, while the highest P-PO₄ value ($240.0 \mu\text{g L}^{-1}$) was found in Sarı stream. The water bodies showed alkaline condition and their pH ranged between 8.1 and 8.7. With regard to geographical parameters, the lowest (14.2°C) and highest (23.8°C) mean of temperature were measured in Delin creek and Çayıçi Creek, respectively. Conductivity and salinity showed their highest values $823.2 \mu\text{S cm}^{-1}$ and 0.4 ppt in Dalaman stream (A7) and Çavdır Stream. The Pearson's correlation test indicated positive and negative significant relationships among environmental variables (Table 4.3). Temperature is positively correlated with pH ($r=0.432$, $p<0.01$), conductivity is positively correlated with total nitrogen ($r=0.574$, $p<0.01$) while dissolved oxygen was negatively correlated with total suspended solid ($r=-0.266$, $p<0.05$) and salinity ($r=-0.319$, $p<0.05$).

The seasonal distribution of BOD₅ and the nutrients especially TP, TN and N-NO₃ in the water bodies during the study period are given in Figure 4.7. According to the boxplots of Figure 4.7, there was no uniform distribution of the environmental variables in all running sampling stations.

Table 4.2 Mean values of environmental variables combined from sampling stations. For stations' codes Table 4.1

	Alti m	Temp °C	Cond μS cm ⁻¹	pH	BOD mg L ⁻¹	TN μg L ⁻¹	N-NH ₄ μg L ⁻¹	N-NO ₂ μg L ⁻¹	N-NO ₃ μg L ⁻¹	TP μg L ⁻¹	P-PO ₄ μg L ⁻¹	Salinity ppt	CaCO ₃ mg L ⁻¹
A2	433	18.1	179.4	8.3	14.5	1037.3	123.3	10.7	226.7	443.3	240.0	0.10	10.0
A3	15	20.1	535.5	8.7	8.93	724.7	128.3	8.25	262.5	337.8	179.5	0.27	30.0
A4	36	20.7	318	8.7	5.30	194.4	100.0	2.0	100.0	50.0	10.0	0.17	10.0
A5	18	18.4	495.0	8.4	15.8	990.0	193.0	31.3	390.0	336.0	175.5	0.26	25.0
A6	31	18.0	477.3	8.4	12.3	936.0	226.8	9.15	292.5	357.5	183.3	0.26	20.0
A7	603	19.7	627.81	8.7	7.93	1786.3	117.5	34.5	1307.5	386.5	192.8	0.34	27.5
A8	821	16.1	773.3	8.3	8.33	4029.7	1220.0	223.3	1503.3	783.3	341.7	0.43	25.0
A9	710	19.4	776.5	8.6	5.65	1722.3	125.0	27.8	1030.0	374.3	214.0	0.42	35.0
A10	264	14.5	331.7	8.1	5.07	574.2	126.7	6.60	306.7	184.3	137.3	0.18	20.0
A11	200	16.0	330.0	8.5	11.7	940.5	438.0	81.5	680.0	240.5	163.8	0.16	17.5
A13	1060	17.8	301	8.5	5.10	755.0	100.0	12.1	370.0	64.0	10.5	0.17	15.0
A14	1043	14.6	670	7.4	5.20	1089.0	100.0	4.9	140.0	90.08	23.0	0.40	20.0
A12	151	23.8	352.0	8.7	4.00	615.8	115.0	4.78	182.5	164.0	111.0	0.18	15.0
A15	936	17.8	623.3	8.7	10.8	2208.8	195.0	33.0	1340.0	175.3	106.3	0.34	32.5
A16	135	18.7	484.3	8.4	7.67	486.4	130.0	2.83	260.0	596.3	110.0	0.24	27.5
A17	341	17.0	281.5	8.5	5.00	930.9	125.0	20.3	632.5	122.0	105.0	0.15	12.5
A18	64	19.3	344.5	8.2	4.00	479.8	100.0	3.45	165.0	38.5	10.0	0.17	17.5
A19	98	15.6	319.8	8.2	5.50	763.2	112.5	4.75	480.0	295.8	183.8	0.17	25.0
A20	44	20.7	380.0	8.1	7.95	254.8	100.0	3.00	100.0	30.0	10.0	0.20	15.0
A21	85	22.8	439.0	8.7	8.42	1265.0	123.3	23.8	917.5	170.5	117.5	0.22	25.0
Bak4	516	16.8	234.0	8.4	9.50	149.5	100.0	4.00	100.0	50.0	10.0	0.13	15.0
Bak9	86	16.4	312.0	8.3	10.2	208.9	100.0	3.00	100.0	40.0	10.0	0.18	10.0
Bak12	144	14.2	563.0	8.4	4.00	285.0	100.0	2.00	100.0	40.0	10.0	0.35	30.0
Bak14	375	18.2	271.9	8.6	6.5	609.9	323.0	21.0	150.0	54.5	21.5	0.15	17.5
R1	615	20.5	504.0	8.8	4.41	231.7	100.0	3.05	130.0	65.0	10.0	0.27	27.5

Table 4.3 Pearson's correlation test result among the environmental variables. Abbreviations of physico-chemical variables are given in the list of symbols/abbreviations

	Altitude	pH	Temperature	Conductivity	BOD ₅	TN	N-NO ₂	N-NO ₃	TP	P-PO ₄
Altitude										
pH	-0,081									
Temperature	-0,104	0,432**								
Conductivity	0,244	0,133	0,488**							
DO	-0,149	0,321**	-0,303*	-0,360**						
TSS	0,220	-0,079	0,057	0,286*						
BOD ₅	-0,163	0,174	0,243	0,150						
COD	-0,145	0,158	0,244	0,111	0,961**					
TOC	-0,132	0,262*	0,281*	0,006	0,591**					
TN	0,427**	0,050	0,074	0,574**	0,059					
N-NH ₄	0,055	0,011	-0,030	0,214	0,011	0,463**				
N-NO ₂	0,284*	-0,011	0,098	0,493**	-0,038	0,687**				
N-NO ₃	0,397**	0,185	0,008	0,468**	-0,057	0,814**	0,455**			
TKN	0,353**	-0,097	0,192	0,441**	0,138	0,778**	0,483**	0,348**		
TP	-0,139	0,163	-0,139	0,009	0,207	0,225	0,112	0,261*		
P-PO ₄	-0,150	0,158	-0,141	-0,008	0,180	0,238	0,109	0,275*	0,987**	
Salinity	0,324**	0,039	0,312*	0,955**	0,074	0,587**	0,459**	0,478**	0,024	0,024
Alkalinity	0,086	0,177	0,256*	0,573**	0,101	0,212	0,085	0,320**	0,045	0,032

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

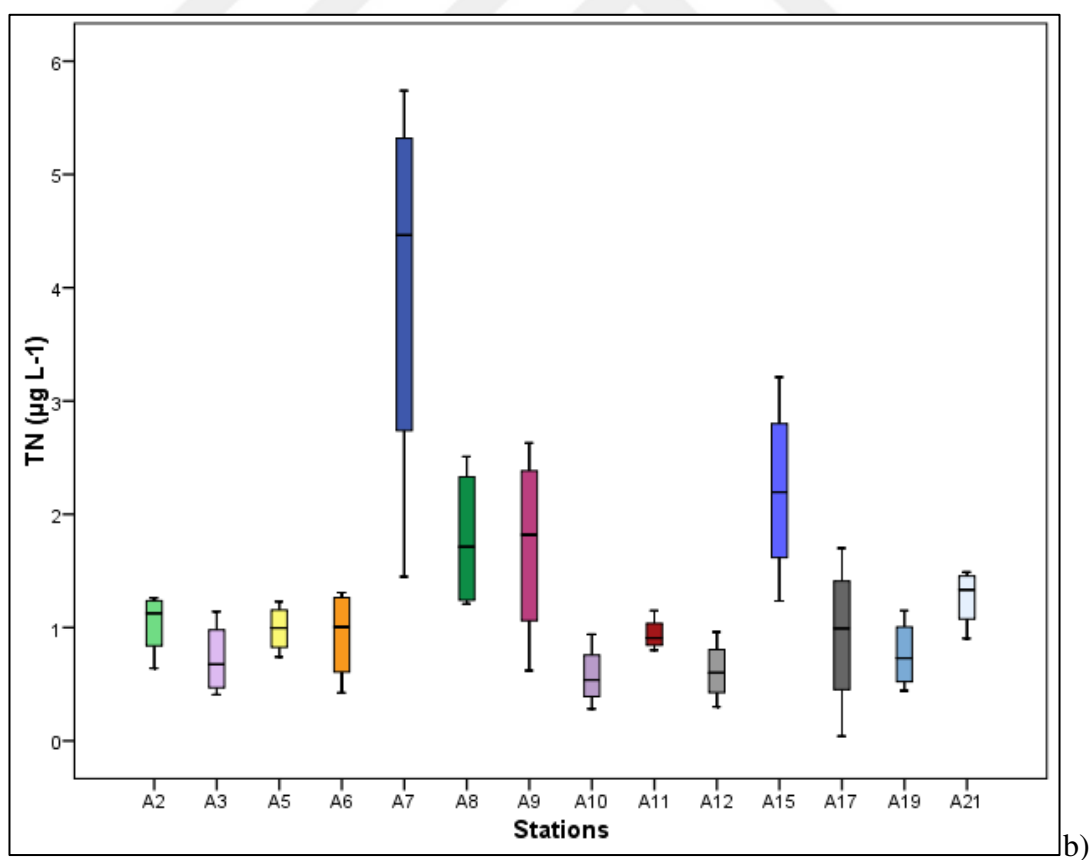
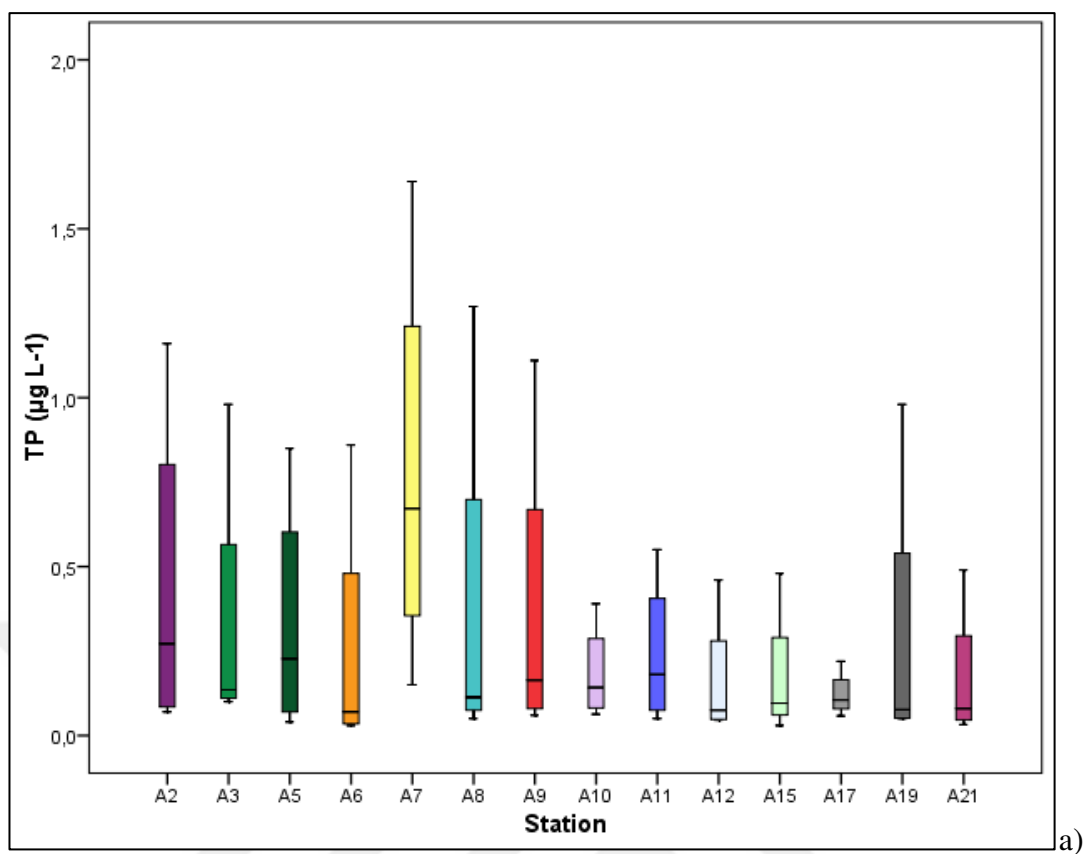


Figure 4.7a Box plot diagrams showing the distribution of a) TP and b) TN in the running water bodies

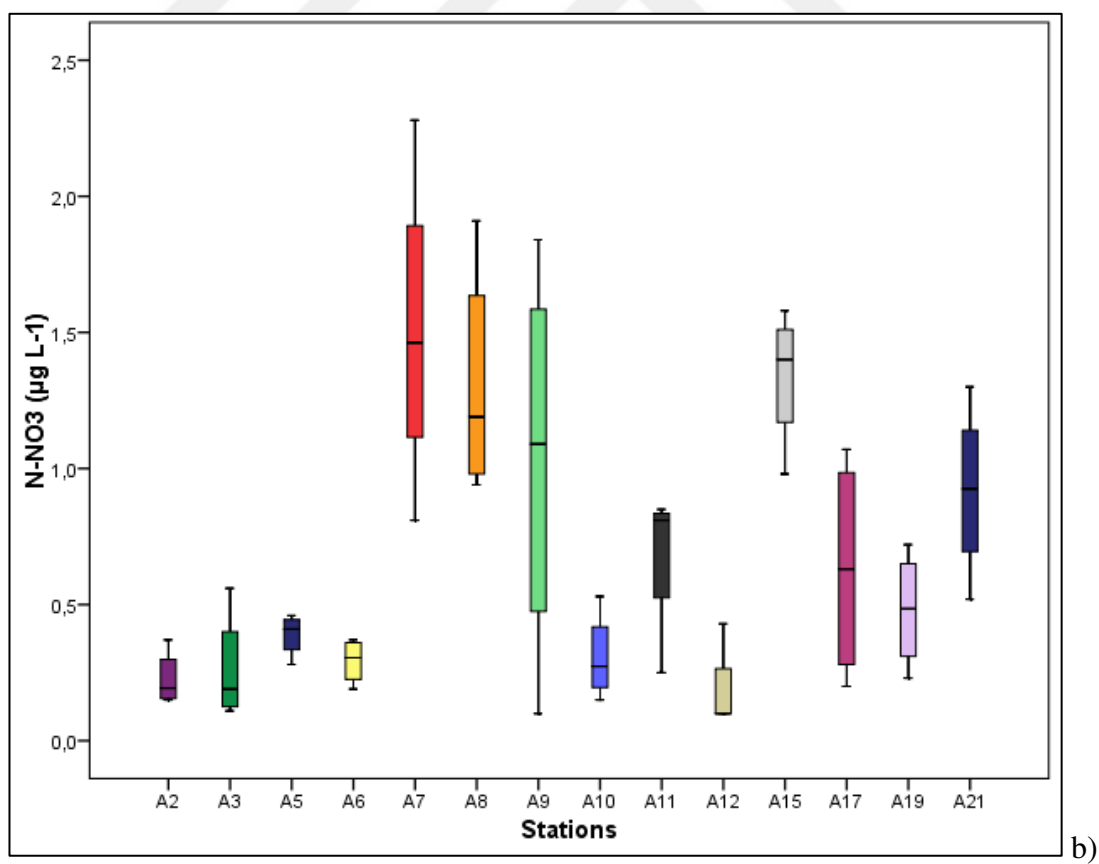
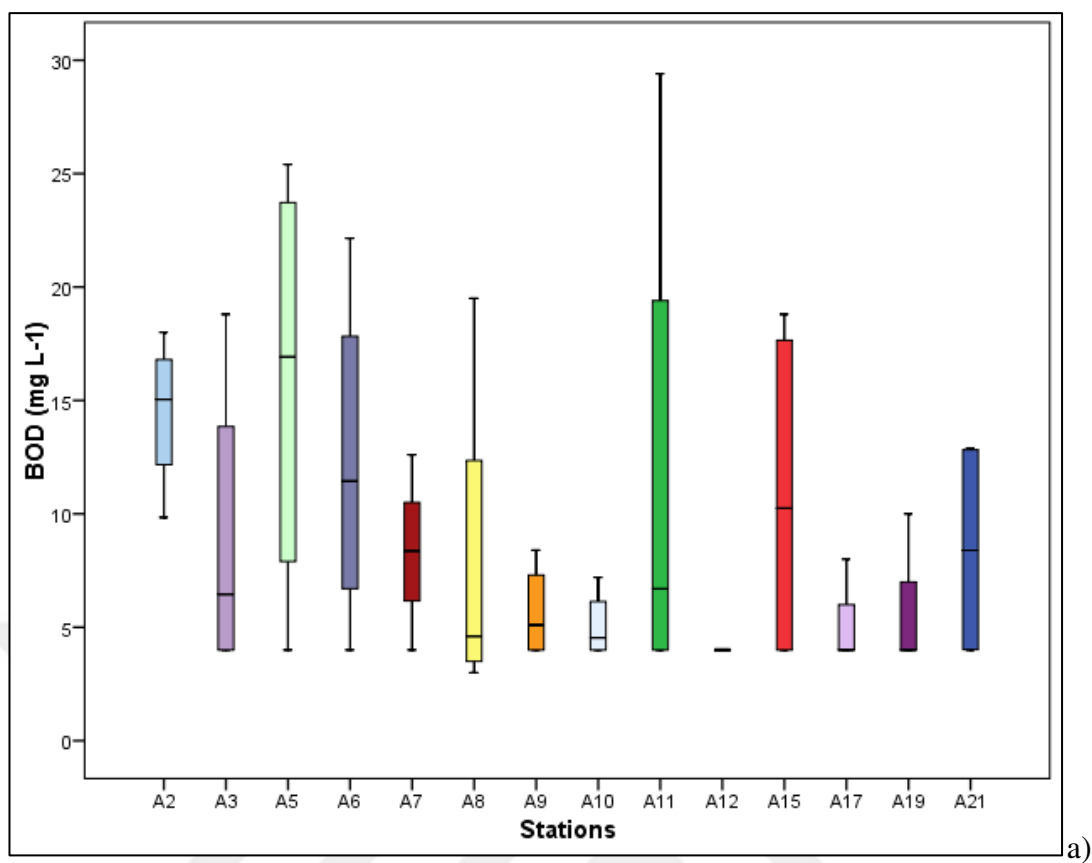


Figure 4.7b Box plot diagrams showing the distribution of a) BOD and N-NO₃ in the running water bodies

4.3.2 Diatom Species Richness and Assemblages

During the four seasons covering the study time, a maximum of 102 epilithic diatom taxa distributed in 22 genera including *Achnanthes*, *Amphora*, *Aulocoseria*, *Craticula*, *Cyclotella*, *Melosira*, *Hantzschia*, *Navicula* and others were identified (Appendix A). The most dominant genera were *Navicula* and *Cymbella* followed by *Nitzschia*, *Gomphonema*, *Fragilaria*, and *Cocconeis*. The most common and abundant species found during the study were *F. capucina*, *Cymbella excisa*, *Gomphonema parvulum*, *Ulnaria ulna* and *Cocconeis placentula* (Table 4.4). *Fragilaria capucina* species was recorded in all sites, while some of the rarely observed species were *Amphora inariensis*, *Cocconeis disculus*, *Diploneis modica*, *Cymatopleura amphicephala*, *Epithemia adnata*, *Navicula expecta* and *Surirella minuta*. The species number per site varied from a maximum of 28 at Dalaman stream (A8) and Eşen stream (A21) in the Muğla province to a minimum of 20 at Dalaman stream (A7), Ak stream (A17), Eşen stream (A19) in the Antalya province, Koca stream (A20), Karabeyyurdu Creek (BAK 12) and Kocabük Creek (BAK 9) (Figure 4.8). A slight change was observed in the seasonal diatom composition. A total of 49 diatom taxa were recorded in spring 2015, while 38 taxa were identified in fall 2014 (Figure 4.9). the list of species recorded in the different station in each season is presented in Appendices B, C, D, and E. Some of the diatom species recorded during the present study are shown in Figure 4.10.

Table 4.4 The most common and abundant species found during the study

Species	Maximum (%)	Mean (%)	Number of site
<i>Fragilaria capucina</i> Desmazières	28.28	7.4	25
<i>Cymbella excisa</i> Kützing	38.33	15.68	21
<i>Gomphonema parvulum</i> Kützing	60.00	13.81	21
<i>Ulnaria ulna</i> (Nitzsch) Compère	26.48	7.25	21
<i>Cocconeis placentula</i> Ehrenberg	80.00	15.47	20
<i>Diatoma vulgaris</i> Bory	9.27	3.48	17
<i>Cyclotella ocellata</i> Pantocsek	12.76	3.96	15
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	3.61	2.16	13
<i>Gomphonema truncatum</i> Ehrenberg	30.61	8.22	10
<i>Navicula oppugnata</i> Hustedt	7.07	2.61	10
<i>Navicula laterostrata</i> Hustedt	7.70	2.46	10
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	8.96	3.37	9
<i>Cymbella minuta</i> Hilse in Rabenhorst	30.03	5.73	8
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	15.76	4.83	8

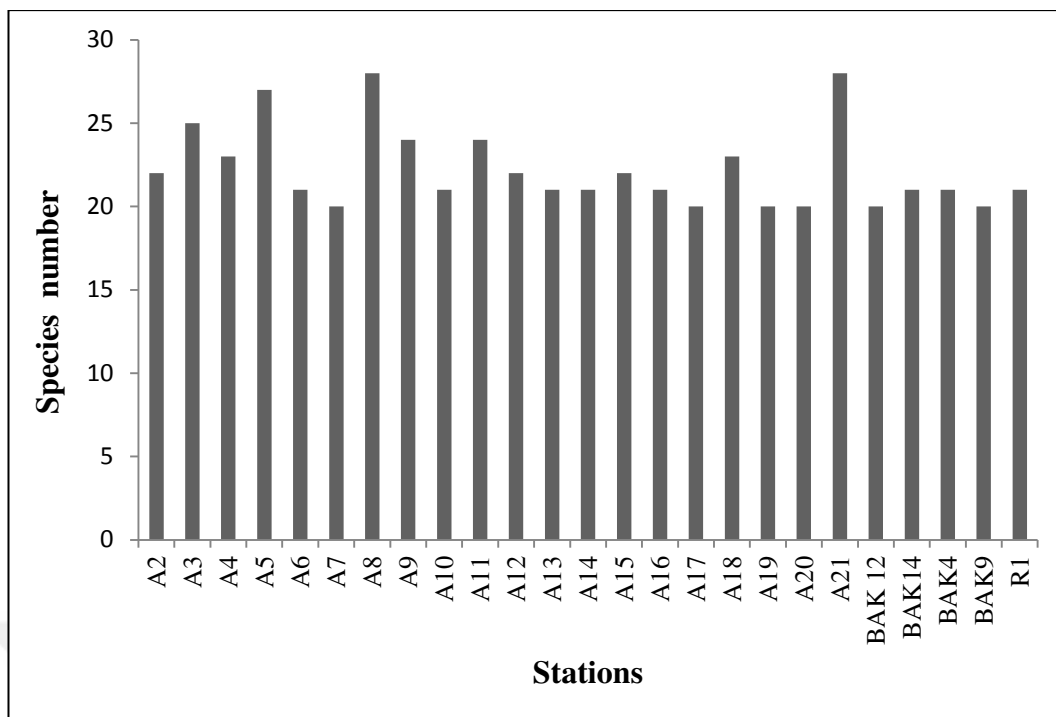


Figure 4.8 Species number by site during the study. For station codes, see table 4.1

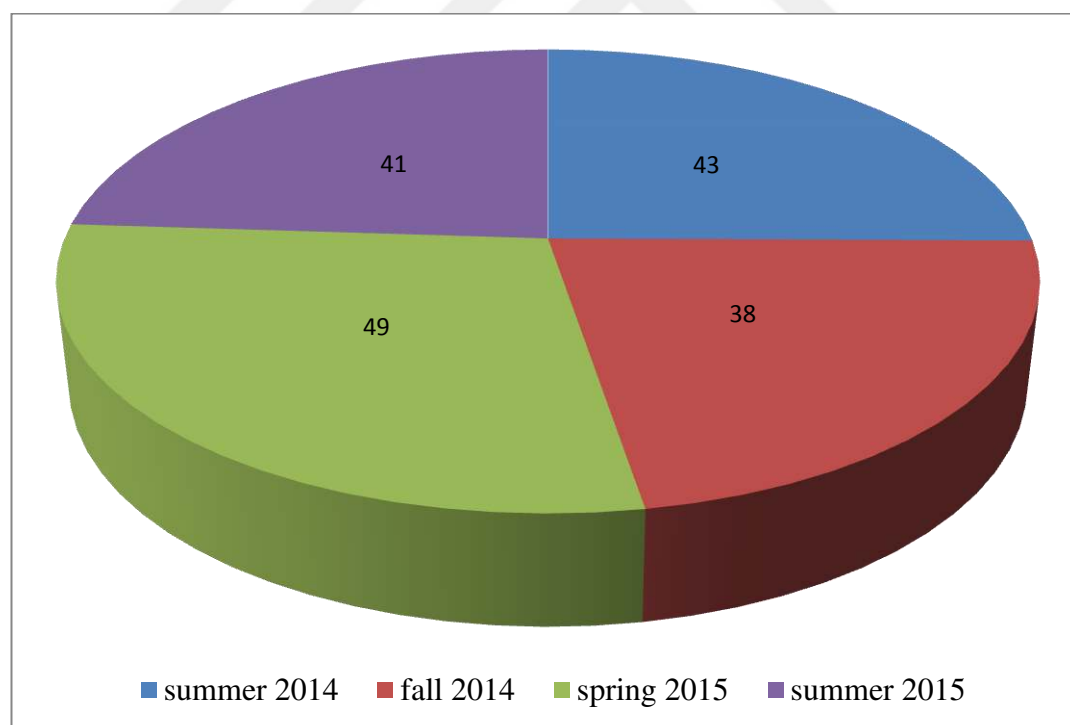


Figure 4.9 Epilithic diatom species number per season in the running waters

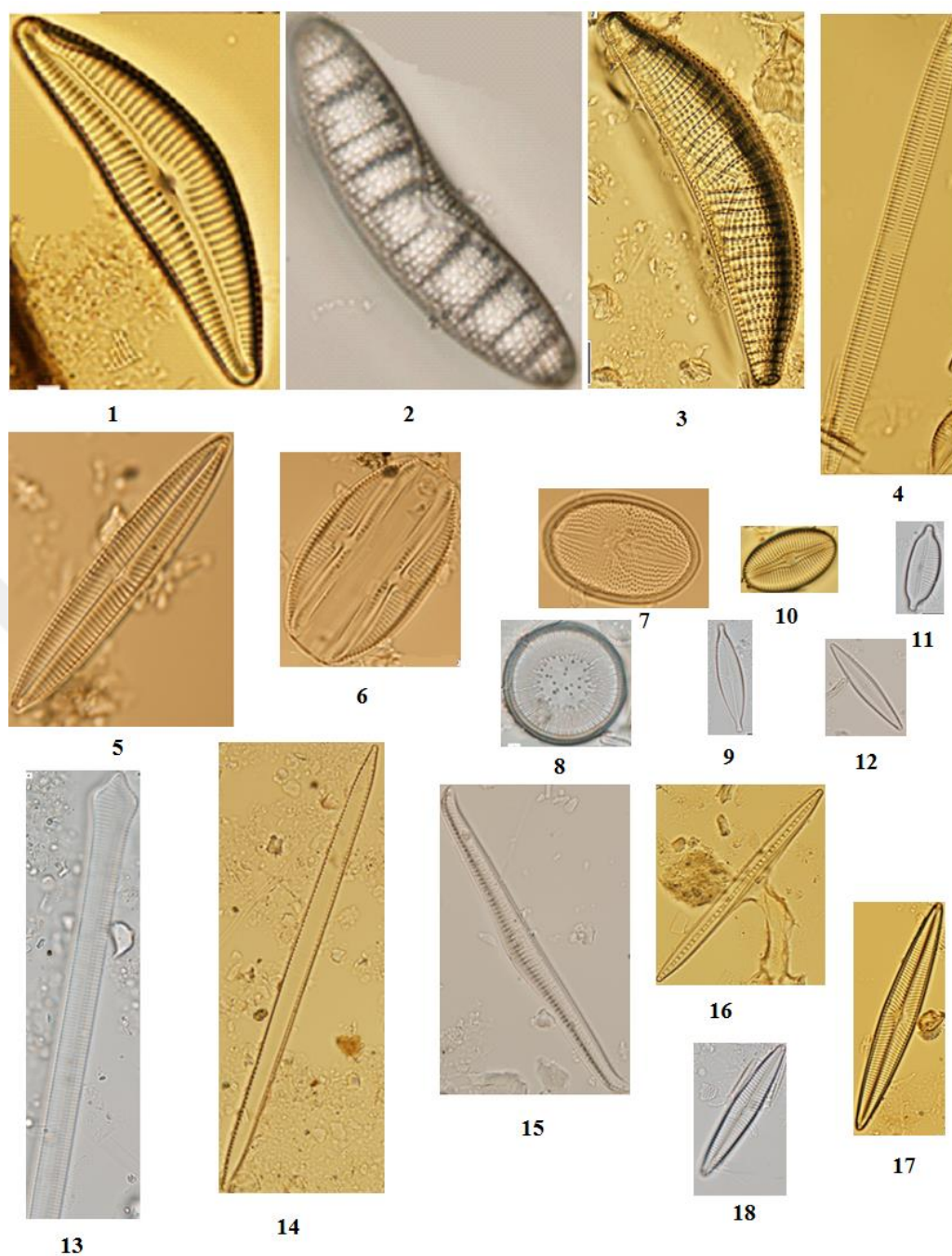


Figure 4.10 1) *Cymbella excisa*; 2) *Epithemia adnata*, 3) *E. turgida*; 4) *Ulnaria ulna*; 5) *Navicula tripunctata*, 6) *Clevamphora ovalis*, 7) *Cocconeis placentula*, 8) *Cyclotella meneghiniana*; 9) *Navicula cryptocephala*; 10) *Diploneis modica*; 11) *Cymboplectra amphicephala*; 12) *Gomphonema parvulum*; 13) *Ulnaria capitata*; 14) *Nitzschia sigmoidea*; 15) *Rhopalodia gibba*; 16) *Nitzschia dissipata*; 17) *Navicula cryptotenella*; 18) *N. radiosa*

4.3.3 Species environment relationships

The result of the Canonical Correspondence Analysis (CCA) using Monte Carlo permutation test for epilithic diatom species-environment variables relationship

showed that the distribution of diatom composition was governed by physicochemical variables and the first axes gave the species-environment correlations at 89.4% and Cumulative percentage variance of species data at 9.4 % (Table 4.5). The most structuring factors were temperature, Altitude conductivity, CaCO₃, TP, N-NO₃, and P-PO₄.

Table 4.5 Summary of canonical correspondence analysis using Monte Carlo permutation test for epilithic diatom species-environment variables relationship

Axes	λ_1	λ_2	λ_3	λ_4	T. inertia
Eigen values	0.507	0.355	0.333	0.259	9.155
Species-environment correlations	0.906	0.894	0.865	0.823	
Cumulative percentage variance of species data	5.5	9.4	13.0	15.9	
of species-environment relation	24.6	41.8	58.0	70.5	
Sum of all Eigen values					9.155
Sum of all canonical Eigen values					2.060
Test of significance of first canonical axis : Eigen value = 0.507					
	F-ratio = 3.222		p-value = 0.0240		

The ordination of CCA indicated that *Nitzschia umbonata*, *N. angustata*, *N. dissipata*, *Surirella angusta*, *Surirella brebissonii*, *Diatoma moniliformis*, and *Cyclotella meneghiniana* were assigned with high altitude and high content of N-NO₃ in Dalaman stream (A8 and A9) and Ak stream, while *Gomphonema minutum* and *C. placentula* preferred low concentration in Ak stream. *N. lanceolata*, *G. attenuatum*, *S. minuta* were correlated with high water conductivity and P-PO₄ in Seki stream, Boğluca stream and Eşen stream (A19), contrary to *D. vulgaris* and *Epithemia frickei* which preferred low P-PO₄ in Dalaman stream (A5), when *C. solea*, *Cymbella hantzschiana*, and *Cymbella excisa* were associated with low conductivity. *Cymbella minuta*, *N. capitatoradiata*, *Achnanthes impexa*, *Cyclotella iris*, *Navicula bryophila*, *Cymbella cistula* and *Navicula tuscula* were found at low temperature and low alkalinity (CaCO₃) in Eşen stream (A19) and Dalaman stream (A5) (Figure 4.11).

With regard to water temperature, *A. impexa*, *Cymbopleura naviculiformis*, *N. capitatoradiata*, *N. bryophila*, *N. cryptotenella* preferred warm water in contrast to *C. disculus*, *Gomphonema pseudoaugur*, *S. angusta*, *S. brebissonii* and *Lysigonium lineatum* were related to low temperature.

In contrast to *Nitzschia flexa*, *N. cryptotenella*, *N. bryophila*, *Gomphonema angustum*, *Cymbella simonseni*, *C. cymbiformis*, *C. naviculiformis* which were at low altitude, species such as *G. pseudoaugur*, *N. umbonata*, *S. minuta* were located at high altitude. *Cyclotella meneghiniana*, *G. attenuatum*, *N. dissipata*, *N. umbonata* and *S. minuta* were associated with high conductivity, while *Surirella subsalsa*, *M. varians*, *G. pseudoaugur*, *Gomphonema clavatum*, and *C. ocellata* preferred low conductivity.

The trophic preferences of diatom species established based on TP classes as: oligotrophic (<10 µg/l), mesotrophic (10-30 µg/l), eutrophic (30-100 µg/l), polytrophic (>100 µg/l) were determined based on the mean abundance of each species in each trophic class. The result indicated that species such as *Clevamphora ovalis*, *Cocconeis placentula*, *D. vulgaris* and *Ulnaria ulna* indicated a preferential occurrence polytrophic conditions, while species such as *Fragilaria capucina*, *Gomphonema angustum* and *G. truncatum* were distributed in eutrophic and polytrophic conditions.

Table 4.6 Weighted average regression in sampling stations, u_k and t_k indicated optima and tolerance, respectively. For species code see Appendix A

	Temperature			Altitude		Conductivity		TP		P-PO ₄		TN		N-NO ₃		CaCO ₃	
	N2	°C		m		µS cm ⁻¹		µg L ⁻¹		µg L ⁻¹		µg L ⁻¹		µg L ⁻¹		mg L ⁻¹	
Code		uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut
Acim	1.12	26.06	0.71	152	56	435.1	112.0	253.9	678.8	123.0	289.9	598.9	615.2	149.4	622.3	15.84	10.61
Amov	4.41	22.78	3.05	106	257	715.4	244.0	224.7	129.5	94.7	58.8	1227.8	1073.6	334.9	410.7	24.20	6.12
Augr	8.74	16.50	3.99	502	424	410.8	269.5	374.8	380.7	172.7	192.5	1098.3	891.3	511.2	415.3	21.48	8.23
Codi	1.16	14.50	2.90	140	35	546.7	157.0	41.5	14.1	10.0	120.0	355.6	680.2	156.5	544.5	29.63	3.54
Copl	8.28	20.53	4.37	435	301	688.1	386.9	220.3	160.1	100.8	72.0	1537.7	1039.2	972.0	683.6	22.51	7.77
Coplo	1.53	23.43	5.16	164	605	513.3	127.9	114.9	48.0	22.7	22.0	566.0	272.4	148.4	73.3	27.33	9.63
Cyir	4.56	19.65	5.39	128	236	557.6	320.9	549.3	370.0	257.0	188.6	1113.0	1273.0	392.1	349.2	25.36	5.24
Cyme	1.26	23.84	1.05	669	372	1165.1	251.9	219.2	167.0	107.2	71.4	2642.5	1342.5	1714.7	875.3	20.72	5.47
Cyoc	9.70	16.49	3.71	278	334	294.0	118.0	251.8	363.0	102.8	167.7	701.2	330.2	274.4	165.8	17.03	5.52
Cyso	1.13	23.69	5.61	607	238	871.9	330.7	197.8	61.1	97.1	41.6	2112.8	671.3	1332.0	492.9	39.51	9.05
Cyaf	9.87	21.32	4.94	197	320	427.0	216.9	148.7	103.0	80.5	43.3	403.4	353.6	128.7	107.0	23.78	6.53
Cyci	2.52	23.72	1.53	348	473	708.3	176.0	185.8	83.1	70.0	55.8	868.9	838.0	379.6	507.9	28.32	5.41
Cyey	1.16	23.16	8.13	22	113	775.6	351.4	258.3	359.9	109.9	152.7	1105.6	332.3	124.6	141.4	24.63	3.54
Cyex	3.73	21.92	4.94	305	347	821.8	307.7	217.3	22.3	100.8	1.1	1440.5	982.8	780.4	847.7	24.42	6.73
Cymi	2.81	16.05	4.79	130	119	468.5	181.1	191.8	78.7	90.7	39.4	813.6	340.3	465.8	290.8	24.11	4.27
Cyna	2.23	27.12	6.03	83	52	388.7	51.8	32.7	11.0	16.4	14.3	387.6	272.4	156.0	192.3	19.33	4.16
Cysi	1.93	24.68	0.48	45	189	543.1	35.4	173.5	119.0	69.6	48.3	621.9	356.9	290.0	177.2	26.73	3.42
Cytu	1.84	21.93	3.70	475	70	328.3	287.9	194.0	104.2	82.4	51.9	1142.9	816.4	275.4	552.6	14.82	13.01
Deel	1.61	22.29	5.07	99	271	659.1	464.5	185.9	106.3	75.8	59.8	1132.9	251.7	166.7	199.1	22.11	10.13
Dimo	1.27	13.76	3.32	542	334	485.0	106.1	1176.4	551.5	558.0	247.5	2387.6	721.2	1836.8	431.3	16.20	7.07
Dite	3.24	24.75	5.16	142	316	470.9	111.7	123.2	116.1	29.8	31.1	800.5	972.9	318.3	446.0	25.36	7.04
Divu	6.30	17.32	5.09	256	245	529.9	265.6	231.4	190.9	107.8	87.5	984.5	936.8	519.8	526.4	21.66	3.86
Frbi	1.48	15.95	4.84	121	352	421.5	435.5	101.1	244.2	34.4	117.1	1032.4	1970.1	349.3	584.3	16.87	7.54
Frca	15.17	17.16	3.97	282	299	338.6	126.3	246.3	339.0	104.6	159.2	872.8	484.2	408.8	294.7	19.18	6.63
Frdi	3.37	16.36	4.12	228	317	310.6	148.0	203.3	410.0	82.6	194.8	760.4	452.9	270.9	191.6	15.84	4.45

Table 4.6 Continue

	Temperature			Altitude		Conductivity		TP		P-PO ₄		TN		N-NO ₃		CaCO ₃	
	°C			m		µS cm ⁻¹		µg L ⁻¹		µg L ⁻¹		µg L ⁻¹		µg L ⁻¹		mg L ⁻¹	
Code	N2	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut
Frul	18.99	18.12	4.65	311	329	504.9	314.5	277.8	340.1	116.8	156.0	1182.8	959.2	585.5	557.8	21.28	6.96
Goan	2.69	14.80	1.32	51	42	349.9	29.6	88.0	165.4	44.2	86.2	923.6	372.9	410.8	383.3	19.36	6.26
Gocl	3.54	14.65	2.23	117	91	297.4	59.5	658.8	199.9	266.2	72.2	868.9	69.5	537.8	324.3	20.11	7.54
Gomi	2.40	14.60	4.41	367	283	504.6	247.4	402.4	528.8	193.8	248.1	1110.1	1137.8	672.6	956.5	21.09	7.57
Gopa	13.07	18.40	4.26	253	267	459.1	237.0	279.5	346.7	124.5	162.7	876.8	880.6	390.5	433.7	19.87	6.41
Gops	1.46	14.13	9.05	1004	343	242.1	272.2	75.9	21.2	10.0	120.0	567.7	350.9	236.4	120.2	21.97	7.07
Gotr	4.95	18.84	3.39	195	266	415.7	225.3	439.1	510.1	202.7	235.0	760.6	519.6	346.5	308.2	20.88	5.51
Gyat	1.98	24.00	3.83	396	485	986.9	323.1	217.8	27.6	100.9	1.4	1846.8	1216.2	1129.6	1103.1	22.28	3.54
Lyli	2.83	14.52	1.38	564	614	475.2	227.4	141.1	278.5	45.1	114.1	872.2	291.2	238.5	203.2	19.01	5.67
Lyva	1.92	15.77	2.58	345	364	183.3	173.9	61.0	20.8	17.1	14.4	1209.0	200.5	364.4	33.1	11.96	4.27
Nabr	2.00	25.47	2.12	42	47	702.3	158.4	681.0	664.0	297.9	294.1	1277.2	198.0	536.2	615.2	27.45	3.54
Naca	4.04	26.79	7.57	308	277	461.0	216.5	100.3	90.8	46.7	49.0	757.0	836.7	391.9	553.3	21.12	12.08
Naci	2.15	18.21	2.03	149	256	320.4	24.8	592.4	672.9	269.8	315.8	644.6	897.7	323.2	405.4	16.40	7.12
Nacl	2.61	14.97	0.30	139	82	294.5	35.7	262.0	314.5	99.0	132.1	792.4	158.4	566.8	367.9	15.00	6.76
Nacr	3.79	22.06	4.20	574	190	804.4	446.6	353.9	397.8	170.0	187.9	2159.0	637.7	1377.7	748.7	22.06	11.48
Nacri	2.09	24.65	0.90	81	261	554.5	86.2	183.4	223.0	47.9	99.0	754.3	507.6	283.9	377.5	29.39	2.98
Nala	7.72	17.94	4.73	452	304	336.2	147.2	270.1	310.0	107.4	146.2	658.3	549.4	394.6	313.9	21.06	6.90
Naop	6.61	20.83	4.34	400	276	623.6	370.1	323.5	337.9	151.4	160.7	1573.9	784.4	946.9	762.5	20.05	7.44
Naph	2.82	20.36	4.06	384	355	656.6	471.0	372.3	179.3	180.2	94.7	1870.6	700.5	1426.4	362.3	25.02	8.53
Nara	2.12	22.37	4.42	153	168	342.3	40.5	68.1	63.9	25.2	22.5	369.1	205.2	128.9	77.9	20.00	6.76
Nasl	2.04	19.29	5.26	176	130	318.4	50.7	53.0	51.5	16.1	45.2	306.5	108.3	103.3	76.7	20.00	6.76
Natr	2.83	18.32	3.34	381	298	476.6	470.8	239.0	199.0	101.9	96.3	2179.7	1570.8	1019.5	276.7	17.86	11.45
Nidi	1.64	22.43	4.45	563	406	986.5	585.9	247.2	197.9	124.1	103.4	2282.8	1061.1	1644.1	675.2	20.43	5.04
Nifl	1.31	15.27	5.16	72	470	378.0	6.4	40.6	14.2	34.3	19.5	988.5	499.5	396.9	189.2	23.67	6.95
Nire	2.93	19.32	4.68	240	246	317.2	58.7	135.8	113.2	60.5	47.1	1216.0	95.6	399.3	344.6	11.00	6.76
Nium	1.69	21.80	6.56	704	122	1045.8	427.5	397.3	653.6	191.5	306.3	2568.1	313.2	1812.1	253.2	20.21	7.62
Suan	1.22	13.72	3.61	547	357	486.5	114.6	1146.5	855.6	539.8	417.2	2381.1	893.1	1803.9	735.4	16.02	7.07

Table 4.6 Continue

	Temperature			Altitude		Conductivity		TP		P-PO ₄		TN		N-NO ₃		CaCO ₃	
	°C			m		μS cm ⁻¹		μg L ⁻¹		μg L ⁻¹		μg L ⁻¹		μg L ⁻¹		mg L ⁻¹	
Code	N2	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut	uk	ut
Subr	1.46	14.12	3.32	507	334	473.7	106.1	1117.6	551.5	531.6	247.5	2310.7	721.2	1790.8	431.3	16.95	7.07
Sumi	1.67	21.40	2.55	943	13	907.3	259.5	169.8	77.1	76.4	60.1	2069.6	817.4	1254.7	268.7	36.39	3.54
Susu	2.15	17.54	1.66	295	283	212.2	163.4	65.3	8.1	11.0	4.9	1207.8	140.4	575.9	338.5	16.66	10.06
	RMSE	2.86		258.45		168.19		190		87.76		670		290		5.72	
	R ²	0.66		0.36		0.48		0.62		0.63		0.30		0.60		0.45	

4.4 Discussion

4.4.1 Physicochemical Parameters

The assessment of water quality by using physical and chemical variables of ecosystems has a long history. Physicochemical parameters as well as the hydro-morphological characteristics such as Temperature (T), Conductivity, Dissolved oxygen (DO), Total Suspended solids (TSS), Biological oxygen demand (BOD₅), Chemical oxygen demand (COD), Total organic carbon (TOC), Ammonium nitrogen (N-NH₄), Nitrate nitrogen (N-NO₃), Nitrite nitrogen (N-NO₂), Total N-nitrogen, orthophosphorus (P-PO₄), Salinity, Alkalinity, Transparency, Total phosphorus (TP), Total nitrogen (TN) and Chlorophyll-A concentration were used to support our biological elements. The changes in physical characteristics and chemical elements of water provide valuable information on the quality of the water, the source(s) of the variations and their impacts on the functions and biodiversity of the reservoir (Mustapha, 2008). Phosphorus is thought to be the most important nutrient responsible for eutrophication in these rivers (Litke, 1999; Potapova et al., 2004). Nutrient concentrations were relatively higher in Dalaman Stream (A8) than those of the other watercourses during the study period. In fact, Dalaman Stream (A8) was under pressures of human activities from Acipayam District, several villages, and farming process. The water quality of this station could be degraded by the excessive addition of inorganic and organic matters generated municipal wastewater input, agricultural activities, livestock and nutrients enriched runoff. High population and the variety of industrial and agricultural activities could expose most watersheds near to big cities to adverse and increasing environmental effects, principally to the pollutants generated by domestic and industrial sewages (Mancini and Arcà, 2000; Salomoni et al., 2006; Toudjani et al., 2017). In accordance with our results, Bellinger et al. (2006) reported that the combination of agricultural practices, known as major sources of inorganic and organic matters, livestock, organic pollutants coming from anthropogenic waste, washing of clothes, dishes and people with soaps and detergents has generated an important nutrient enrichment in the streams draining deforested watersheds in Tanzania. Phosphorus and nitrogen are the primary nutrients that in excessive amounts pollute our lakes, streams, and wetlands (MPCA, 2008). Phosphorus and nitrogen are key elements to life on Earth. Phosphorus is the second most abundant element in human bodies, found mostly in bones and teeth.

Phosphorus also is found in foods such as soft drinks and baking soda, fertilizers and cleaning products. Nitrogen makes up about 78 percent of the air is a major component of foods and fertilizers (OEPA, 2014).

4.4.2 Diatom richness and diversity

The diatoms taxa diversity and abundance are controlled by environmental parameters such as nutrients, temperature, light intensity, grazing pressure, substrate stability and discharge (Izagirre and Elozegi, 2005; Toman et al., 2014). A total of 102 taxa belonging to 22 genera were identified in the studied area (25 stations). The of species ranged between from 28 taxa to 20 taxa in the stations. The total species number in the present study is lower than those of a review study including the most common diatom taxa (129) from rivers, lakes, and reservoirs in Turkey (Solak et al., 2012b) and those of a limnologic study in Felent creek with 117 diatom taxa (Solak et al., 2012a). The commonly found species in the present study (e.g., *C. placentula*, *C. excisa*, *D. vulgaris*, *U. ulna*, *G. parvulum*, and *N. cryptocephala*) were listed in the previous studies in others running water bodies (Rott et al., 1999; Potapova and Charles, 2003; Dell’Uomo, 2004; Kelly et al., 2008; Della Bella et al., 2012; O’Driscoll et al., 2012; Solak et al., 2012a, b; Beltrami et al., 2012; Solak et al., 2012a; Çelekli and Öztürk, 2014; Delgado and Pardo, 2014; Kelly et al., 2014).

The species number per site found during this study was higher compared to some early literatures (e.g. Bellinger et al., 2006; O’Driscoll et al., 2012). This situation was confirmed by the highest nutrient concentrations especially orthophosphate ($P-PO_4$) and nitrate nitrogen ($N-NO_3$) found in our studied sites compared to those obtained in these previous studies. In contrary, our species number and species number per site were respectively lower than the result obtained by Tokatlı and Dayioğlu (2011) in Murat stream in which there was a gradual pollution and by Della Bella et al. (2012) in Italy. From that point, it can be seen that nutrients, especially nitrogen and phosphate play important role in algal species growing and richness in an aquatic system. Species richness is generally expected to be greater in not impacted ecosystems, but it has been reported that low stressors could be the source of increasing in diversity (Jüttner et al., 1996; Lowe and Pan, 1996; Jüttner et al., 2003) or may not show any changes despite high anthropogenic stressors (Jüttner et al., 1996; Bellinger et al., 2006). The increase of nutrient contents results in higher

species number up to a median level, where high nutrient contents are shown as a limiting factor, which leads in decreased species number (Manyolov and Stevenson, 2006; Fidlerová and Hlúbíková, 2016). The differences in species diversity represent also the general higher heterogeneity of physicochemical, hydromorphological and geographical conditions in reservoirs having several purposes in comparison with much more homogeneous conditions in drinking water-supply reservoirs with lower levels of anthropogenic influences and the consequent pollution (Fidlerová and Hlúbíková, 2016). It has previously been reported that human activities could influence diatom assemblages and water quality state (e.g. Van Dam et al., 1994; Reynolds et al., 2002; Àcs et al., 2004; Bellinger et al., 2006; Hlúbíková et al., 2007; Padisák et al., 2009; Delgado and Pardo, 2014). Soininen (2007) reported that total phosphorus, nitrates, pH, conductivity, temperature, alkalinity, biological oxygen demand (BOD₅), calcium, chlorophyll a, altitude, and substrate type are major environmental factors for diatom distribution in streams. According to Beltrami et al. (2012), altitude and geology characteristics were the principle environmental variables influencing diatom assemblages in alpine streams. Biggs and Close (1989) indicated that stressors such as floods decrease the impact of grazing pressure because recolonization of macroinvertebrates take generally more time than the phytobenthic organisms growth while, Lange et al. (2011) demonstrated that light availability, nutrient contents, and grazing pressure characterize the stream diatom diversity. *Achnanthes*, *Amphora*, *Aulocoseria*, *Cocconeis*, *Craticula*, *Cyclotella*, *Cymbella*, *Cymatopleura*, *Melosira*, *Hantzschia*, *Gyrosigma*, *Gomphonema*, *Fragilaria*, *Eunotia*, *Epithemia*, *Diploneis*, *Diatoma*, *Denticula*, *Navicula*, *Surirella*, *Stauroneis*, and *Nitzschia* were recorded in our studied station. Most species of *Nitzschia*, *Gomphonema*, *Fragilaria*, *Navicula*, *Melosira* and *Surirella* genera belong to codon T_B which inhabited highly lotic (streams and rivulets) environments (Reynolds et al. 2002; Padisák et al., 2009). In accordance with our finding, most the diatom genera found during our study were listed by Della Bella et al. (2012), in central Italy streams and in Murat stream (Turkey) by Tokatlı and Dayıoğlu (2011) and in tropical streams (Tanzania) by Bellinger et al. (2006). *F. capucina*, *C. excisa*, *G. parvulum*, *U. ulna* and *C. placentula* were the most dominant species. These species were mostly found in sites with high to moderate ecological state and were early listed in others running water (Potapova and Charles, 2003; Tokatlı and Dayıoğlu, 2011; Della Bella et al., 2012; O'Driscoll et al., 2012). *Cymbella excisa*

has been indicated as dominant taxa in I-II. water quality (less polluted) by Gómez and Licursi (2001). Furthermore, many species of the genus *Cymbella* are indicators of the oxygen-rich waters (Van Dam et al., 1994). According to Krammer and Lange-Bertalot (1991a), *U. ulna* has been adapted to several ecological states.

4.4.3 Structuring factors of diatom community composition

Relationships between diatoms and environmental parameters can be quantified to make diatoms suitable quantitative bioindicators of ecological states in ecosystems (Pan and Stevenson, 1996; Oliveira et al., 2001). Diatom species react quickly to environmental variations due to their short life-time, rapid dispersal and a large number of taxa with various tolerance range to physicochemical variables (Lotter et al., 1999). The species number and proportion of diatoms are determined by environmental variables such as nutrients, temperature, light intensity, grazing pressure, support stability and discharge (Izagirre and Elozegi, 2005). Phosphorus is thought to be the most important nutrient responsible for eutrophication in these rivers (Litke, 1999; Potapova et al., 2004).

In order to understand the interactions between the diatom species and environmental factors in the watercourses, direct gradient analysis technique, canonical correspondence analysis (CCA), was performed. The Monte Carlo permutation test result demonstrated that the distribution of diatom composition was influenced by physicochemical variables and the first axes on the CCA ordination gave the species-environment correlations at 89.4% and Cumulative percentage variance of species data at 9.4 %. Nutrients such as TP, TN, P-PO₄, N-NO₂, and N-NO₃ were the most explanatory factors. This confirmed our hypothesis (II) that environmental variables especially pollution parameters (nutrients, organic matter) are the most important structuring factors of phytoplankton composition and validated that diatoms and previously used to assess nutrient enrichment (e.g. Kelly et al., 1995; Rott et al., 1997; 1999; Coring, 1999; Dell’Uomo, 2004). Abonyi et al. (2012) indicated a relationship between benthic diatoms and nutrient which reflected human impacts along the River Loire (France). According to Strebel et al. (1989) and Almasri and Kaluarachchi (2004), the downstream increase of nutrient is a generally anthropogenic impact by agriculture and could be a useful indicator of eutrophication in large rivers (Turner et al., 2003). The result of CCA (Figure 4.10) supported by

weighted average (WA) regression (Table 4.6) showed that *N. umbonata*, *C. meneghiniana*, *D. moniliformis*, *N. phylepta*, *N. cryptocephala* and *N. angustata* were associated with high concentrations of N-NO₃. *Cyclotella meneghiniana* was reported as pollution tolerant taxa by Venkatachalapathy and Karthikeyan (2013) and as taxa typical of polluted environments by Sladěček (1973), Kobayasi and Mayama (1989), Van Dam et al. (1994), Lobo et al. (1996, 2002) and Salomoni et al. (2006). Contrary species of the genus *Fragilaria*, especially *U. biceps*, *F. capucina*, and *F. incognita* were indicated with low nitrogen content especially TN, NO₃, and NO₂. Reynolds (2006) reported that *Fragilaria* did not have an unusually high demand for phosphorus or nitrogen. On the other hand, *N. lanceolata*, *S. minuta*, and *G. attenuatum* preferred high concentrations of TP and P-PO₄. According to Hlúbíková et al. (2007), a natural increase in nutrient concentration and organic compounds could lead to the appearance of pollution-tolerant species like *N. lanceolata*. *N. lanceolata* has been also indicated as tolerant to organic/eutrophic conditions (Van Dam et al., 1994; Delgado and Pardo, 2014). In accordance of our finding, *N. cryptocephala* has been previously noted to be nutrient tolerant taxa by Àcs et al. (2004) and tolerant to organic pollution and eutrophication by Salomoni et al. (2006). *N. angustata* and *N. umbonata* belong to codon D which habitat template is shallow nutrients enriched and low transparent watercourses including running waters (Reynolds et al., 2002; Padisák et al., 2009).

The trophic preferences of diatom species established based on TP classes. *Clevamphora ovalis*, *C. placentula*, *D. vulgaris* and *U. ulna* indicated a preferential occurrence polytrophic conditions, while species such as *F. capucina*, *G. angustum* and *G. truncatum* were distributed in eutrophic and polytrophic conditions. *U. ulna* has been reported to be highly pollution tolerant species (Round, 1991; Biggs and Kilroy, 2000; Potapova and Charles, 2003; Duong et al., 2006; Bere and Tundisi, 2011) and more tolerant to heavy organic pollution and eutrophication (Salomoni et al., 2006). *F. capucina* has been cited as an indicator of trophic state species (Wang et al., 2014) and low pollution tolerant species (Bere and Tundisi, 2011). In contrast to our findings, *C. excisa* was indicated as a low trophic indicator species (Wang et al., 2014), as a pollution sensitive diatom taxon (Delgado et al. 2012) and has been considered to be dominant in I-II. water quality (less polluted) (Gómez and Licursi, 2001).

CHAPTER V

BIOASSESSMENT OF RUNNING WATER BASED PHYTOBENTHOS (EPILITHIC DIATOMS)

ABSTRACT

Water quality in aquatic ecosystems is an important issue because it maintains the ecological processes that support biodiversity. Diatoms in lotic ecosystems are an important group of phytobenthos and are good indicators of water quality especially in rivers and streams. This chapter aimed to evaluate 25 watercourses ecological status in the western Mediterranean basin of Turkey by the use of the trophic index Turkey (TIT) based on diatom combined to trophic index (TI) and eutrophication and pollution index diatom-based (EPI-D). Data were sampled during four seasons in the Western Mediterranean basin for biological and physicochemical analysis.

The Ecological status of the watercourses was evaluated following the ecological quality ratio based on TIT (EQR-TIT) and EIP-D. TIT values ranged between 1.53 and 2.8, while the highest and lowest EPI-D values were 1.01 and 1.88 respectively. With regard to ecological status, stations indicated poor, moderate, good or high according to the indices. A significant positive correlation between TIT, TI and EIP-D values and logTP proved that these diatom metric are a useful tool for the assessment of running water quality in a Western Meterranean basin approach.

5.1 Introduction

Water is a precious and essential natural resource for multiple uses (domestic, industrial and agricultural) (Sadat et al., 2011). Surface freshwaters are the most essential sources of water for human activities, but are unfortunately under several environmental disturbances and are being threatened as result of economical activities. Water quality in aquatic ecosystems is an important issue because it maintains the ecological processes that support biodiversity. Degradation of water quality in surface waters has increased worldwide because of the release of wastes resulting from urbanization, demographic change and industrial hazardous effluent into receiving waters (Schindler, 2006; Smol, 2008). Apart from that, climate change resulting from environmental disturbances causes a decrease in water quality and threatens the organization and functions of aquatic ecosystems. Water pollution, defined as physical, chemical, or biological degradation caused by human activity, disrupts living conditions and aquatic balances compromising thereby their multiple uses. However, the quality of surface water is influenced by both natural processes (soil erosion, precipitation, and evaporation) and human activity (agriculture, urban, and industrial wastewater) (Strobl & Robillard, 2008).

To deal with these problems and to provide solutions, the water framework directive (WFD) required member states of the Union to regularly assess running water quality by the use of so-called biological quality elements, among these are phytobenthos (especially benthic diatoms) (Directive, 2000; EC, 2009, Rott and Schneider, 2014). For each of these biological quality elements, the taxonomic composition and proportion of the species should be determined, and five categories of ecological status (high, good, moderate, poor and bad) should be established following normative definitions in the Directive (Directive, 2000; Schaumburg et al., 2004). The environmental aims of the WFD insist all surface watercourses to be “good ecological status” (defined as having a biota consistent with only slight alterations from that expected in the absence of human impacts) by 2015. Each Member State (MS) has had to define national systems for assessing ecological status for a range of biological quality elements, as defined in Annex V of the WFD (Kelly et al., 2009). In order to ensure comparability of such monitoring systems, the results of the systems operated by each member state shall be expressed as ecological quality ratios for the purposes of classification of ecological status. These ratios shall

represent the relationship between the values of the biological parameters observed for a given watercourse and the scores for these parameters in the pristine conditions applicable to that watercourses.

Benthic diatoms in rivers and streams are a great part of phytobenthos and aquatic biodiversity. Due to this fact, they are widely used worldwide in freshwaters quality monitoring especially rivers and streams, and their based indices have been developed for the evaluation of water quality (e.g., Rott et al., 1999, 2003; Lobo et al., 2004; Potapova et al., 2004; Herring et al., 2006; Kelly et al., 2009; Rimet, 2012; Wang et al., 2014; Rimet et al., 2016). The advantage in benthic diatoms is that they can be found in every surface water at any time (Ács et al., 2004; Kiss et al., 2012). Diatom-based metrics usually indicate a highly significant and well response to nutrient gradients (Rott et al., 2003; Kelly et al., 2014) with higher correlation coefficients than those of other biological elements (Birk et al., 2012).

The aims of the chapter were to assess the ecological status in running watercourses located in the western Mediterranean basin of Turkey using trophic index Turkey (TIT), trophic index (TI) and Eutrophication and pollution index diatom-based (EPI-D).

5.2 Material and Methods

The water quality of watercourses was evaluated following three (3) diatom indices: Trophic index Turkey (TIT), Eutrophication and Pollution Index Diatom-based (EPI-D) and the trophic index (TI). TIT was calculated using the equation (Eq.1) proposed by Çelekli et al. (2016, 2018a). TIT values were then transformed as ecological quality ratios (EQRs) to evaluate the ecological status and the water quality of the studied watercourses according to the WFD (Eq.2). As suggested by the European Union waterframe work directive, EQR scores varied from zero (0) which indicates poor water ecological status to one (1) indicating high water ecological status. EPI-D was calculated according to the equation (Eq.3) proposed by Dell’Uomo (2004). TI score was determined based on the formula (Eq.4) developed by Rott et al. (1999) for Austrian rivers.

$$TIT = \frac{\sum_{i=1}^n b_i * e_i * c_i}{\sum_{i=1}^n e_i * c_i} \quad (\text{Eq. 1})$$

where b_i is the trophic number of species i (1-5), e_i the indicative weight number of species i (0-5) and c_i abundance of species i (%). Increasing water quality corresponds to TIT values from 4 down to 0.

$$EQR = \frac{(4 - TIT_o)}{(4 - TIT_{ref})} \quad (\text{Eq. 2})$$

where, TIT_o and TIT_{ref} are the observed and reference TIT, respectively. The ecological status boundary classes were determined according to the WFD, using the box-plot analysis for each typology.

$$EPI - D = \frac{\sum_{j=1}^n a_j * r_j * i_j}{\sum_{j=1}^n a_j * r_j} \quad (\text{Eq. 3})$$

where, EPI-D = Eutrophication/pollution index of the stations; a_j = abundance of the species j ; r_j = reliability of the species j , inversely proportional to its ecological

"range"; values used: 5 for an optimum indicator, 3 for a good indicator, 1 for a sufficient indicator only; i_j = weighted integrated sensitivity index of the species j; the values attributed go from 0 (for environment of excellent quality) to 4 (degraded water body) (Table 5.1).

Table 5.1 The ecological status levels based on EPI-D (Dell’Uomo, 2004)

EPI-D values	Water quality
0.0<1.0	High
1.0<1.7	Good
1.7<2.3	Moderate
2.3<3.0	Poor
3.0<4.0	Bad

$$TI = \frac{\sum_{i=1}^n TW_i * Gi * Hi}{\sum_{i=1}^n Gi * Hi} \quad (\text{Eq. 4})$$

Where TI is the trophic index, TW_i is trophic number of species i, G_i is the indicative weight number of species i (0 to 5), H_i is the abundance of species i (%). The ecological status levels according to this index, are presented in Tables 5.2.

Table 5.2 The ecological status levels of the Trophic Index (TI) (Rott et al., 1999) modified by Çelekli et al. (2018b)

Water quality	TI values
High	<1.4
Good	1.4-<2.0
Moderate	2.0-<2.7
Poor	2.7-3.3
Bad	>3.3

5.3 Results

The studied sites' water quality was assessed based their ecological status using the trophic index Turkey (TIT) combined to the eutrophication and pollution index based on diatom (EPI-D). Values of TIT, EPI-D, TI and EQR based on TIT and its related ecological status are presented in Table 5.3. The highest (2.73) and lowest (1.53) TIT values were recorded respectively in Dalaman stream (A8) and Kaya creek, while EQR based on TIT changed from a minimum of 0.44 at Dalaman stream (A8) to a maximum of 0.99 Kaya and creeks. On the other hand, the highest (1.88) and lowest (1.01) EPI-D values were recorded respectively in Dalaman stream (A9) and Kaya creek, while the highest (2.89) and lowest (1.43) TI values were recorded respectively Sarıçay creek (A2) and Kızıllöz creek (A13).

Ecological status of the sampled water bodies ranged from high to poor according to ecological quality ratios based on TIT. The good ecological status was the most recorded during the study followed by high ecological status while a poor water status was found only at the stations A8 of Dalaman stream (Table 5.3). According to EPI-D, the water quality all the stations indicated a good ecological status, except Dalaman stream (A9) which showed a moderate status during the study (Table 5.3). The water quality of the sampled stations based on the trophic index (TI) varied from poor to good status. Sarıçay creek (A2) was the only station indicating a poor ecological status and most of the stations showed a moderate water quality (Table 5.3).

In order to compare the suitability and applicability of the TIT, EPI-D and TI indices in the watercourses of the western Mediterranean basin, TIT and EPI-D were plotted against logTP (Figure 5.1). Based on R^2 scores, a higher part of variance of TIT was explained by logTP than it done with EPI-D and TI (Figure 5.1a,b and c). Total phosphorous (TP) also indicated a significant positive correlation with the trophic index (TIT). Besides, the EQR_{TIT} scores showed a significant negative correlation with logTP ($R^2 = 0.67$, $p < 0.01$) (Figure 5.1d).

The test of no difference between the two indices was carried out using a variance analysis (one-way ANOVA). The result of this analysis indicated no significant difference between the lakes and reservoir. (one-way ANOVA; $P > 0.05$; $F = 0.76$) in related to their ecological status.

Table 5.3 TIT, EQR, EPI-D and TI values and ecological status of the stations

Stations	Code	TIT	EQR	Status	TI	Status	EPI-D	Status
Sarıçay creek	A2	2.18	0.64	Moderate	2.89	Poor	1.28	Good
Namnam creek	A3	2.34	0.76	Good	2.34	Moderate	1.19	Good
Kargıcak creek	A4	2.11	0.61	Moderate	1.87	Good	1.40	Good
Dalaman stream	A5	2.33	0.63	Moderate	2.17	Moderate	1.29	Good
Dalaman stream	A6	2.28	0.77	Good	2.16	Moderate	1.18	Good
Dalaman stream	A7	2.48	0.71	Moderate	1.53	Good	1.39	Good
Dalaman stream	A8	2.73	0.44	Poor	2.57	Moderate	1.37	Good
Dalaman stream	A9	2.66	0.61	Moderate	2.64	Moderate	1.88	Moderate
Seki stream	A10	2.50	0.84	Good	2.42	Moderate	1.22	Good
Seki stream	A11	2.36	0.65	Moderate	2.31	Moderate	1.25	Good
Çayıçi creek	A12	2.21	0.80	Good	1.96	Good	1.29	Good
Kızılöz creek	A13	2.13	0.64	Moderate	1.43	Good	1.19	Good
Kızılöz creek	A14	2.28	0.69	Moderate	1.81	Good	1.15	Good
Çavdır stream	A15	2.62	0.61	Moderate	2.20	Moderate	1.35	Good
Boğluca stream	A16	2.56	0.83	Good	2.62	Moderate	1.33	Good
Akçay stream	A17	2.40	0.80	Good	2.44	Moderate	1.55	Good
Alakır stream	A18	2.13	0.83	Good	1.64	Good	1.36	Good
Eşen stream	A19	2.30	0.78	Good	1.68	Good	1.10	Good
Kanlı stream	A20	1.85	0.93	High	1.80	Good	1.32	Good
Eşen stream	A21	2.36	0.77	Good	2.13	Moderate	1.50	Good
Karabeyyurdu creek	BAK 12	1.98	0.96	High	2.19	Moderate	1.07	Good
Delin creek	BAK14	1.88	0.98	High	1.81	Good	1.03	Good
Kaya creek	BAK4	1.53	0.99	High	1.51	Good	1.01	Good
Kocabük creek	BAK9	1.89	0.99	High	2.14	Moderate	1.28	Good
R1creek	R1	1.84	0.91	High	1.84	Good	1.18	Good

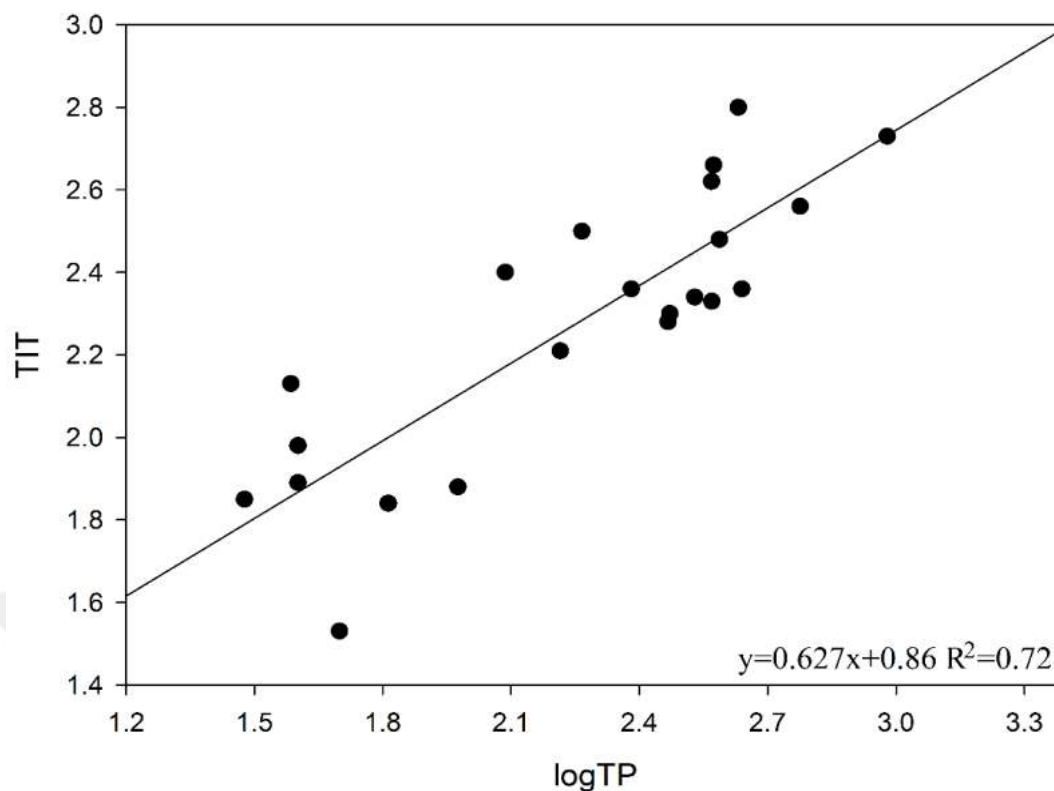


Figure 5.1a Plot of relationship between TP and trophic index Turkey (TIT)

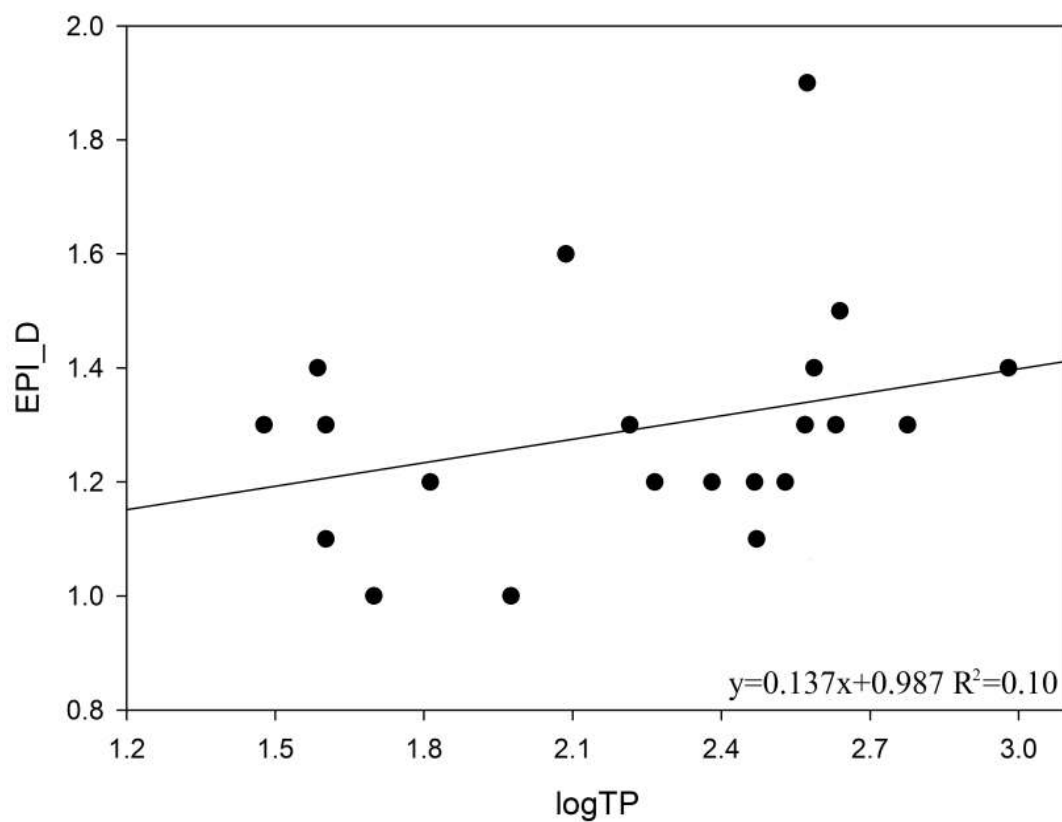


Figure 5.1b Plot of relationship between TP and eutrophic an pollution index (EPI-D)

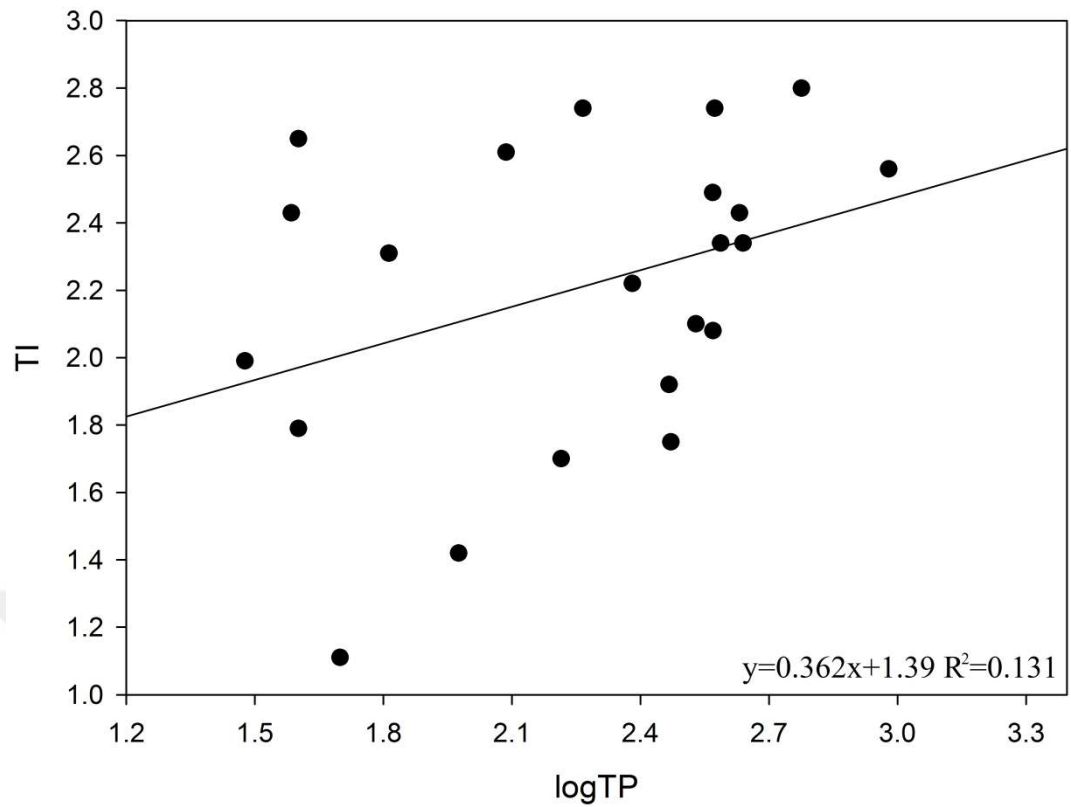


Figure 5.1c Plot of relationship between TP and trophic index (TI)

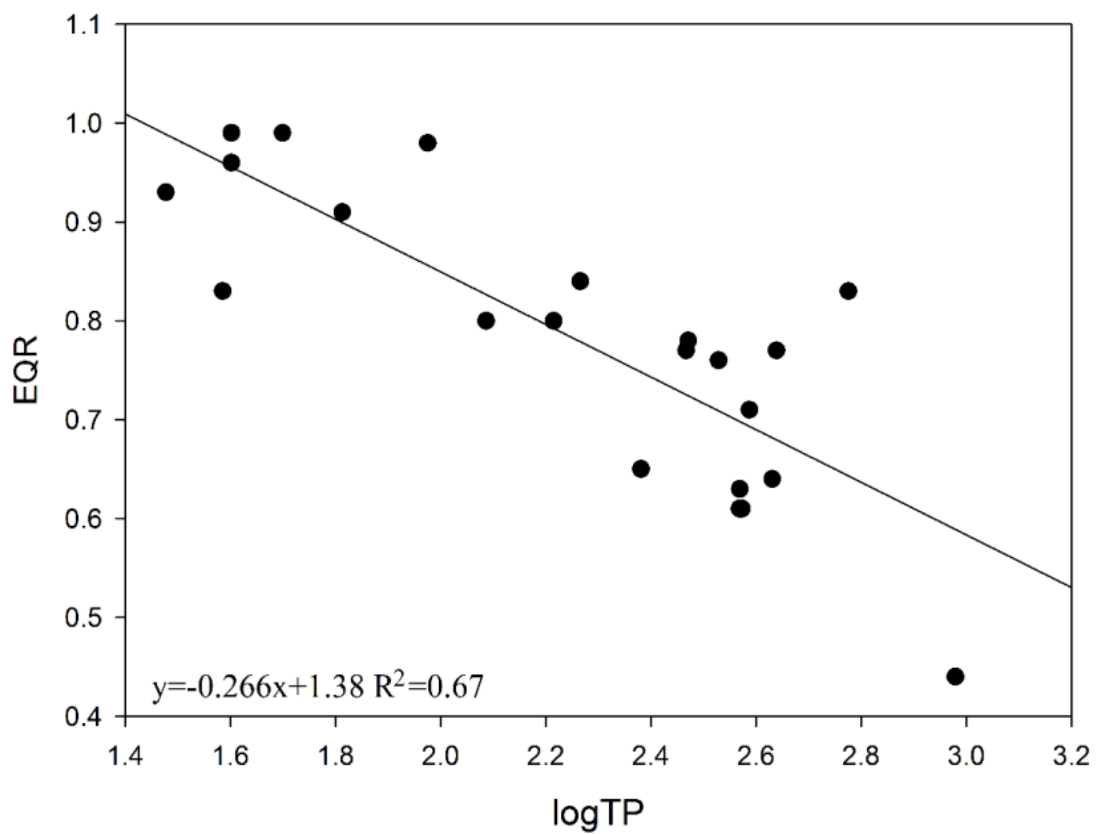


Figure 5.1d Plot of relationship between TP and and EQR-TIT

5.4 Discussion

As basin concept, we used during this study for the first time three diatom indices to evaluate running water including 25 stations in the western Mediterranean of Turkey. Using biological quality elements for Water quality assessment is a new topic in Turkey, but is getting more importance. As suggested by the UE WFD, many studies have been carried out recently in the different regions of Turkey.

Biological monitoring provides a direct measure of ecological integrity by using the species responses to the changes in environmental variables (Karr, 1991; Angermeier and Karr, 1994; EC, 2009). Among biological quality elements, diatom assemblages often considered as important ecological indicators of water bodies due to their rapid and sensitive response to environmental changes (Kelly, 1998; Potapova et al., 2004; Bona et al., 2007; Delgado and Pardo, 2014). For the assessment of water quality conditions, benthic diatoms have been used since the 1900s (Kolkwitz and Marsson, 1908). In Europe, benthic diatom metrics such as IPS (Cemagref, 1982), TI (Rott et al., 1999), EPI-D (Dell’Uomo, 2004) and TDI (Kelly et al., 2008) developed for running waters are successfully applied for water quality assessment worldwide. But, the environmental optima and tolerance ranges of diatom species may vary from one ecoregion to another, requiring local indices to be developed (Gomá et al., 2004; Lobo et al., 2004). Several authors (e.g. Kelly et al., 1998; Pipp, 2002; Rott et al., 2003; Dell’Uomo, 2004) indicated that diatom indices/metrics developed in certain parts of Europe are not effective when used in other areas of the same continent.

In related to the ecological status following TIT, a high ecological status was observed in Dalaman stream (R1), Karabey yurdu creek, Delin creek, Boğluca stream, Kocabük Creek, Koca stream and a good ecological status was observed in Alakır stream, Boğluca stream, Namnam Creek, Eşen stream, Dalaman stream (A6), Seki stream, Çayıçi Creek and Ak stream. Aforementioned stations had also low nutrient contents and non degraded hydromorphological characteristics representing pristine conditions watercourse type as recommended by the WFD (Directive, 2000). In according with our finding, high ecological status had been previously reported from different running water in many countries using various diatom-based indices (e.g. Beltrami et al., 2012; O’Driscoll et al., 2012). These watercourses as well as our sampled water bodies were located at similar altitudes (<400 m) and showed similar

geological characteristics. The high water quality of aforementioned aquatic systems could be the consequence of the moderate land use for agriculture which generates inorganic and organic nutrients and the low free to roam through the water bodies of organic pollutants generated from human activities. On the other hand, a poor status was observed at Dalaman stream (A8) which showed the highest nutrient (TN, TP; N-NO₃, N-NO₂, and P-PO₄) contents and characterized by the occurrence of pollution tolerant or eutrophication indicator species such as *N. umbonata*, *N. cryptocephala*, and *C. meneghiniana* during this study. As indicated before, Dalaman Stream (A8) was under pressures of human activities from Acipayam District, several villages, and farming process. The water ecological status of this station could be impacted by huge addition of inorganic and organic matters generated by municipal waste input, agricultural practices, livestock and nutrient enriched runoff. Besides, agricultural practices, the input of wastewater, and organic pollutants generated by fish farming could be the causes the moderate states observed in Sari stream, Dalaman stream (A5, A7, and A9), Seki stream (A11) and Çavdır Stream which are near to human settlements. Most of the highest nutrient (TN, TP; N-NO₃, N-NO₂, and P-PO₄) contents were found in these stations. *N. angustata*, *N. umbonata*, *N. cryptocephala*, *N. lanceolata*, and *N. dissipata* were commonly found in these stations. Besides, WA result (Table 5.7) supported by CCA (Figure 5.10) showed these species were relatively tolerant to high nutrient contents. Besides, relationships test (Figure 5.11) showed a significant positive correlation ($R^2 = 0.72$, $p < 0.01$) between TP and TIT. This situation has also been indicated early by other authors in different countries (e.g. Van Dam et al., 1994; Reynolds et al., 2002; Ács et al., 2004; Salomoni et al., 2006; Hlúbíková et al., 2007; Padisák et al., 2009; Delgado and Pardo, 2014). Salomoni et al. (2006) reported that high population densities and the variety of anthropogenic activities expose many hydrographic basins close to large urban centers to heavy and rising environmental impacts, especially to the pollution by domestic and industrial waste residues.

This situation was confirmed by (Mancini and Arcà (2000) who reported that urbanization and agriculture practices in the study area affect the basins of watercourses.

According to EPI-D, all the watercourses indicated a good ecological status, except Dalaman stream (A9) which showed a moderate status during the study. This result

is in accordance with that indicated by Della Bella et al. (2007) in wetlands of central Italy. It is also similar to those reported by Beltrami et al. (2012) found from rivers of Northern Italy. In fact, the authors indicated water quality changing mostly from high to good except three sites were considered of class III–IV or IV for EPI-D (moderate to bad quality). The water quality of the sampled stations based on the trophic index (TI) varied from poor to good status and most showed a moderate water quality. As it can be seen, TIT and EPI-D were more efficient than TI. This can be explained by the ecoregional characteristics. In fact, the two first indices were dedicated for Mediterranean rivers, while TI was developed for Alpine rivers 20 years ago.

Trophic status of the watercourses was also evaluated using TI. Two (2) mean trophic classes (Mesotrophic and eutrophic) and three (3) semi-classes (oligo-mesotrophic, meso-eutrophic and eutro-polytrophic) were observed during the study. Similar water quality was previously reported by Beltrami et al. (2012) from rivers of Alto Adige/Südtirol (Northern Italy) using the same trophic index. Anthropogenic activities, geology, and climatic structures of catchment region had an impact on the water quality in the watersheds of the western Mediterranean basin. The studied water bodies were under human pressures such as agriculture, livestock, and urban waste. According to Anonymous (2016), 25% of the basin area is agricultural land, 57% of the agricultural areas are mixed agricultural lands and 34% is arable lands. The most common crops which grow in the basin are tomatoes, peppers, citrus fruits, apples, pears, quince, wheat, corn, and olives. Aforementioned factors could explain the highest nutrient concentrations observed in the stations. Besides, the water quality could be strongly deteriorated by the more addition of inorganic and organic matters from discharging wastes mining. Nutrients and pollutants deposition along the watersheds lead to the increase of nutrient the water bodies. It is stated that direct sewage inputs, runoff from fertilized soils and land degradation through seasonal (agriculture, irrigation) activities had a adverse effect on water resources over the last centuries (Wunsan et al., 2002; Ducharne et al., 2007; Delgado and Pardo, 2014).

CHAPTER VI

LIMNOECOLOGY OF PHYTOPLANKTON IN THE LAKES AND RESERVOIRS

ABSTRACT

Freshwater quality, especially lakes and reservoirs are seriously affected by anthropogenic activities and natural factors. Due to this reason, Lakes and reservoirs should be assessed at a local, regional and national level not using only physicochemical parameters, but also biological quality elements. Phytoplankton, as one of these biological quality elements are useful tool for the bioassessment surface waters. The aim of this work was to assess the water quality of three lakes and six reservoirs in the Western Mediterranean Basin (Turkey) using phytoplankton indices. Data were sampled seasonally from summer 2014 to summer 2015. The phytoplankton composition was characterized by 206 species belonging to 15 functional groups (**B, D, E, F, G, J, L_M, L_O, M, MP, P, T, W1, X1, and Y**). Environmental variables especially pH, nutrients, total organic carbon (TOC) and conductivity had a significant impact ($p < 0.01$) on the distribution of phytoplankton composition. The canonical correspondence analysis used to investigate the relationships between phytoplankton species and physicochemical parameters indicated a 93.2% correspondence between species-environment correlations and 13.6% of cumulative percentage variance of species data.

6.1 Introduction

Freshwater quality, especially lakes and reservoirs are seriously affected by anthropogenic activities and natural factors. The types of organisms in these aquatic ecosystems are mainly determined by their waters' quality. Lotic aquatic systems are characterized by several groups of organisms that are classified by their location and by their adaptations in these environments. These organisms are categorized as plankton (free living organisms including phytoplankton and zooplankton), nekton (free-moving organisms) and benthos (attached organisms to hard substrates).

Phytoplankton are prokaryotic or eukaryotic free living photosynthetic microorganisms in aquatic systems and play a central role in the structure and functioning of aquatic ecosystems. They primary producer in food web, responsible for the total primary production in the limnetic zone and can grow rapidly and live in harsh conditions due to their unicellular or simple multicellular structure. Microalgae are present in all existing surface aquatic ecosystems, representing a big diversity of species living in a large range of environmental conditions. Phytoplankton response quickly to nutrient variations through changes in its abundance and composition (Reynolds, 1984; Rott, 1984; Naselli Flores and Barone, 1998; Reynolds et al., 2002; Schaumburg et al., 2004; Reynolds, 2006; Poikane et al., 2011; Cellamare et al., 2012).

Phytoplankton could be used as a good bioindicator of water quality due to its sensitivity and ability to response in the surrounding environment changes (Reynolds, 1984; Padisák et al., 2003; EC, 2009; Padisák et al. 2006; Poikane et al., 2011; Katsiapi, et al., 2011). Spatial and temporal distribution of phytoplankton species could be explained by complex relationships between physical, chemical, and biological variables (Margalef, 1983; Pannard et al., 2008; Padisak et al., 2010). The impacts of chemical parameters on phytoplankton taxa have been largely demonstrated, while a few research concerning the role of physical factors on phytoplankton communities were carried out (Zohary et al., 2010; López et al., 2012). Throughout its annual development, the phytoplankton changes several quite different successional steps during which equilibrial compositions occur whether for shorter or longer periods (Padisák et al., 2003), thereby they have developed morphological and physiological adaptive strategies for surviving in different

conditions (Margalef, 1978; Reynolds, 1998, 2002, 2006; Padisák et al., 2003; Becker et al., 2010). Data related to the composition, biomass, and biovolume of phytoplankton are being used for the biomonitoring of ecological status in watercourses (Marchetto et al., 2009; Çelekli and Öztürk, 2014). One of the most recent phytoplankton research interests is the establishment of the so-called phytoplankton functional groups (Abonyi et al., 2012).

Aims of this chapter were to i) estimate the phytoplankton composition, ii) determine phytoplankton functional groups in different ecosystems, and iii) evaluate the relationship between phytoplankton assemblages and environmental variables in three lakes and six reservoirs in the Western Mediterranean basin.



6.2 Material and Methods

6.2.1 Sampling Stations

The studied water bodies are located in the West Mediterranean Basin of Turkey which subdivided into five provinces including Aydın, Antalya, Burdur, Denizli and Muğla. According to the results of the Address-based census (ADNKS) of 2015 conducted by the Turkish Statistical Institute (TURKSTAT), the total population of in the Western Mediterranean Basin was 5.502.620. The surface area of the basin is 2103004,9 ha and consists of 65 sub-basins and 372 micro-basins (Anonymous, 2016). The study was conducted in all the provinces except the Aydın province. The locations of the water courses are given in Figure 6.1. Hydro-geographical features characteristics of the lakes and reservoirs are sorted in Table (6.1). Pictures of these water bodies are given in Figure 6.1.



Figure 6.1 Location of sampling stations on the map. For stations codes, see Table 6.1

Table 6.1 Hydro-geographical features of the lakes and reservoirs

Province	Lakes/Reservoir	Latitude (N)	Longitude (E)	Area (km ²)	Depth (m)	Category	Typology
Denizli	Lake Gölhisar (G1)	37° 7' 28.056"	29° 36' 24.408"	292.9	3	Alkaline	R2D1A1J2
Muğla	Çayboğazı reservoir (G3)	36° 31' 38.748"	29° 40' 47.675"	135.4	52	Calcareous	R2D1A1J2
Muğla	Lake Avlan (G4)	36° 34' 18.876"	29° 57' 12.491"	581.9	2	Alkaline	R3D1A1J1
Muğla	Geyik reservoir (G5)	37° 23' 47.904"	27° 52' 56.64"	305.7	39	Calcareous	R1D2A1J2
Denizli	Çavdır reservoir (G7)	37° 4' 25.032"	29° 44' 2.22"	180.8	44	Calcareous	R1D2A1J2
Antalya	Toptaş reservoir (G8)	36° 25' 25.14"	30° 21' 17.28"	11.9	16	Calcareous	R2D2A1J2
Burdur	Yapraklı reservoir (G9)	37° 1' 43.716"	29° 27' 6.3"	569.6	55	Calcareous	R2D2A1J1
Antalya	Osmankalfalar reservoir (G10)	37° 6' 39.492"	29° 53' 1.103"	58.7	21	Calcareous	R2D2A1J1
Burdur	Lake Yazır (G11)	37° 0' 9.852"	29° 44' 3.599"	227.8	1	Alkaline	R3D2A1J1

Typology codes are R, altitude; D, depth; A, surface area; and J, geology

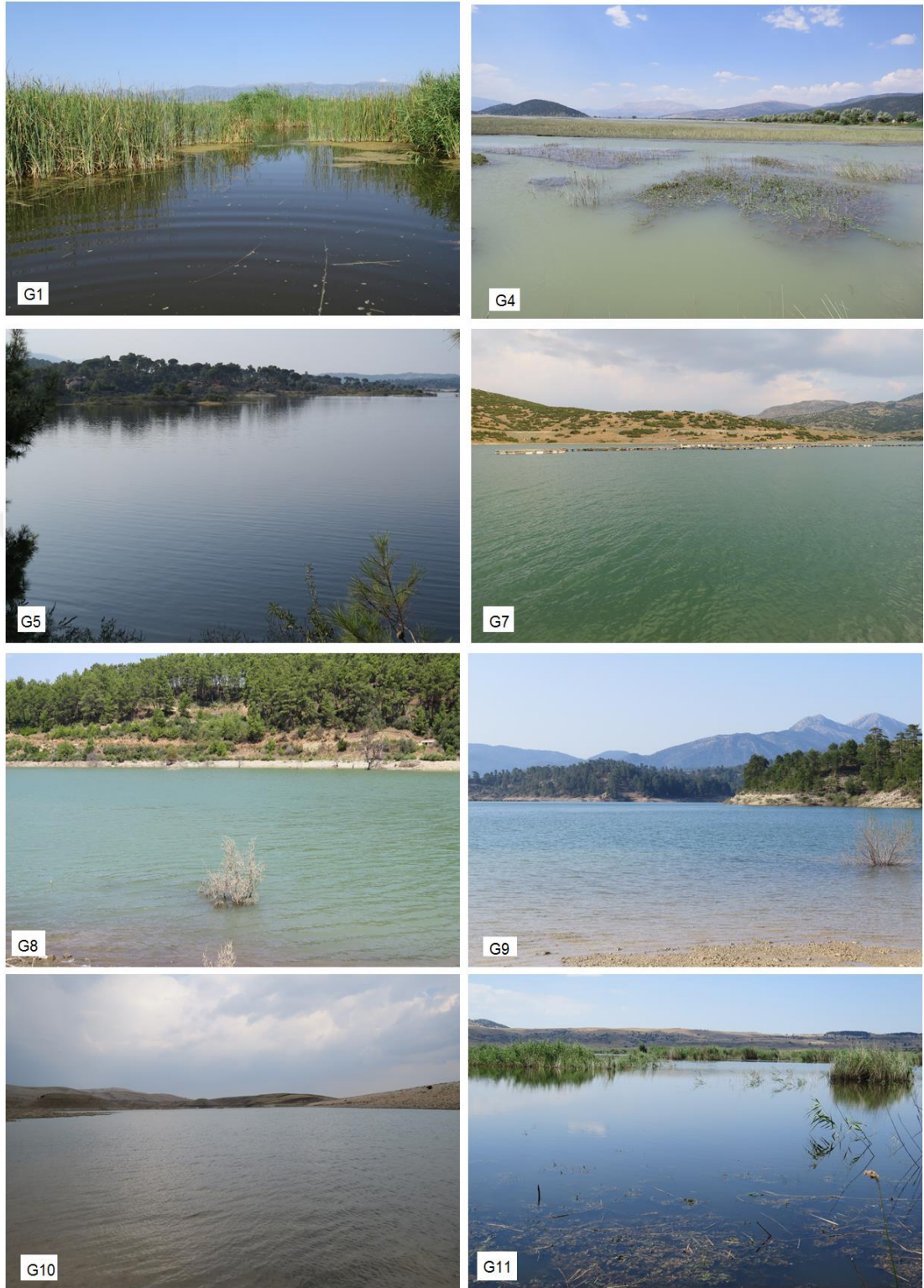


Figure 6.2 Photographes of sampling stations. G1, Lake Gölhisar; G4, Lake Avlan; G5, Geyik reservoir; G7, Çavdır reservoir; G8, Toptaş reservoir; G9, Yapraklı reservoir; G10, Osmankalfalar reservoir; G11, Lake Yazır

6.2.2 Field Sampling

6.2.2.1 Phytoplankton Sampling

Phytoplankton and water samples were collected from the lakes and reservoirs during four (4) seasons (summer 2014, fall 2014, spring 2015 and summer 2015). Hydrobios plankton net (55 μm mesh size) was used to collect net-plankton for phytoplankton identification (Figure 6.3a). Water samples (250 ml) were directly taken from just beneath of the surface water from each lake and reservoir for phytoplankton enumeration. Collected phytoplankton samples (Figure 6.3b) were fixed with lugol-glycerol's solution according to standard methods (CEN, 2006, 20012) and then conserved in thermoses with ice packs during the transport to the laboratory for the analyses (Figure 6.3c).

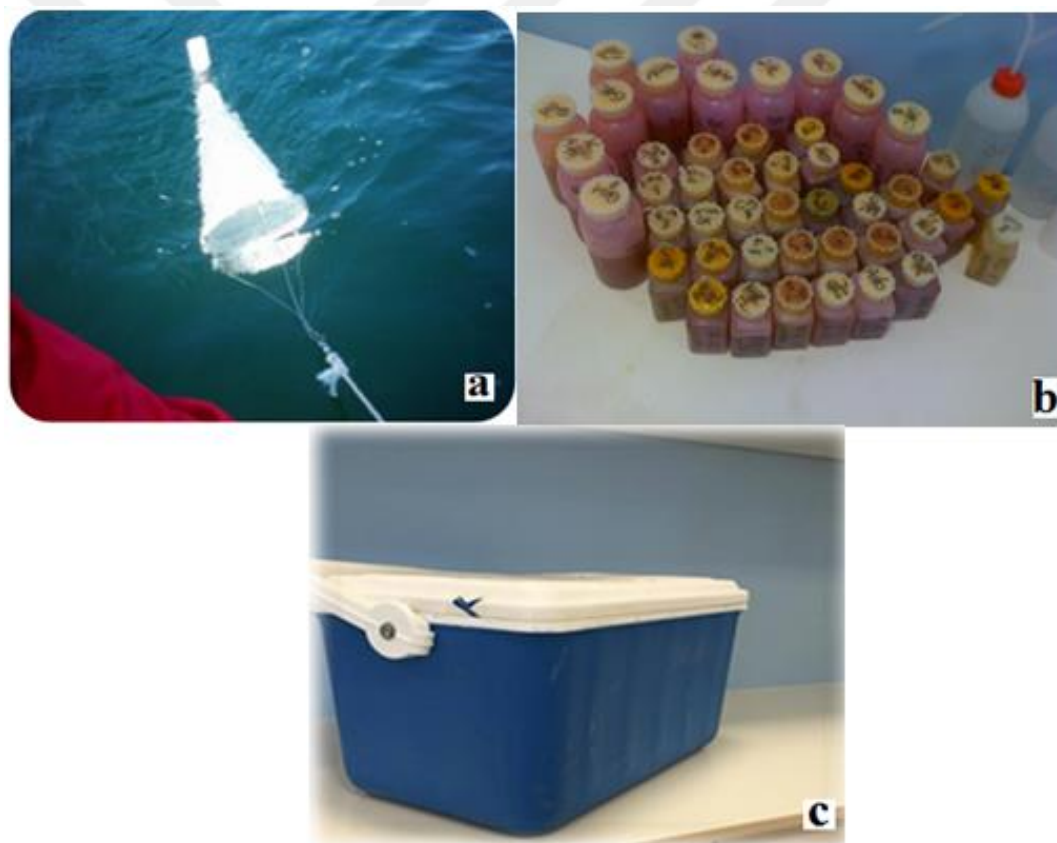


Figure 6.3 Phytoplankton sampling a) Plankton net, b) Collected samples, and c) Cooler with ice packs

6.2.6.2 Physicochemical Variables

Environmental variables such as temperature, pH, conductivity, salinity, total suspended solids and total dissolved solids (TDS) were recorded *in situ* using a YSI professional plus oxygen–temperature meter from just beneath of the surface in the

stations (Figure 6.4a). Water clearness was seasonally estimated using a 20-cm Secchi disk during the different seasons (Figure 6.4b). Geographical characteristics such as altitude, latitude, and longitude were taken from a geographical positioning system (Garmin Vista HCx model GPS).



Figure 6.4 Environmental variables sampling a) YSI professional plus, b) Secchi disk

6.2.3 Laboratory Examinations

6.2.3.1 Chemical Variables

Analyses of chemical variables (e.g., TN, N-NH₄, N-NO₃, N-NO₂, TP, and P-PO₄), dissolved oxygen (DO) and biological oxygen demand (BOD₅) were carried out by an accredited laboratory of DOKAY-ÇED (Ankara) using the standard methods (APHA, 2012).

6.2.3.2 Identification and Enumeration of Phytoplankton

Phytoplankton identification, enumeration, and analyses were carried out in the hydrobiology laboratory of the biology department (Gaziantep University, Turkey) according to standard methods (Utermöhl, 1958; CEN, 2006, 2014). The identification of phytoplankton taxa was carried out using a light microscope (Olympus BX53) attached DP73 model digital camera with imaging software (Olympus CellSens Vers. 1.6) (Figure 6.5a). Phytoplankton enumeration was carried out using the following steps. First, 100–150 ml sampling waters were allowed to settle for 24–48 hours in graduated cylinders (Figure 6.5b). After the waiting, about 85–135 ml sample is carefully siphoned (Figure 6.5c). Remain sample were put in counting chamber, placed on the inverted microscope and permitted to settle for the

least 1 hour (Figure 6.5d). Phytoplankton were then counted following two perpendicular transects at magnifications of 400–600× using an inverted microscope (Olympus CKX41). At least 350-500 species at cell level were counted per sample. Unicellular species as well as colonial and filamentous species were counted as one individual species. Functional groups of phytoplankton assemblages were determined based on the system established by Reynolds et al. (2002) and Padisák et al. (2003, 2009). Identification books (Ettl, 1983; Komárek and Fott, 1983; Popovsky and Pfiester, 1990; Krammer and Lange-Bertalot, 1991a, b; Komárek and Anagnostidis, 1998; Krammer and Lange-Bertalot, 1999a, b; John et al., 2002; Wehr and Sheath, 2003) were used for the identification of phytoplankton. Total biovolume for each taxon was calculated by multiplying the species proportion by the unitary cell biovolume of the taxon calculated using geometric formula proposed by Rott (1981), Hillebrand et al. (1999) and Sun and Liu (2003).



Figure 6.5 Phytoplankton identification and enumeration: a) identification, b) settling (24-48 hours), c) siphoning process, and d) counting chamber on inverted microscope

6.2.4 Multivariate analyses

A canonical correspondence analysis (CCA) using the CANOCO program was used to evaluate the interactions between environmental factors and phytoplankton taxa in the lakes and reservoirs. Unimodal response models were applied for the ordination of analyses along gradients. Prior to this, all predictor factors except pH were transformed $\log(x+1)$ (ter Braak and Smilauer, 1998; Leps and Smilauer, 2003).

Weighted averaging (WA) regression of CALIBRATE program (Juggins and ter Braak, 1992) was used to estimate species optima (u_k) and tolerance (t_k) levels for environmental variables. Weighted averaging indices evaluate both an individual pressure (e.g., nutrients) and associated stressors of environmental factors on the biota. Moreover, calibration is a method widely applied in more complex analyses, such as ordination methods (Leps and Smilauer, 2003; Çelekli and Öztürk, 2014).

Only species having a percentage of over 1% were selected for the multivariate analyses. In order to understand the relationships between the environmental factors, a Pearson's correlation test was carried out.

The hypothesis of no difference in phytoplankton assemblages (similarities in species composition) between lakes and reservoirs was tested by means of vegetation tables (species-by-sites matrix) used by Schaumburg et al. (2004), while the Monte Carlo permutation test was used to test the hypothesis on relationship between the phytoplankton taxa and the environmental variable matrices. On the other hand, the hypothesis of no difference between the ecological status of lakes and reservoir, the variance analysis (one-way ANOVA) was employed at 95% confidence interval using the program SPSS (IBM statistics version 23).

6.3 Results

6.3.1 Predictor variables

The studied watercourses showed different physicochemical characteristics spatially and temporally. The mean values of physicochemical variables in the lakes and reservoirs are given in Table 6.2. The lakes and reservoirs ecosystems showed alkaline water during the study period with pH values ranged from 8.41 at Lake Gölhisar to 9.02 at Lake Avlan. The highest mean temperature (24.9 °C) was measured in Toptaş reservoir located at 182 m a.s.l, while the lowest mean temperature value (17.5 °C) was measured in Osmankalfalar reservoir located at 1428 m a.s.l. With regard to nutrient contents and conductivity, higher values of TP (420.8 µg L⁻¹), P-PO₄ (176.0 mg L⁻¹), N-NO₂ (35.1 mg L⁻¹) and N-NO₃ (532.0 mg L⁻¹) were recorded in Osmankalfalar reservoir, when Lake Gölhisar was characterized by the highest values of conductivity (1308.3 µS cm⁻¹) and BOD₅ (26.8 mg L⁻¹). In related to water transparency, the highest Secchi depth (SD) value (3.43 m) was found at Yapraklı reservoir, while the lowest value (0.32 m) was recorded at Lake Yazır. The highest values of CaCO₃ (37.5 mg L⁻¹) and salinity (0.72 ppt) were found at Lake Gölhisar, in contrast to Geyik and Çayboğazı reservoirs which showed the lowest values 8.8 mg L⁻¹ and 0.11 ppt respectively. On the other hand, the highest dissolved oxygen (8.72 mg L⁻¹) and total suspended solids (48.6 mg L⁻¹) were recorded at Lake Avlan, while Lake Gölhisar and Yapraklı reservoir were associated with the lowest values 7.04 mg L⁻¹ and 3.32 mg L⁻¹, respectively.

Table 6.2 Mean of main physical chemical characteristics from sampling stations

	Göhlisar	Çayboğazı	Avlan	Geyik	Çavdır	Toptaş	Yapraklı	Osmankalfalar	Yazır
Altitude (m)	947	1197	1010	471	1122	182	1135	1428	1481
pH	8.41	8.72	9.02	8.54	8.83	8.52	8.71	8.43	8.21
Temperature (°C)	20.3	18.2	21.3	22.5	17.8	24.9	18.6	17.5	19.1
Conductivity ($\mu\text{S cm}^{-1}$)	1308.3	228.9	360.3	152.2	399.5	584.8	348.3	291.8	385.5
DO (mg L^{-1})	7.04	8.21	8.72	7.91	8.64	8.14	8.43	8.33	7.61
TSS (mg L^{-1})	5.52	5.23	48.6	5.24	7.63	4.84	3.32	25.2	13.0
BOD ₅ (mg L^{-1})	26.8	10.1	11.5	15.8	15.7	6.04	8.92	4.82	12.8
COD (mg L^{-1})	108.2	36.3	56.0	58.9	61.0	27.3	39.5	23.1	47.9
TOC (mg L^{-1})	26.8	14.9	5.31	21.5	20.1	5.83	7.24	6.44	12.3
TN ($\mu\text{g L}^{-1}$)	1569.3	508.7	865.2	766.2	910.0	850.0	747.0	1073.8	950.7
N-NH ₄ (mg L^{-1})	219.3	120.3	161.3	138.0	279.0	133.5	142.8	155.8	100.0
N-NO ₂ (mg L^{-1})	15.6	16.6	29.6	14.0	10.3	9.43	8.72	35.1	2.34
N-NO ₃ (mg L^{-1})	587.5	195.3	252.5	214.5	410.0	380.3	253.5	532.0	215.0
TKN (mg L^{-1})	1246.3	307.6	681.8	393.0	521.3	362.8	468.8	585.5	841.0
TP ($\mu\text{g L}^{-1}$)	309.5	182.0	365.5	248.0	274.3	171.8	196.0	420.8	150.0
P-PO ₄ (mg L^{-1})	108.3	63.0	148.8	60.0	106.3	60.0	61.8	176.0	21.0
Salinity (ppt)	0.72	0.11	0.23	0.12	0.21	0.34	0.24	0.22	0.23
CaCO ₃ (mg L^{-1})	37.5	12.5	15.0	8.8	17.5	21.3	22.5	15.0	20.0
SD (m)	0.62	1.81	0.43	1.44	2.04	0.72	3.43	0.91	0.32

Pearson's correlation test was applied to examine the relationships between the environmental factors. As it can be seen in Table 6.3, Dissolved oxygen (DO), is negatively correlated with the water temperature ($r = -0.462$, $p < 0.01$). Total phosphorus (TP) is positively correlated with BOD₅ ($r = 0.373$, $p < 0.01$) and negatively correlated the temperature ($r = -0.307$, $p < 0.05$). Total nitrogen (TN) is negatively correlated with water temperature ($r = -0.474$, $p < 0.01$) and positively correlated with COD ($r = 0.375$, $p < 0.01$) when water alkalinity (CaCO₃) is positively correlated with water conductivity ($r = 0.755$, $p < 0.01$) and water salinity ($r = 0.776$, $p < 0.01$). The Secchi Disk depth is negatively correlated with total phosphorus concentration ($r = -0.301$, $p < 0.05$). The Pearson's correlation test indicated that there was no correlation between pH and all the tested environmental variables. There was no also correlation between altitude and the other parameters, excepted temperature which indicated a significant negative correlation with altitude ($r = -0.421$, $p < 0.01$).

The seasonal variation of physicochemical parameters was analyzed. Figure 6.6 indicated the seasonal distribution of some these parameters. This Figure showed that, there were no normal distribution of the physicochemical variables during the study period.

Table 6.3 Pearson's correlation test result among the environmental variables

	Altitude	pH	Temperature	Conductivity	BOD ₅	TN	N-NH ₄	N-NO ₂	N-NO ₃	TP	P-PO ₄
pH	0.160										
Temperature	-0.421**	0.175									
Conductivity	-0.078	0.116	0.233								
DO	0.156	0.091	-0.462**	-0.163							
BOD ₅	0.047	0.231	0.020	0.499**							
TOC	0.137	-0.007	0.136	0.259	0.702**						
TN	0.013	-0.105	-0.474**	0.252	0.259						
N-NH ₄	0.258	-0.107	-0.103	0.079	.103	-0.024					
N-NO ₂	0.273	0.146	-0.313*	-0.017	0.139	0.132	0.684**				
N-NO ₃	-0.038	-0.153	-0.497**	0.137	0.054	0.825**	-0.231	-0.004			
TKN	0.028	0.032	-0.237	0.296*	0.239	0.867**	-0.254	-0.140	0.686**		
TP	0.212	0.188	-0.307*	0.177	0.373**	0.190	0.492**	0.761**	0.072		
P-PO ₄	0.121	0.284	-0.315*	0.154	0.332*	0.270	-0.019	0.472**	0.278	0.846**	
Salinity	-0.038	0.110	0.088	0.980**	0.553**	0.399**	0.056	0.017	0.275	0.219	0.220
SD	0.061	0.036	-0.046	-0.185	-0.091	-0.163	-0.181	-0.303*	0.008	-0.301*	-0.242
Alkalinity	0.101	0.008	-0.023	0.755**	0.317*	0.411**	0.043	0-0.010	0.271	0.080	0.082

** . Correlation is significant at the 0.01 level (2-tailed).

* . Correlation is significant at the 0.05 level (2-tailed).

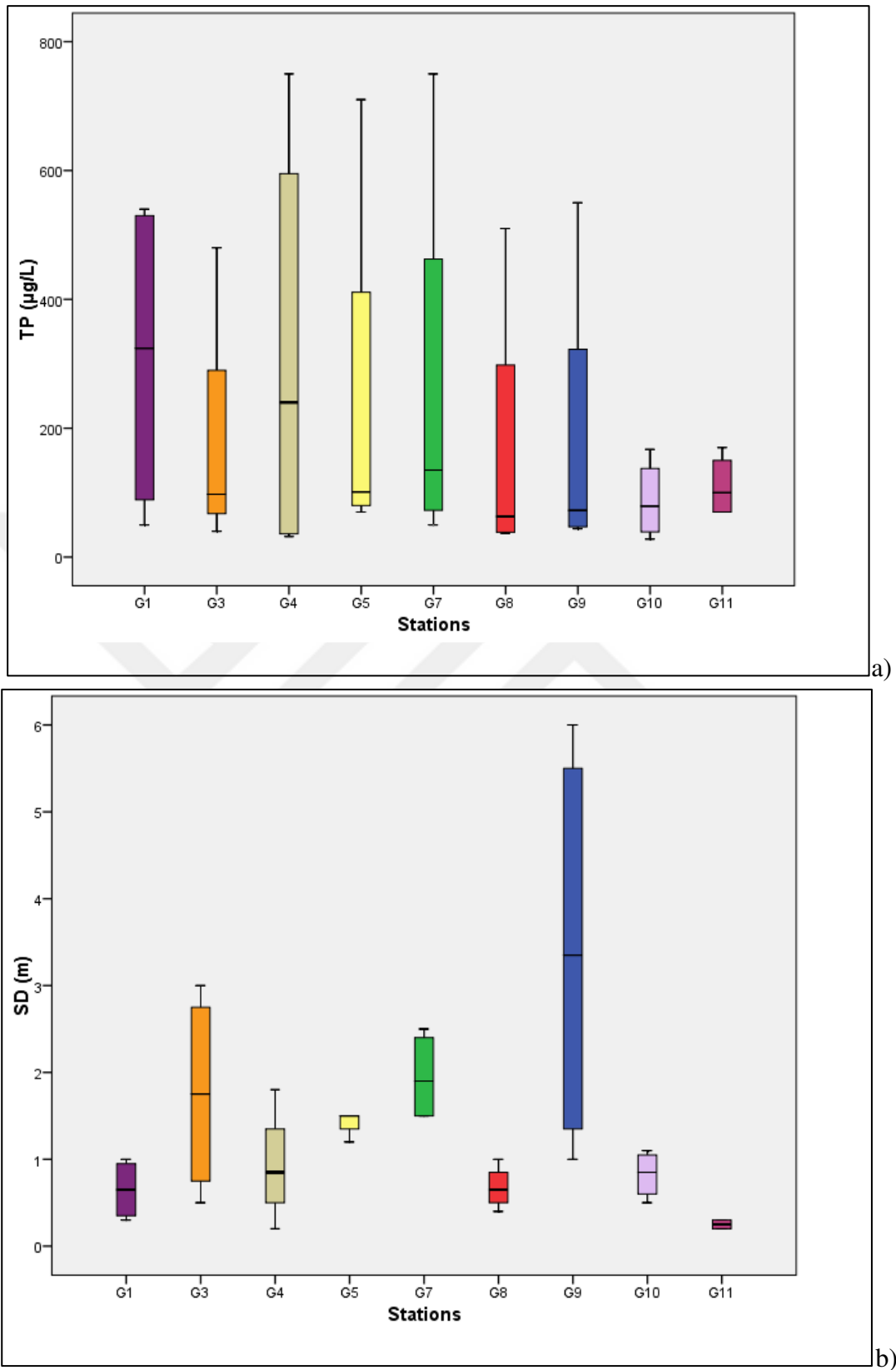


Figure 6.6a Box plot diagrams showing the seasonal distribution of a) TP, and b) SD in the stations. The horizontal black line in the boxes indicates the median value of the variables

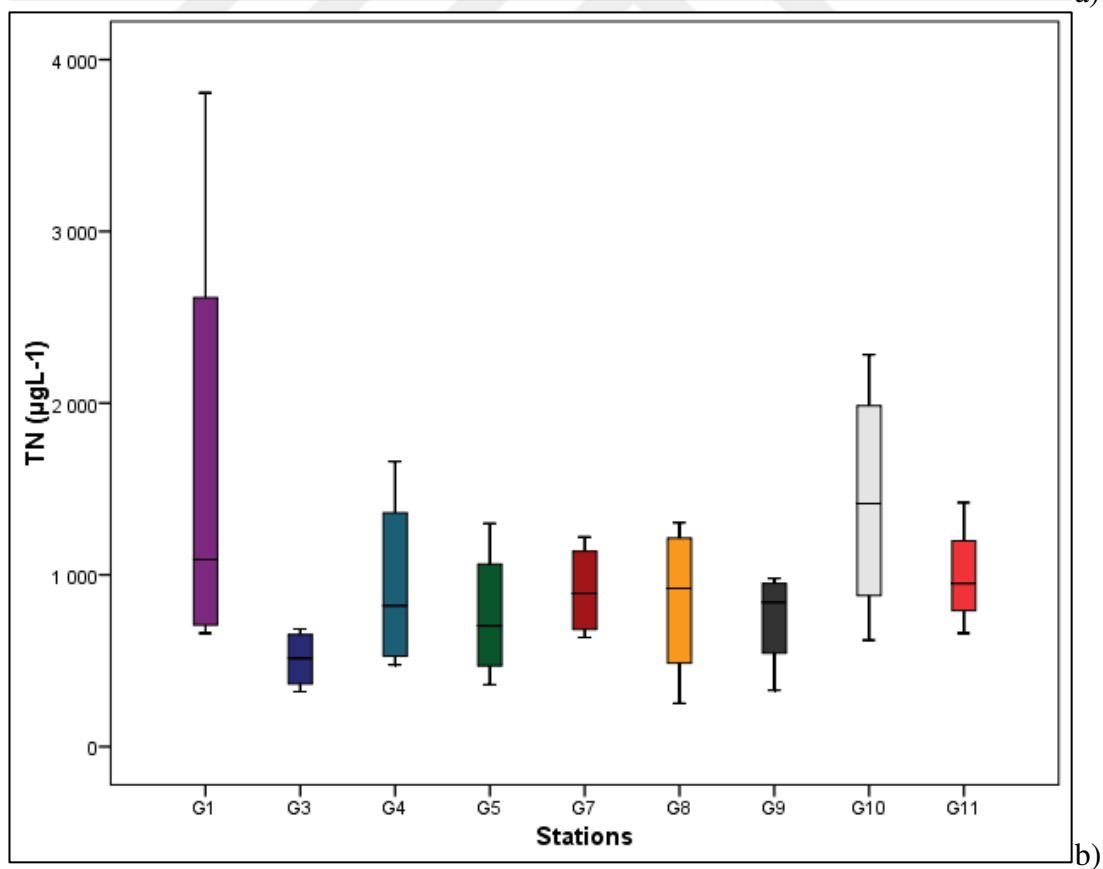
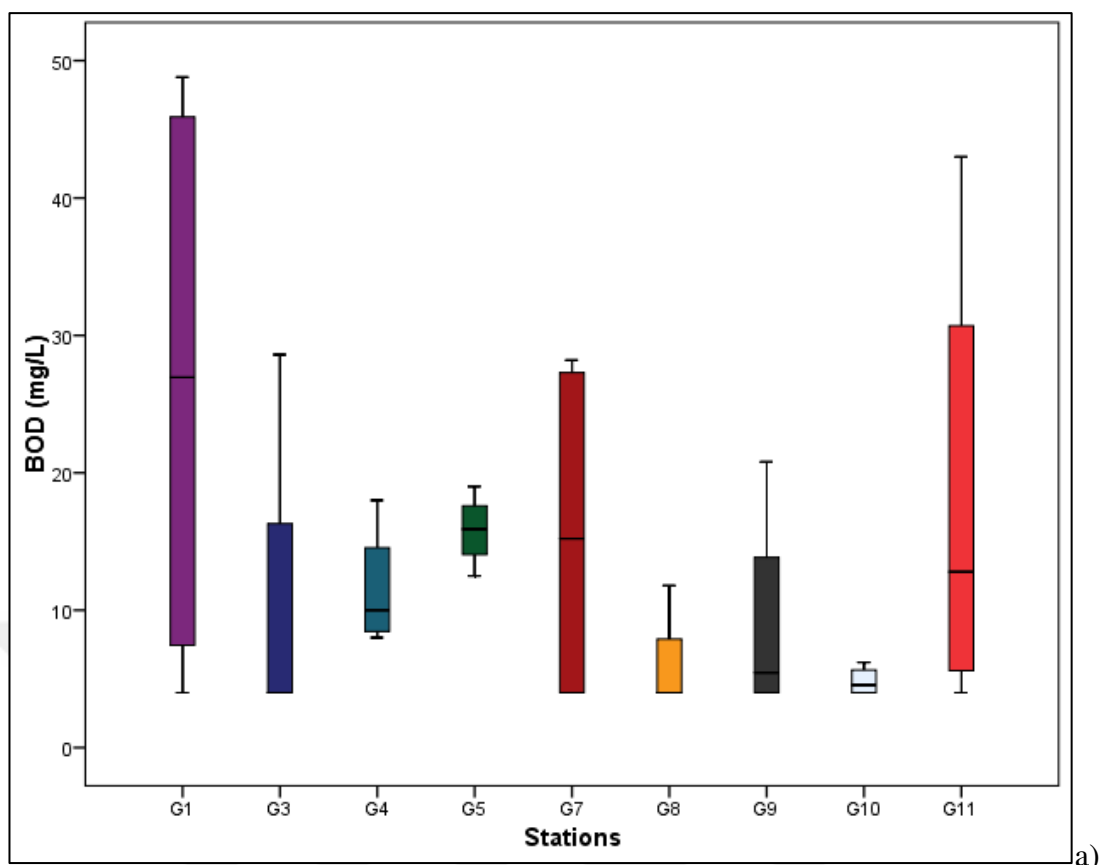


Figure 6.6b Box plot diagrams showing the seasonal distribution of a) BOD, and b) TN in the stations. The horizontal black line in the boxes indicates the median value of the variables

6.3.2 Phytoplankton Composition and Distribution

The phytoplankton composition of three lakes and six reservoirs was characterized by 206 species belonging to 8 phytoplankton phyla including Bacillariophyta, Cyanobacteria, Chlorophyta, Charophyta, Miozoa, Cryptophyta, Euglenozoa, and Ochrophyta were recorded during the study period (Appendix F). Fifteen functional groups: **B, D, E, F, G, J, L_M, L_O, M, MP, P, T, W1, X1, and Y** were recorded to be predominants and were subsequently examined in phytoplankton assemblages of studied water bodies (Table 6.4). Photographs of some non-diatom species and diatom species given in Figure 6.7.

Table 6.4 Phytoplankton functional groups (FG) and their descriptor species and factor values in the lakes and reservoirs

FG	Descriptor species	G1	G3	G4	G5	G7	G8	G9	G10	G11
B	<i>Cyclotella ocellata</i>	-	-	-	3	3	3	3	3	-
	<i>Cyclotella iris</i>	-	3	3	3	-	3	3	-	3
D	<i>Fragilaria biceps</i>	-	-	-	-	-	-	-	5	-
E	<i>Dinobryon divergens</i>	2	-	-	-	-	-	-	-	-
F	<i>Botryococcus braunii</i>	-	-	-	-	-	-	-	5	-
G	<i>Pandorina morum</i>	-	-	-	1	-	-	-	-	-
	<i>Coelastrum microporum</i>	-	-	-	-	-	-	-	1	-
J	<i>Tetraëdron minimum</i>	-	-	-	-	-	-	-	1	-
	<i>Pediastrum simplex</i>	-	-	-	-	-	1	-	-	-
	<i>Pediastrum duplex</i>	-	-	-	1	-	-	-	-	-
L_M	<i>Ceratium furcoides</i>	-	-	-	-	-	-	-	0	-
	<i>Ceratium hirundinella</i>	-	0	-	0	-	-	-	-	-
L_O	<i>Peridinium cinctum</i>	-	-	-	-	-	5	5	-	3
	<i>Peridiniopsis cunningtonii</i>	3	-	-	-	-	-	-	-	-
M	<i>Microcystis aeruginosa</i>	0	-	0	-	-	-	-	0	-
MP	<i>Ulnaria ulna</i>	-	5	-	-	-	-	-	-	-
	<i>Cymbella excisa</i>	5	5	5	-	-	-	-	5	5
P	<i>Fragilaria dilatata</i>	5	5	5	5	-	5	-	5	5
	<i>Staurostrum cingulum</i>	-	-	5	-	-	-	5	-	5
T	<i>Geminella interrupta</i>	-	-	-	-	5	-	-	-	-
W1	<i>Euglena acus</i>	3	-	3	-	-	-	-	0	-
X1	<i>Monoraphidium arcuatum</i>	-	-	-	-	-	-	4	-	-
	<i>Monoraphidium contortum</i>	-	4	-	-	-	-	-	-	-
Y	<i>Cryptomonas ovata</i>	3	-	3	2	-	5	-	-	5

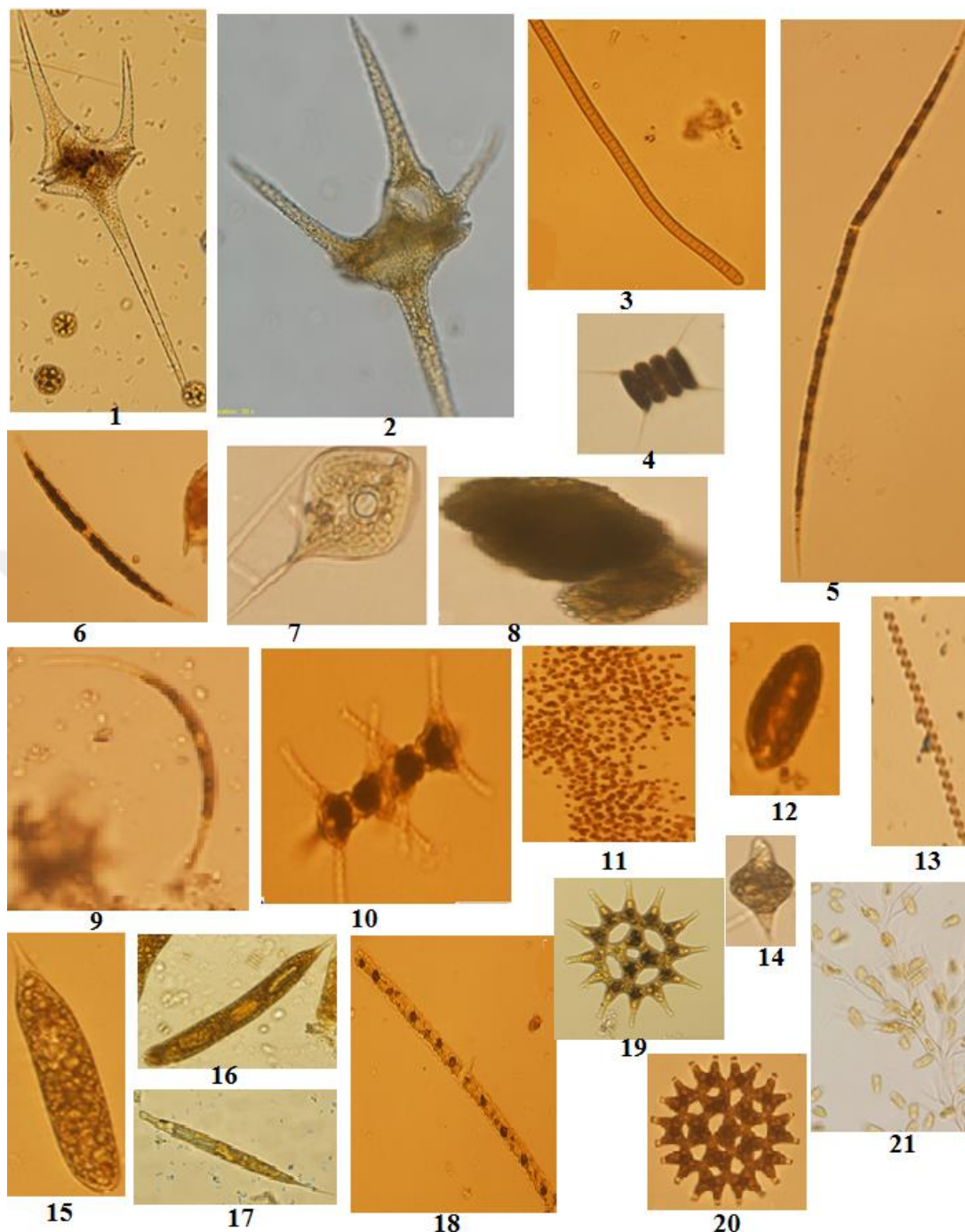


Figure 6.7 1) *Ceratium furcoides*; 2) *C. hirundinella*; 3) *Oscillatoria limosa*; 4) *Scenedesmus communis*; 5) *Closterium aciculare*; 6) *C. littorale*; 7) *Phacus caudatus*; 8) *Botryococcus braunii*; 9) *Monoraphidium arcuatum*; 10) *Staurastrum cingulum*; 11) *Microcystis aeruginosa*; 12) *Cryptomonas ovata*; 13) *Spirulina maior*; 14) *Euglena chlamydophora*; 15) *E. velata*; 16) *E. oxyuris*; 17) *E. acus*; 18) *Mougeotia quadrangulata*; 19) *Pediastrum simplex*; 20) *P. duplex*; 21) *Dinobryon divergens*

The Bacillariophyta phylum dominated the phytoplankton composition in terms of species number (107), when only one taxon (1) was observed in the Ochrophyta (Appendix B) during the study period. *Peridinium cinctum*, *C. ocellata*, *Cyclotella*

iris, *Cymbella excisa*, *Ceratium hirundinella*, *Scenedesmus communis*, *Cryptomonas ovota*, *Microcystis aeruginosa*, *Navicula lanceolata*, and *Fragilaria capucina* were commonly found in the lakes and reservoirs. The species number per site changed also from a water body to another (Figure 6.8) and this number in the lakes was generally higher than those of reservoirs. The highest species number (77) was found at Lake Gölhisar, while the lowest (45) was recorded at Osmankalfalar reservoir. The lists of species recorded at each water body are given in Appendices G to O.

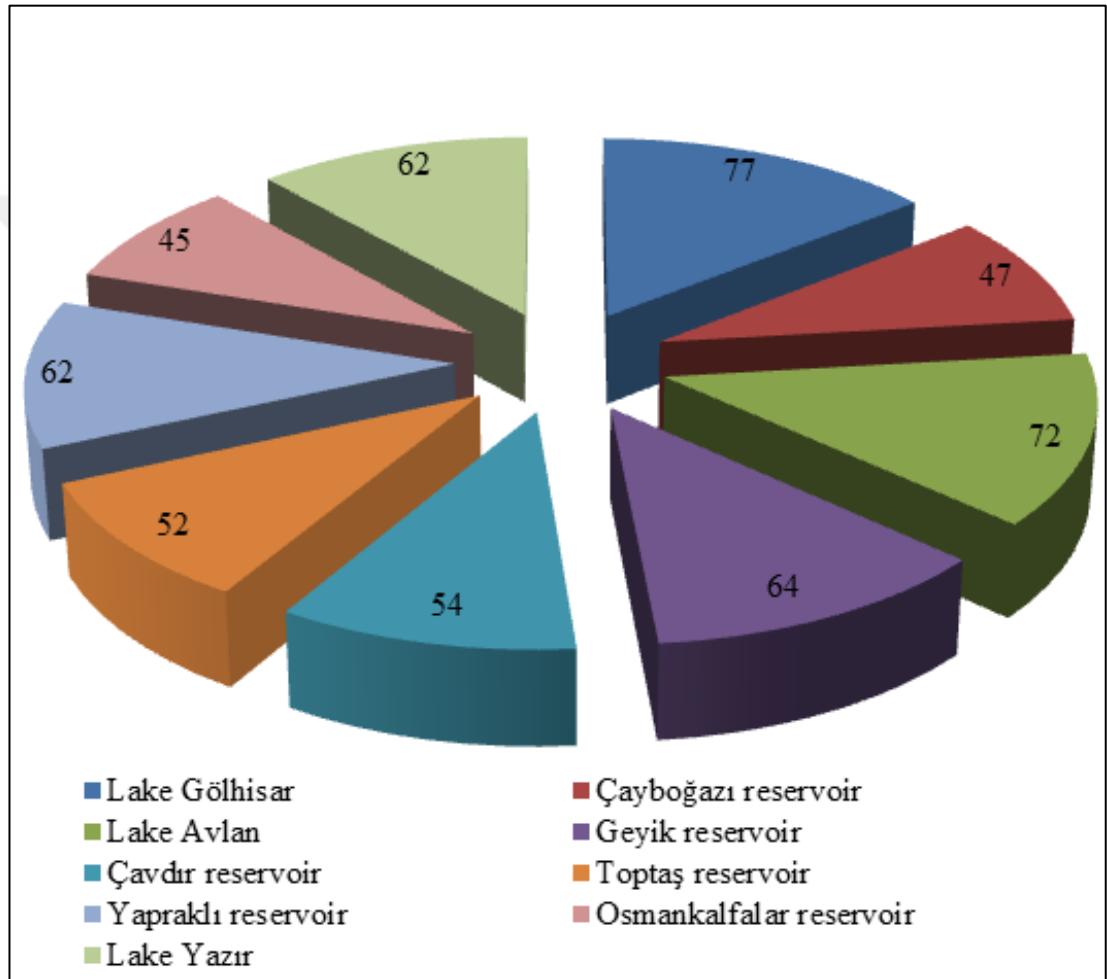


Figure 6.8 Species number per site

6.3.3 Seasonality of Phytoplankton

The phytoplankton composition and biovolume changed specially during the study period according to the seasonal changes in water characteristics. Peaks of phytoplankton abundance did not appear at the same time at the different lakes and reservoirs, and phytoplankton succession showed distinct spatial heterogeneity during the four seasons (summer 2014, fall 2014, spring 2015 and summer 2015) in the different lentic ecosystems. The Figure 6.9a,b and c show the seasonal variations

in the phytoplankton phyla' total biovolume ($\times 10^4 \mu\text{m}^3.\text{L}^{-1}$) in the lakes and reservoirs. The mainly abundant phylum was Bacillariophyta followed by the Chlorophyta phylum. In Lake Gölhisar (G1), the highest total biovolume was observed respectively in Chlorophyta, Miozoa, and Bacillariophyta phyla in fall 2014 and summer 2015 while the lowest contribution was observed in Cyanobacteria, Charophyta and Cryptophyta species in spring 2015. The lowest total biovolume was found in Ochrophyta phylum in spring 2015 and the highest contribution in Bacillariophyta phylum in summer 2014, while Charophyta phylum was absent at the Çayboğazı reservoir during all the study period. Lake Avlan (G4) and Geyik reservoir (G5) had respectively their lowest total biovolume in Ochrophyta and Euglenozoa phyla in spring 2015, while Bacillariophyta species and Chlorophyta species had their peaks of abundance at Lake Avlan and Geyik reservoir during summer 2015 and 2014 respectively. At Çavdır reservoir Chlorophyta species total biovolume was high in summer 2014 when Ochrophyta contributed in a lower part to the phytoplankton total biovolume during spring 2015. On the other hand, at the same reservoir, Bacillariophyta species were the most abundant phytoplankton group in spring 2015. The lowest contributions were found in Cyanobacteria and Euglenozoa during summer and fall 2014, while Miozoa and Chlorophyta were abundant in summer 2014 and 2015 respectively at Toptaş reservoir. Bacillariophyta and Chlorophyta were abundant at Yapraklı reservoir during summer 2014 and 2015 respectively, while Charophyta and Bacillariophyta phyla dominated the phytoplankton community of Yapraklı reservoir during fall 2014 and summer 2015. The phytoplankton abundance was mainly dominated at Osmankalfalar reservoir by Bacillariophyta, while at Lake Yazır it was dominated by Miozoa. Charophyta phylum was not found at Toptaş reservoir, while Cryptophyta and Charophyta were absent at Osmankalfalar reservoir during the study period.

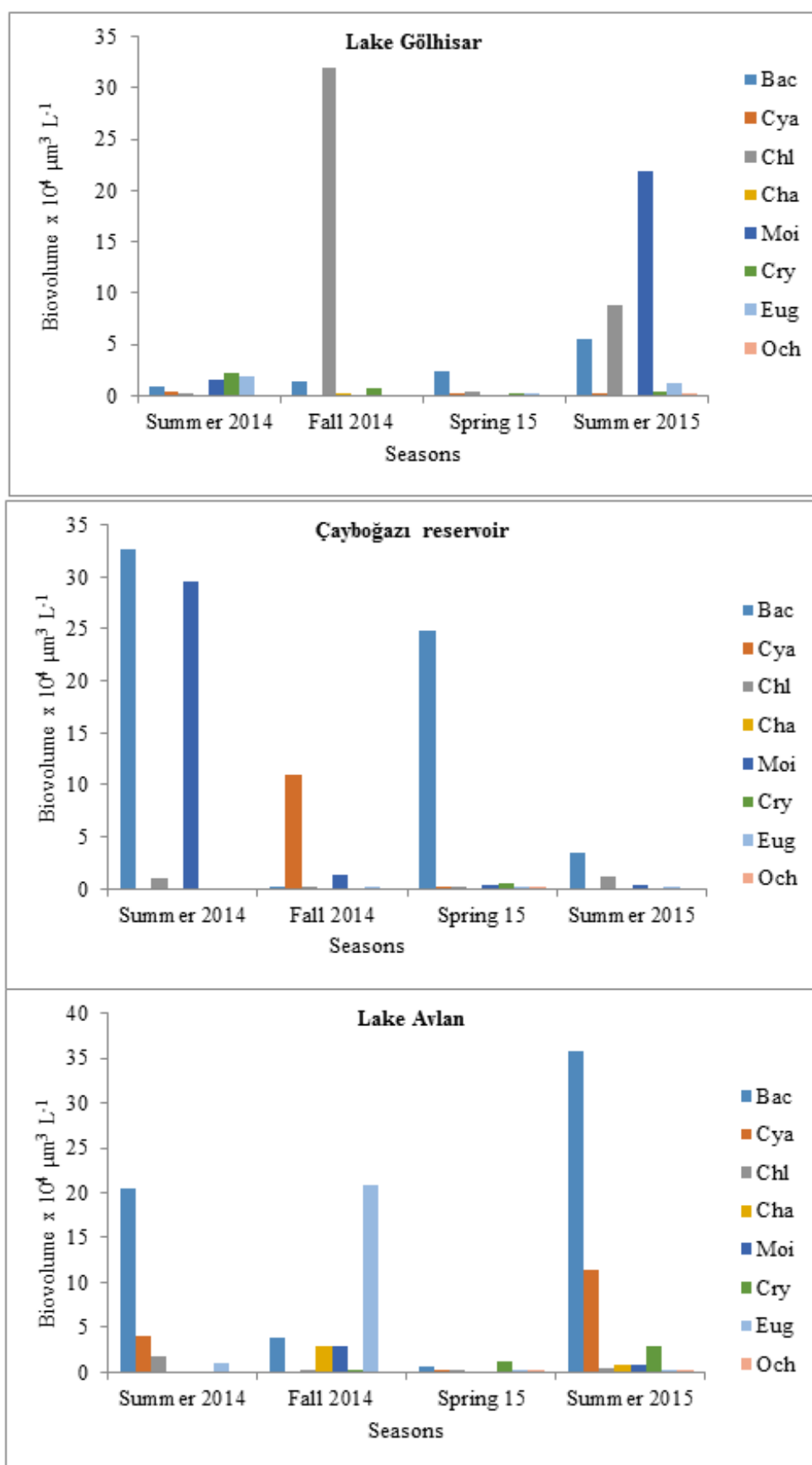


Figure 6.9a Seasonal variations in phytoplankton at Lakes Gölhisar, Avlan and Çayboğazı reservoir

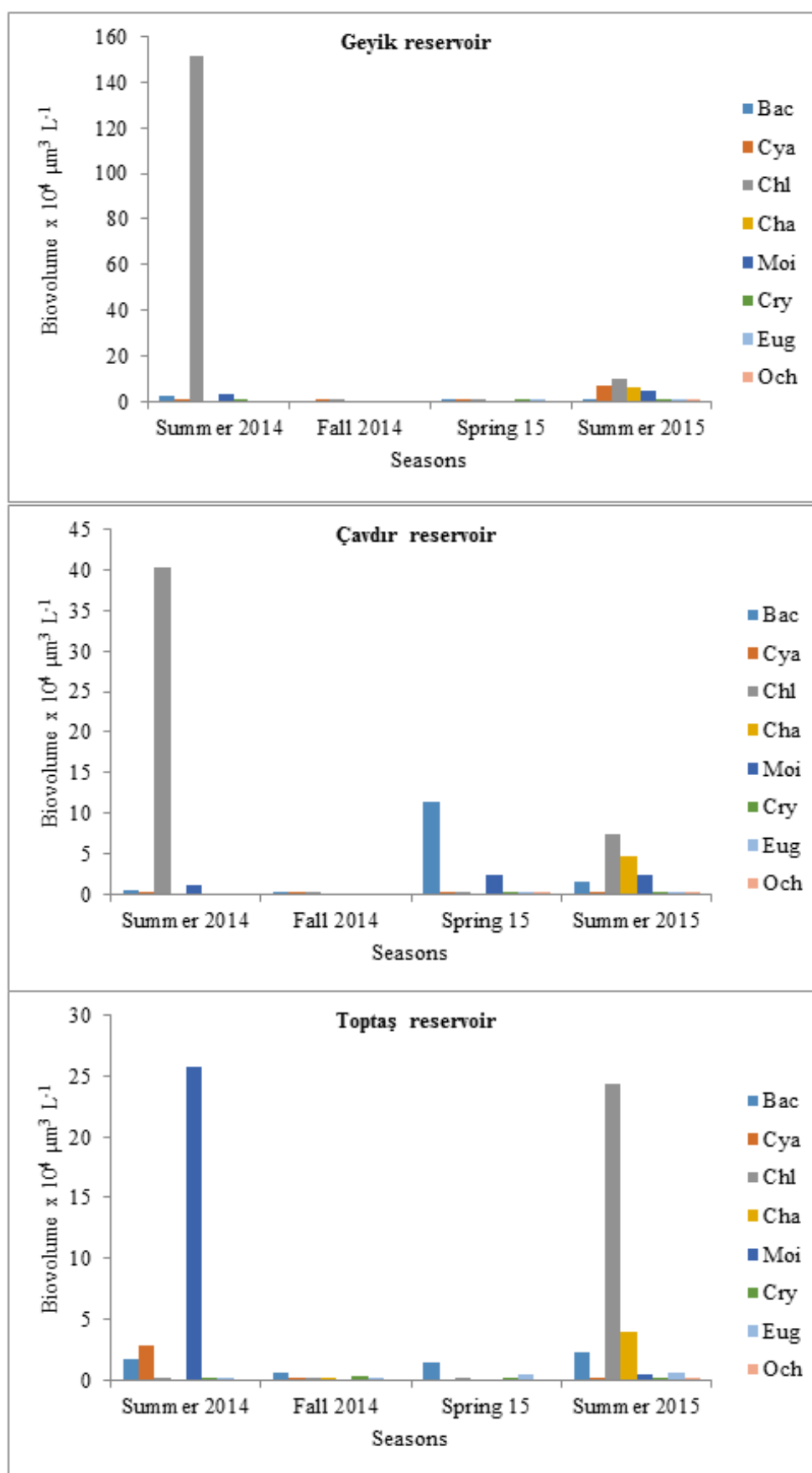


Figure 6.9b Seasonal variations in phytoplankton at Geyik, Çavdır and Toptaş reservoirs

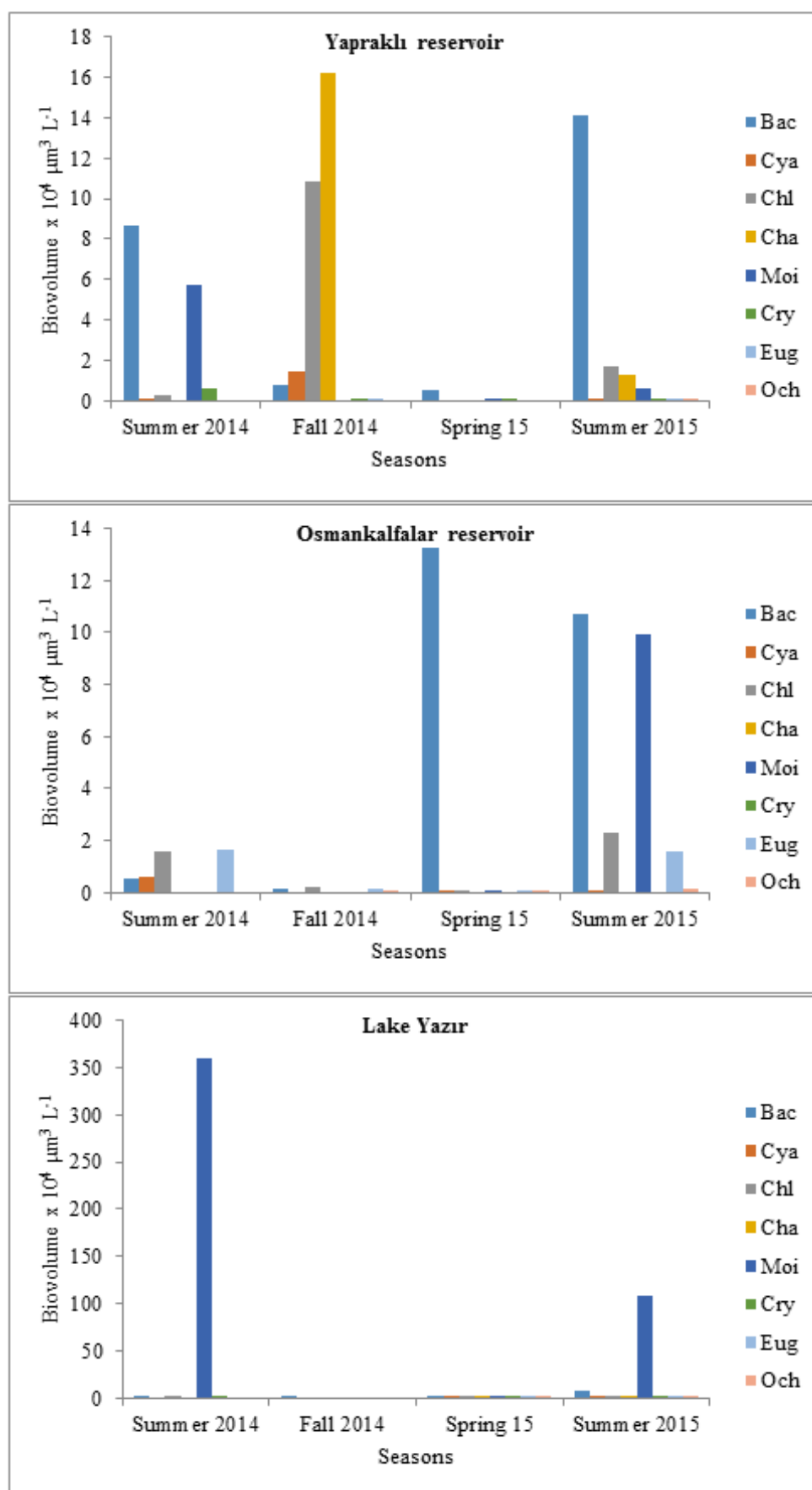


Figure 6.9c Seasonal variations in phytoplankton at Yapraklı, Osmankalfalar reservoirs and Lake Yazır

During the present study, it has appeared that the species number changed not only per site, but changed also per season. the species number changed from a minimum of 10 at Çavdır reservoir in summer 2014 to a maximum of 49 at Lake Avlan in summer 2015. This species number was generally higher in 2015 than those of 2014 (Table 6.5).

Table 6.5 Species number per season during the study period

Sites	Summer 2014	Fall 2014	Spring 2015	Summer 2015
Lake Gölhisar	16	18	33	46
Çayboğazı reservoir	13	14	25	33
Lake Avlan	11	21	23	49
Geyik resevoir	15	16	25	42
Çavdır reservoir	10	16	27	34
Toptaş reservoir	10	14	28	38
Yapraklı reservoir	12	19	18	43
Osmankalfalar reservoir	13	18	24	30
Lake Yazır	14	17	31	34

6.3.4 Species-Environmental variables Relationships

A canonical correspondence analysis was performed to evaluate the relationship between phytoplankton assemblages and environmental factors of the lakes and reservoirs. The results of CCA confirmed by the forward selection using the Monte Carlo test ($p=0.012$, $F = 1.977$) showed that 13.6% of cumulative percentage variance of species data and 93.2% between species-environment correlations were explained by the first two CCA axes. (Table 6.6). The partial CCA results (Table 6.7) indicated that explanatory factors such as pH, TP, N-NO₃, TOC, and conductivity had a key role ($p<0.02$) on the phytoplankton composition, dynamic and distribution among the water ecosystems during the study period.

Table 6.6 Summary of CCA results using Monte Carlo permutation test for phytoplankton species-environment factors relationship

Axes	λ_1	λ_2	λ_3	λ_4	Total inertia
Eigen values	0.820	0.697	0.694	0.595	11.189
Species-environment correlations	0.964	0.932	0.965	0.909	
Cumulative percentage variance					
of species data	7.3	13.6	19.8	25.1	
of species-environment relation	18.4	34.1	49.7	63.0	
Sum of all eigen values					11.189
Sum of all canonical eigen values					4.452
Test of significance of first canonical axis : eigen value = 0.820					
F-ratio = 1.977 p-value = 0.0120					

Table 6.7 Partial CCA result of Environmental variables

Environmental variables	λ	F	P
Temperature	0.578	2.166	0.002
Conductivity	0.545	2.034	0.002
Altitude	0.532	1.985	0.004
CaCO ₃	0.612	2.298	0.004
TOC	0.580	2.174	0.004
TP	0.575	2.153	0.002
PO ₄	0.554	2.071	0.004
NO ₂	0.518	1.929	0.004
BOD ₅	0.438	1.622	0.004

With regard to the CCA ordination (Figure 6.10), environmental factors such as pH, nutrients, TOC, conductivity, and water temperature governed phytoplankton distribution in the lakes and reservoirs. *C. ovata*, *N. lanceolata* and *Cryptomonas rostratiformis* *C. hirundinella*, *Zygnema pectinatum*, *Phacus caudatus*, *Microcystis flosaquae*, *Ulnaria ulna*, *Navicula cryptocephala*, *Gomphonema truncatum*, *Epithemia sorex*, and *Euglena proxima* were attributed to TP, P-PO₄, N-NO₃, in Geyik (G7) and Osmankalfalar (G10) reservoirs. *Fragilaria incognita*, *Navicula slesvicensis*, *Gomphonema parvulum*, *Clevamphora ovalis*, *C. ocellata*, *Cyclotella meneghiniana*, *Cocconeis placentula*, *Cymbella cymbiformis*, *Epithemia turgida*, *F. capucina*, *Diatoma tenuis*, *Gyrosigma attenuatum*, *Geminella interrupta*, *Scenedesmus obliquus*, *Euglena anabaena* and *E. chlamydomphora* preferred high N-NO₂ and conductivity in Lake Gölhisar (G1). Species such as *Anabaena catenula*, *Cosmarium punctulatum*, *M. aeruginosa* *Cymbella gracilis*, *Diatoma vulgaris*,

Eunotia tenella, *Fragilaria dilatata*, *Melosira lineata*, *Melosira varians*, *Chlamydomonas globosa*, *Pediastrum duplex*, *Chroococcus turgidus*, *Staurastrum crenulatum*, *Mougeotia boodlei* and *Trachelomonas scabra* were associated with warm waters while TOC governed especially the distribution of *Pediastrum boryanum*, *P. simplex*, *Navicula capitatoradiata*, and *Eunotia glacialis* in Toptaş reservoir (G8). The distribution of *C. iris*, *E. sorex*, *Chlamydomonas lapponica*, and *E. proxima* in Çayboğazı reservoir (G3) were associated with high alkaline water.

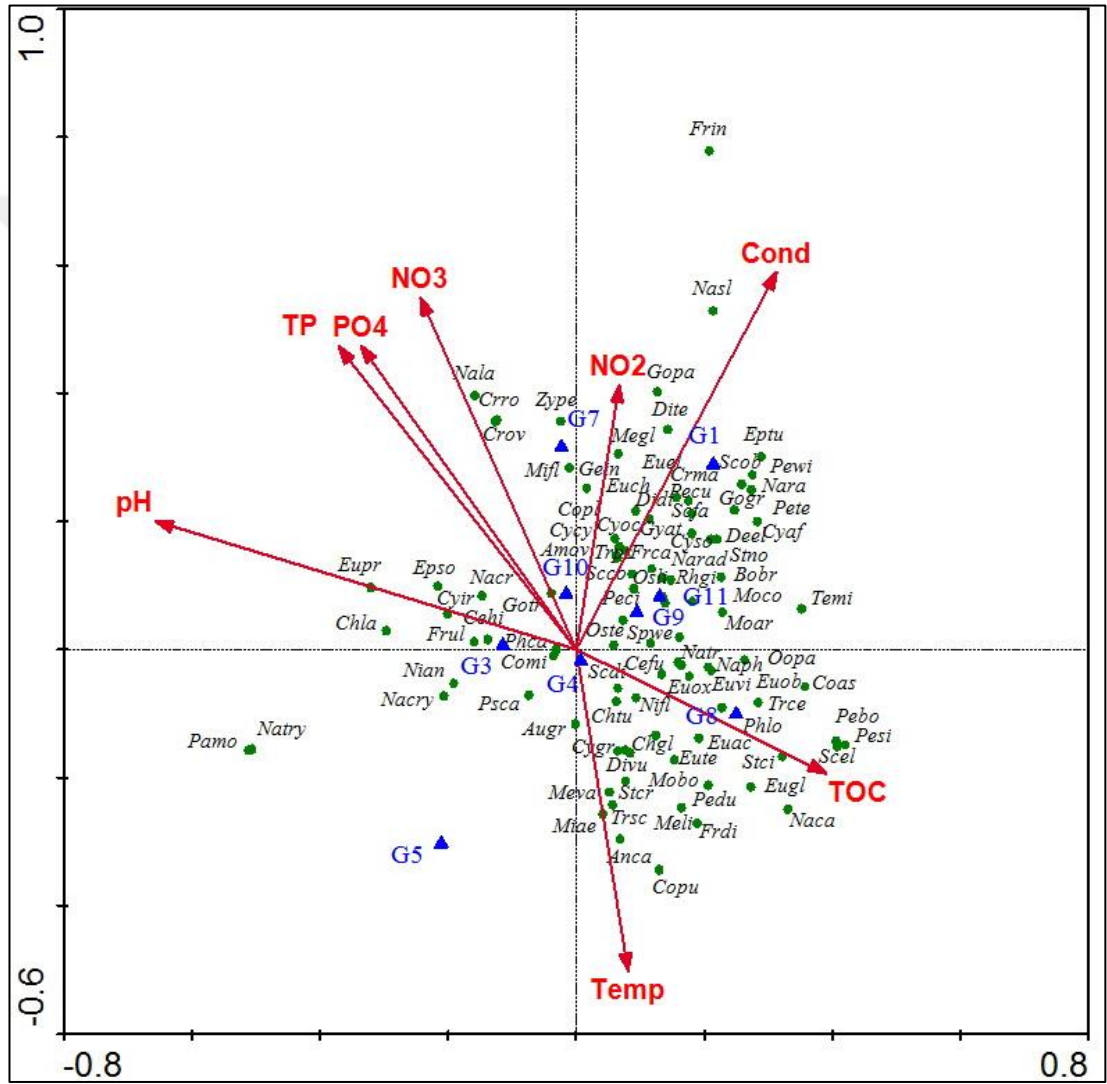


Figure 6.10 Canonical correspondence analysis ordination of species-environmental relationships in the lakes and reservoirs (up triangular). Abbreviations of physico-chemical variables are given in the list of symbols/abbreviations. For species codes see Appendix F

In addition to the canonical correspondence analysis, diatom species-environment relationships were evaluated using weighted average (WA) regression. The results of the weighted averaging regression evaluation of species optima (u_k) and tolerance (t_k)

scores to environmental parameters are given in Table 6.8. *F. capucina*, *S. communis*, *C. ocellata*, *C. turgidus*, *C. excisa*, *Euglena oxyuris*, and *G. parvulum* were common species in lakes and reservoirs with high N₂ (Hill's diversity) values.

According to these results, *C. ovata*, *N. lanceolate*, *Euglena viridis*, *Tetraedron minimum*, *Coelastrum microporum*, *Cymbella tumidula*, *E. sores*, *F. incognita*, *G. parvulum* and *C. rostratiformis* preferred high nutrients (TP, P-PO₄, N-NO₃ and N-NO₂). On the other hand, *Stauroneis nobilis*, *Staurastrum cingulum*, *Scenedesmus ellipticus*, *P. boryanum*, *Navicula tripunctata*, *Navicula phylepta*, *Mougeotia parvula*, *N. capitatoradiata*, *G. attenuatum*, *Ulnaria biceps*, *C. gracilis*, *Cymbella tumida*, *Botryococcus braunii*, *Ulothrix subconstricta*, *M. boodlei*, *E. tenella*, *Euglena oblonga*, *D. vulgaris*, *C. punctulatum*, *A. catenula*, *P. cinctum*, *Pandorina morum*, *Nitzschia angustata*, *Navicula cryptotenella*, and *G. truncatum* were associated to lower TP and N-NO₃. Phytoplankton species such as *G. parvulum*, *F. incognita*, *D. tenuis*, *E. turgida* var. *granulata*, and *E. anabaena* were associated with high conductivity, in contrast to *U. subconstricta*, *T. scabra*, *S. crenulatum*, *Spirogyra weberi*, *P. morum*, *N. cryptotenella*, *M. aeruginosa*, *M. lineata*, *M. varians*, *E. glacialis*, *C. punctulatum*, *A. catenula*, and *Aulacoseira granulata* which preferred lower conductivity.

In related to water temperature, *A. catenula*, *C. ovalis*, *C. globosa*, *C. punctulatum*, *Coelastrum astroideum*, *Euglena acus*, *E. oxyuris*, *E. glacialis*, *G. truncatum*, *M. varians*, *M. aeruginosa*, *N. capitatoradiata*, *Navicula cryptotenella*, *P. morum*, *P. boryanum*, *P. Duplex*, *Phacus longicauda*, *S. cingulum*, *S. crenulatum*, *Trachelomonas cervicula*, and *T. scabra* were found to be abundant in warm water than others phytoplankton species. With regard to pH, most of the species preferred alkaline water. The results of the weighted averaging regression evaluation indicated also that species such as, *Z. pectinatum*, *S. crenulatum*, *M. flos-aquae*, *F. incognita*, *C. ovalis* preferred high values of TOC, while *Cymatopleura solea*, *C. tumidula*, *C. rostratiformis*, *U. biceps*, *G. attenuatum*, *Navicula cryptotenella*, *P. morum*, *S. nobilis*, *Trachelomonas pseudofelix* were associated with low TOC.

Table 6.8 Weighted average regression in lakes and reservoirs, u_k and t_k indicated optima and tolerance, respectively. For species codes see Appendix F

Codes	Temperature °C		pH		Conductivity $\mu\text{S cm}^{-1}$		TP $\mu\text{g l}^{-1}$		P-PO ₄ mg l^{-1}		N-NO ₃ mg l^{-1}		N-NO ₂ mg l^{-1}		TOC mg l^{-1}	
	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k
Amov	24.0	4.2	8.6	0.3	578.9	276.6	118.6	115.1	27.7	47.1	226.6	251.2	6.6	9.0	19.0	22.2
Anca	29.4	3.4	8.2	0.1	151.6	175.4	112.3	9.3	10.1	4.8	70.5	53.1	3.3	1.3	58.0	29.0
Augr	17.2	8.2	8.6	0.4	173.9	211.1	570.3	418.9	158.7	123.6	181.8	154.9	30.1	22.8	11.2	8.8
Bobr	23.0	2.2	8.8	0.4	294.7	161.3	41.8	51.8	11.4	4.9	276.9	114.2	38.9	25.6	12.7	20.5
Cefu	23.5	4.7	8.6	0.4	282.8	168.9	89.7	116.8	22.6	47.9	233.6	199.2	24.1	24.2	21.7	24.4
Cehi	20.5	5.8	8.8	0.2	293.6	107.5	197.8	230.4	66.1	105.7	253.2	192.3	17.3	32.0	12.5	12.0
Chgl	25.5	4.7	8.2	0.3	335.3	327.0	103.1	35.7	13.5	4.7	146.3	111.7	8.0	11.1	44.0	24.3
Chla	23.1	0.1	9.0	0.3	434.6	130.8	307.0	240.4	129.6	127.3	449.6	289.9	6.9	5.7	7.5	3.4
Chtu	22.1	5.7	8.5	0.4	299.2	135.1	167.6	260.0	51.9	102.8	171.5	186.8	14.7	23.6	23.2	23.0
Copl	20.8	3.2	8.5	0.3	369.6	167.6	94.2	63.1	17.8	16.9	223.8	152.1	15.0	12.6	39.0	26.9
Coas	24.7	4.1	8.3	0.2	458.8	304.4	89.4	43.6	15.1	3.3	201.2	96.3	11.6	12.1	34.4	26.4
Comi	22.2	6.6	8.7	0.3	335.0	77.1	513.1	222.0	195.7	129.6	401.9	148.5	29.5	16.7	5.0	2.2
Copu	27.3	3.2	8.3	0.3	216.2	131.9	87.2	57.6	12.2	3.2	90.1	33.8	4.1	1.6	36.6	32.1
Crma	19.0	7.1	8.2	0.2	576.3	360.6	122.9	41.3	18.4	3.2	264.7	207.3	3.6	2.5	9.8	10.0
Crov	18.5	6.4	9.3	0.7	504.7	796.8	752.0	459.7	289.4	188.9	234.6	294.6	73.9	61.3	11.7	20.5
Caro	14.8	2.7	9.0	0.7	310.5	43.4	500.8	562.9	193.1	221.7	511.2	456.6	56.1	57.5	3.4	2.9
Cyey	13.7	0.7	8.5	0.2	298.6	1.6	212.3	325.6	75.2	120.1	392.9	147.1	13.4	6.4	4.8	7.1
Cyir	19.7	5.5	9.0	0.3	395.1	119.4	441.6	333.8	195.3	167.7	392.2	209.9	11.1	11.2	10.6	7.5
Cyoc	15.0	3.7	8.5	0.2	297.0	66.3	61.0	83.1	17.1	30.6	536.0	293.4	13.7	12.0	4.2	11.9
Cyso	17.6	5.2	8.1	0.5	385.2	171.7	89.4	76.7	13.8	6.7	413.8	312.4	4.5	3.3	3.2	5.0
Cyaf	21.2	4.9	8.4	0.3	552.1	289.6	141.2	178.5	42.5	72.2	385.0	335.2	14.6	13.7	14.7	18.5
Cygr	23.2	1.9	8.7	0.3	325.9	39.0	41.8	6.3	10.5	2.2	71.4	40.4	4.0	6.0	10.2	6.7
Cytu	11.4	2.7	8.4	0.1	285.1	13.2	33.4	20.5	10.4	2.4	786.0	313.5	7.9	2.1	1.0	13.9
Dite	19.0	7.5	8.8	0.2	737.9	632.8	228.2	296.1	84.2	117.3	595.6	836.8	17.2	19.8	27.2	23.2

Table 6.8 Continue

Codes	Temperature °C		pH		Conductivity $\mu\text{S cm}^{-1}$		TP $\mu\text{g l}^{-1}$		P-PO ₄ mg l^{-1}		N-NO ₃ mg l^{-1}		N-NO ₂ mg l^{-1}		TOC mg l^{-1}	
	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k
Divu	23.0	1.8	8.7	0.3	327.3	69.6	43.8	49.1	13.3	18.0	97.3	103.3	9.5	13.4	14.0	8.5
Didi	14.0	4.1	8.3	0.3	333.3	119.7	194.5	326.2	46.6	155.6	295.7	468.4	7.4	17.1	3.3	4.4
Epso	23.5	1.3	9.0	0.7	562.9	235.0	362.5	223.8	156.8	123.8	520.0	261.8	8.4	4.6	8.4	8.7
Eptu	24.9	0.1	8.1	0.0	873.9	128.1	128.0	0.9	19.0	1.3	241.5	27.3	3.3	0.6	18.4	2.6
Euel	20.4	3.2	8.6	0.3	372.3	152.9	82.1	69.4	15.0	5.6	293.2	131.9	26.2	20.7	21.7	28.7
Euac	24.6	1.9	8.6	0.2	809.4	1062.7	212.7	303.2	79.6	113.1	186.6	137.5	8.0	12.0	15.9	17.4
Euch	21.3	6.8	8.9	0.7	462.2	274.1	395.2	483.7	145.2	195.2	194.4	117.3	46.5	50.9	8.5	6.1
Euob	28.8	4.5	8.5	0.3	635.8	231.9	62.0	54.7	13.3	5.2	96.3	125.8	4.7	1.3	10.1	6.9
Euox	18.0	5.8	8.4	0.3	321.3	104.1	54.2	105.7	18.6	40.2	356.3	194.8	14.8	17.0	4.5	7.5
Eupr	23.3	2.4	9.2	0.4	617.3	894.8	436.5	153.4	195.2	73.9	575.8	295.2	9.4	5.1	7.5	18.9
Euvi	14.6	6.4	8.5	0.8	243.7	84.7	602.7	477.4	261.4	214.8	494.3	422.8	46.4	25.2	13.0	3.2
Eugl	25.5	4.6	8.0	0.4	190.8	247.8	138.1	65.8	13.2	2.7	122.7	76.2	6.9	2.1	17.7	21.5
Eute	23.0	1.9	8.6	0.3	325.7	89.3	55.4	45.7	13.8	4.8	96.4	65.8	8.8	9.6	19.4	16.4
Frbi	13.9	1.2	8.4	0.1	283.8	2.1	42.8	7.1	11.1	2.8	528.2	68.8	13.7	0.7	1.0	13.9
Frca	17.0	5.8	8.5	0.3	391.4	337.0	140.2	175.1	39.7	69.9	365.3	415.2	14.1	18.2	15.3	22.1
Frdi	22.1	6.4	8.5	0.2	346.8	199.4	58.2	61.0	16.0	15.0	203.6	178.8	5.3	3.9	10.0	10.9
Frin	14.4	11.8	8.7	0.5	1160.1	849.7	453.9	278.4	168.7	131.7	1198.8	931.4	30.7	22.2	50.0	13.2
Frul	19.7	5.9	8.6	0.3	366.0	202.6	106.3	144.2	31.8	66.0	343.0	232.0	6.4	8.8	7.6	10.1
Gein	23.7	2.0	8.7	0.3	598.7	255.9	104.9	58.2	19.9	20.8	139.6	60.2	5.7	12.5	8.7	29.6
Gogr	24.5	1.4	8.2	0.4	726.6	368.5	112.9	51.0	18.0	4.3	208.6	104.0	4.1	4.9	19.1	9.2
Gopa	17.7	6.0	8.7	0.4	699.2	554.8	280.6	282.9	98.7	116.4	657.5	643.6	24.6	25.8	28.6	24.1
Gotr	22.9	3.0	8.5	0.3	499.5	193.9	103.6	25.8	19.7	2.5	262.8	142.2	2.7	4.4	6.7	10.9
Gyat	16.7	5.4	8.3	0.3	394.2	137.2	45.6	7.1	12.2	2.8	835.6	261.6	7.8	3.5	1.0	13.9
Meli	19.8	7.2	8.3	0.7	273.0	127.5	53.6	33.6	14.3	3.2	277.9	327.1	6.2	4.1	8.0	5.1
Meva	26.5	9.0	8.2	0.2	174.3	117.5	104.9	42.7	11.0	2.7	216.5	504.5	4.9	4.5	45.6	36.3

Table 6.8 Continue

Codes	Temperature °C		pH		Conductivity $\mu\text{S cm}^{-1}$		TP $\mu\text{g l}^{-1}$		P-PO ₄ mg l^{-1}		N-NO ₃ mg l^{-1}		N-NO ₂ mg l^{-1}		TOC mg l^{-1}	
	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k
Megl	23.9	2.0	8.2	0.5	408.7	976.4	189.8	173.5	33.6	81.5	153.9	47.8	7.0	2.6	22.9	22.5
Miae	27.8	3.5	8.3	0.4	240.3	265.8	129.7	105.6	22.6	52.9	136.4	166.7	4.4	3.8	40.1	28.4
Mifl	20.3	1.1	8.3	0.3	356.9	3.7	67.0	31.8	15.1	3.3	228.8	29.5	27.7	13.6	60.7	18.1
Moar	23.3	1.8	8.5	0.3	470.4	268.4	57.9	48.0	14.3	4.2	126.1	86.1	8.3	7.7	14.6	5.3
Moco	24.4	1.2	8.3	0.4	667.8	497.9	103.6	56.6	17.5	3.5	182.2	133.6	3.0	0.7	22.7	10.1
Mobo	24.3	1.2	8.6	0.3	399.7	220.4	51.1	39.7	13.0	3.9	99.0	71.9	4.3	4.0	12.5	6.6
Mopa	22.6	1.8	8.6	0.3	351.1	24.7	41.5	14.9	12.6	3.2	102.5	75.0	11.2	11.5	19.2	20.9
Naca	29.1	4.4	8.6	0.2	468.7	199.7	42.0	32.3	11.8	3.2	56.8	57.9	5.0	1.0	10.8	17.1
Nacr	22.3	4.0	8.9	0.5	654.2	759.1	463.8	254.1	193.7	117.5	487.1	256.6	14.8	18.7	11.0	14.0
Nacry	24.2	3.9	8.8	0.3	319.0	116.7	97.3	25.4	19.5	3.7	181.6	74.7	2.2	1.2	9.5	7.5
Nala	17.1	5.5	9.4	1.0	338.0	222.1	842.0	572.3	324.6	240.2	188.6	88.3	89.3	66.9	7.2	9.2
Naph	23.7	1.2	8.8	0.1	365.2	183.9	43.4	35.0	13.4	4.9	215.9	123.0	26.3	27.6	7.4	2.6
Nara	24.5	1.4	8.2	0.4	759.6	427.7	117.8	52.2	18.4	4.0	209.2	107.6	3.8	4.7	19.3	7.7
Narad	21.0	5.9	8.5	0.2	698.4	1183.8	225.8	283.8	79.9	108.1	283.2	302.0	4.6	5.7	14.5	21.1
Nasl	18.7	9.1	8.7	0.3	863.4	584.5	288.7	267.5	104.1	110.0	756.1	765.4	23.3	16.8	41.1	23.7
Natr	21.3	2.6	8.5	0.3	392.0	149.7	42.0	10.3	12.2	2.6	190.8	193.8	10.1	9.0	7.5	6.9
Natry	27.7	5.8	9.2	0.4	232.4	19.9	100.6	23.9	20.0	2.4	144.3	179.2	2.1	2.8	2.4	18.1
Nian	23.2	0.0	8.8	0.0	320.1	34.1	99.8	17.8	20.0	2.5	198.3	40.5	2.0	0.2	10.7	15.3
Nifl	24.7	2.5	8.3	0.3	381.3	188.0	117.3	48.1	19.2	4.6	248.9	183.9	3.0	2.1	23.3	9.8
Oopa	24.9	4.0	8.3	0.4	639.1	378.3	84.2	67.9	17.2	2.8	180.7	91.9	3.8	0.8	14.1	6.6
Osli	23.8	2.3	8.5	0.4	322.0	314.0	140.0	17.8	20.9	3.3	165.8	133.3	2.9	2.2	28.5	10.8
Oste	24.7	4.5	8.3	0.5	441.6	219.9	146.9	210.8	27.0	64.3	291.7	111.8	3.9	12.6	20.1	10.0
Pamo	27.8	4.9	9.2	0.7	237.9	108.7	99.9	13.8	20.0	1.9	141.9	59.6	2.1	7.9	1.9	12.3
Pebo	30.9	2.3	8.5	0.2	587.9	54.1	41.8	7.2	12.1	2.8	186.6	223.9	6.5	2.3	5.5	2.4
Pedu	27.5	5.8	8.4	0.3	335.9	236.5	133.5	203.7	28.8	62.2	110.1	121.9	8.8	10.7	25.2	26.2

Table 6.8 Continue

Codes	Temperature °C		pH		Conductivity $\mu\text{S cm}^{-1}$		TP $\mu\text{g l}^{-1}$		P-PO ₄ mg l^{-1}		N-NO ₃ mg l^{-1}		N-NO ₂ mg l^{-1}		TOC mg l^{-1}	
	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k	u_k	t_k
Pecu	24.3	5.9	8.5	0.3	1406.3	1202.3	318.0	271.8	109.1	110.5	213.0	87.2	6.0	11.9	27.2	20.8
Peci	25.4	1.6	8.3	0.3	521.0	199.7	123.1	21.4	20.8	1.0	273.5	116.3	2.4	0.4	19.0	9.5
Pewi	24.8	2.4	8.2	0.4	821.8	393.6	124.2	46.0	18.5	4.6	226.0	105.7	3.6	3.8	19.8	17.9
Phca	22.0	1.0	8.7	0.2	327.9	32.8	473.0	332.8	179.9	130.7	395.2	100.9	30.9	9.1	8.0	27.8
Phlo	25.9	3.8	8.5	0.3	526.9	246.2	68.2	55.4	15.9	3.3	193.0	124.3	5.4	2.8	14.5	17.5
Psca	22.6	7.2	8.6	0.4	312.6	157.3	57.8	35.3	14.0	4.5	357.3	324.8	13.7	17.0	5.8	5.0
Rhgi	24.7	1.2	8.2	0.3	540.9	254.5	114.0	45.3	19.0	4.5	255.0	112.1	3.6	4.4	19.6	4.5
Scco	23.2	5.4	8.4	0.3	448.3	382.6	132.9	208.0	41.4	83.6	231.6	186.8	13.1	15.1	16.3	17.7
Scdi	23.1	5.8	8.5	0.4	384.7	144.2	99.6	161.6	30.5	63.2	253.3	228.2	8.6	9.3	7.7	5.4
Scel	31.0	3.3	8.5	0.3	582.7	75.0	47.9	104.6	14.7	46.5	144.0	231.2	6.5	7.4	5.7	2.1
Scfa	23.9	6.6	8.2	0.1	601.6	262.6	106.4	46.3	18.0	3.5	288.9	101.6	4.5	3.5	12.7	9.7
Scob	22.7	6.9	8.1	0.4	725.7	483.5	123.9	37.7	17.9	5.3	263.7	151.5	4.5	5.4	16.2	7.3
Spwe	24.5	5.9	8.1	0.3	227.6	462.9	168.2	138.3	24.6	67.8	205.5	471.5	8.3	11.0	26.1	27.6
Stci	28.6	4.8	8.5	0.2	515.6	133.4	40.9	16.3	12.3	2.8	144.0	163.8	6.9	4.4	7.5	4.1
Stcr	27.0	10.7	8.3	0.4	212.9	475.4	171.3	268.6	35.9	119.5	148.8	526.0	6.4	14.9	52.8	27.3
Stan	15.3	3.5	7.5	0.8	267.4	363.6	69.9	21.2	11.4	2.8	476.6	7.1	14.0	6.4	8.7	8.2
Stno	20.4	1.8	8.1	0.1	525.6	75.0	42.2	7.1	10.9	2.8	594.4	113.1	8.3	10.6	1.0	13.9
Temi	16.1	8.7	8.1	0.6	799.9	1125.3	784.9	436.4	324.2	217.0	601.6	460.5	31.1	27.8	17.1	15.2
Trce	26.6	3.4	8.4	0.3	597.6	273.0	67.5	52.5	15.5	3.5	174.3	122.4	4.5	1.5	11.2	6.4
Trps	14.2	3.2	8.3	0.1	305.4	105.5	99.9	61.0	16.3	5.1	275.9	210.9	5.3	5.8	1.5	0.9
Trsc	25.8	4.6	8.2	0.2	189.2	110.1	118.6	39.1	12.6	3.6	123.5	75.7	9.1	11.0	45.9	24.7
Ulsu	23.3	0.1	8.7	0.0	179.0	273.0	105.1	70.7	18.6	5.7	76.7	31.8	2.4	0.1	31.7	1.1
Zype	20.7	1.2	8.8	0.3	348.6	4.8	149.4	67.8	20.5	4.2	191.9	34.4	10.3	12.6	56.3	13.1
	RMSE	3.15		0.32		271.98		140.76		60.92		217.05		12.01		9.80
	R ²	0.70		0.44		0.47		0.72		0.72		0.45		0.60		0.63

6.4 Discussion

6.4.1 Predictor variables

Water quality of a water body is characterized by the physical and chemical limnology (Sidnei et al., 1992; Mustapha, 2008). Water quality studies in various regions have demonstrated that climate and hydrology are dominant factors influencing lake functions and processes, such as nutrients, algal biomass and water clarity, nutrient relations and the mixing regime (Thornton and Rast, 1993; Lebo et al., 1994; Huzar et al., 2006; Lee et al., 2010). Variations in physical characteristics like temperature, transparency, and chemical elements of water provide valuable information on the water quality (Sidnei et al., 1992; Mustapha, 2008). Phytoplankton response directly to pressure through changes in its abundance and assemblage (Reynolds et al., 2002; Marchetto et al., 2009). The physical organization of the habitat and the presence of nutrients are, directly or indirectly, the principal factors which could influence the original phytoplankton community (Reynolds, 1980; Becker et al., 2010). Nutrient (i.e. mainly phosphorus (P) and nitrogen (N)) enrichment has resulted in strong deterioration of ecological dynamic and function of freshwater especially lakes worldwide (Smith, 2003; Spears et al., 2016).

In the present study nutrients in 2014 were mainly higher than those found in 2015 when the precipitation increased. In fact, it has been reported the effect of precipitation on the water quality of surface watercourses. In this order, Salmaso et al. (1998) attributed partially the maximum water input and power production of Lake S. Croce (Italy) to precipitation in autumn. Hooker and Hernandez (1991) and Costa and Silva (1995) supported that precipitation induce modifications in the physical and chemical characteristics of the water. TP variable was mainly high in the studied water bodies during the study period. The highest value of TP ($420.8 \mu\text{g L}^{-1}$) was found in Osmankalfalar reservoir (Table 3.3). This was probably the consequence of excessive fish farming and water seasonal level fluctuations (from 15 m to 3 m) due to irrigation. Sevindik et al. (2017) indicated that the seasonal difference in water inflow and the summer use of water for irrigation purposes

caused the water level to decrease by 10m in İkizcetepeler reservoir (Turkey). According to Sgro et al. (2006), local factors may be more important determinants of water quality and increase variability among sites during low flow periods (typically warm seasons). In the same idea, Kennedy and Walker (1990) indicated that maximum environmental gradients are expected in reservoirs with high water retention time, high sedimentation rates and flow that are controlled by the water transport processes. With regard to lakes, Lake Avlan had higher TP value ($365 \mu\text{g L}^{-1}$) than other lakes. This lake was a very shallow and eutrophic ecosystem invaded by macrophytes. The highest conductivity, TN, TOC, N-NO_3 , BOD_5 and salinity were obtained in Lake Gölhisar, a shallow lake with dense vegetation and turbid water. The lake is near to the District of Gölhisar and surrounded by several villages. This result could be the consequences of anthropogenic activities especially discharge of domestic waste and agriculture fertilizers which can affect the lake through generated organic nutrients. Mancini and Arcà (2000) reported that urbanization and agriculture practices in a study area could affect the basins of watercourses. High TN values were measured in Lake Gölhisar ($1569.3 \mu\text{g L}^{-1}$) and Osmankalfalar reservoir ($1073.8 \mu\text{g L}^{-1}$). These results could be partially due to human activities in the water bodies' catchments. In accordance with our result, Strebel et al. (1989) and Almasri and Kaluarachchi (2004) indicated that the downstream increase of nitrate is a common human impact by agriculture. Loehr (1974) and Dodds (2002) indicated that human activities that lead to cultural eutrophication include the use of agricultural fertilizers, livestock practices, and release of nutrient-rich sewage into the surface water. Besides, Lake Gölhisar has a long history and could be affected by natural eutrophication over years. Dodds (2002) stated that the filling of lakes with sediments is a natural process because lakes are depressions in the watershed that collect sediment over time. The water transparency was low and ranged between 0.32 m in Lake Yazır and 3.43 m in Yapraklı Reservoir. Lake Yazır was an eutrophic shallow (1 m depth) lake, invaded by macrophytes and surrounded by vegetation. However, this lake is located at the high altitude (1481 m) with the absence of human settlements and agricultural activities in its catchment. This could be due to its long history time and could be affected by natural eutrophication. Yapraklı was a deep (52 m) and newly constructed reservoir and had relatively low nutrients due to the absence of human settlements and agricultural practices in its catchment.

6.4.2 Phytoplankton composition and limnoecology

The phytoplankton showed temporally and spatially distribution in lakes and reservoirs in the West Mediterranean basin during the study period. Bacillariophyta was the most dominant phylum in 2014, while Dinophyceae of the Miozoa dominated the phytoplankton composition in 2015. This change could be due to decreased nutrient concentrations in 2015 with the increase of precipitations in this year. Besides, previous limnoecological studies (Van-den-Hoek et al., 1995; Oda and Bicudo, 2006) reported that dinoflagellates may persist in the ecosystem even when that element is no longer available due to their capability of surviving with low P consumption. During the present study, commonly found species (e.g. *P. cinctum*, *C. ocellata*, *C. iris*, *C. excisa*, *C. hirundinella*, *S. communis*, *C. ovata*, *M. aeruginosa*, *N. lanceolata*, and *F. capucina*) were also widespread on the Earth (Padisák et al., 2003; Xiao et al., 2011; Çelekli and Öztürk, 2014).

In term of functional groups, fifteen phytoplankton groups as descriptors were evaluated in the lentic ecosystems of the West Mediterranean basin of Turkey. Functional groups approach has recently been used in lakes ecological status assessment. An advantage of functional groups is that ecological features are linked with the trophic state or habitat preferences (Salmaso et al., 2015). Functional approaches such as functional groups (FG), Morpho-Functional Groups (MFG) and the Traditional Taxonomic Size Spectrum (TTSS) were included in studies aiming at assessing ecological status (Salmaso et al., 2015). With regard to the functional groups in the present study, the dinoflagellate was mainly characterized by *P. cinctum* of functional group **L₀** in Toptaş reservoir (171.7 µg L⁻¹ TP) and Lake Yazır (150.0 µg L⁻¹ TP) very shallow and eutrophic lake. Relatively, these high TP values for freshwater ecosystems were found to be lower compared to the others studied water bodies. WA result (Table 3.9) also indicated that this species preferred 123.1 µg L⁻¹ TP and warm waters during the study. Reynolds et al. (2002) and Padisák et al. (2009) stated that the habitat template of this groups is deep and shallow, oligo to eutrophic, medium to large lakes.

With regard to diatoms, *Cyclotella* species of functional group **B**, *C. ocellata* in Yapraklı reservoir in fall 2014 and *C. iris* in Yapraklı and Çayboğazı reservoirs in summer 2014 were predominant, when the nutrients values were relatively low. WA

results (Table 6.8) indicated that *C. ocellata* preferred low TP ($61.0 \mu\text{g L}^{-1}$). This functional group has previously indicated as dominant in several Mediterranean fresh watercourses (Moreno-Ostos et al., 2008, Hoyer et al., 2009; Molina-Navarro et al., 2012; Çelekli and Öztürk, 2014). Besides, *Cyclotella* taxa were largely recorded in various trophic state aquatic ecosystems such as oligo-mesotrophic hard-water reservoirs (Dasí et al., 1998), Alleben Reservoir (Çelekli and Öztürk, 2014), Lake Vättern (Willén, 2001), Lake Skadar (Rakočević, 2012), Paraja limno-Reservoir in Spain (Molina-Navarro et al., 2014) and Mediterranean Marathonas Reservoir (Katsiapi et al., 2011). *Cyclotella* species are frequently associated with nutrient poor lakes (Bennion, 1994; Wunsam and Schmidt, 1995; Carvalho et al., 2013). The habitat of this group is mesotrophic small and medium sized lakes with species sensitive to the onset of stratification (Reynolds et al., 2002, Padisák et al., 2009), in agreement of the characteristic of Yapraklı and Çayboğazi reservoirs.

The cyanobacteria species *M. aeruginosa*, functional group **M** (eutrophic to hypertrophic, small- to medium-sized water bodies) was dominant in Osmankalfalar reservoir, Lake Avlan, and Lake Gölhisar in summer 2014. All these sites were classified as hypertrophic based on TP. WA result showed that *M. aeruginosa* preferred $129.7 \mu\text{g L}^{-1}$ TP value. Besides, it is known that *M. aeruginosa* is bioindicator of eutrophication phenomena (Padisák et al., 1999; Kurmayer et al., 2003; Cook et al. 2004; Moustaka-Gouni et al., 2007; Katsiapi et al., 2011). *M. aeruginosa* commonly inhabits freshwater lakes during eutrophic periods (Davis et al., 2009; Lehman et al., 2013). This species was found respectively in Lake Mälaren, Sweden by Willén (2001), Lake Sumin (Poland) by Pasztaleniec and Poniewozik (2010) and Marathonas Reservoir in Greece (Katsiapi et al., 2011). The presence of these species in our studied stations could be associated with pressure (i.e. eutrophication) due to fish farming, the release of nutrient-rich sewage into water bodies and land use.

Relating to cryptophytes, *C. ovata* of group **Y** (usually small and enriched lakes) was the most dominant species in Lake Gölhisar with BOD_5 (26.8 mg L^{-1}) and NO_3 (587.5 mg L^{-1}), highest values recorded in this study and in Lakes Yazır which showed mainly high BOD_5 (12.8 mg L^{-1}) value. The WA result assigned respectively $752 \mu\text{g L}^{-1}$ and $825 \mu\text{g L}^{-1}$ values of TP and TN to this species respectively. The presence of this species in these sites could indicate the water quality deterioration

which could be the result of anthropogenic activities. Çelekli and Öztürk (2014) associated a high biovolume to *C. ovata* during the stratification period in Alleben Reservoir. Becker et al. (2010) argued that Y functional group has adapted to low light intensity in a Mediterranean reservoir. According to Sommer (1981), Huszar et al. (2000) and Padisák et al. (2009) *Cryptomonas* spp. of codon Y which are known for their rapid phosphorus uptake rates and relatively fast growth, are commonly found in low nutrient enriched aquatic ecosystems.

The Euglenoid species, *E. arcus* of the codon W1 predominated the phytoplankton composition of Osmankalfalar reservoir and Lake Gölhisar during summer 2014. These sites showed 1670 $\mu\text{g L}^{-1}$ and 540 $\mu\text{g L}^{-1}$ TP in summer 2014, respectively in this period (Table not shown), while WA result associated 212.7 $\mu\text{g L}^{-1}$ TP and 636.3 $\mu\text{g L}^{-1}$ TN values for this species. The habitat template of this group has been indicated by Reynolds et al. (2002) and Padisák et al. (2009) as ponds, even temporary, rich inorganic matter from husbandry or sewage. This situation could be similar to the conditions of Osmankalfalar reservoir and Lake Gölhisar due to their proximity to human settlements and agriculture activities in their catchments. In accordance with our finding, *E. arcus* was listed in Lake Mogan (Turkey) polluted by industrial and municipal wastes from the increased human population (Demir et al., 2014).

The interaction between predictor factors and the phytoplankton taxa in the lakes and reservoirs was elucidated by use of multivariate approaches. The results of the WA (Table 6.8) indicated that *C. ovata*, *N. lanceolata* and *C. rostratiformis* preferred high nutrient concentrations (PO_4 , TP, NO_2 , and NO_3), confirmed by CCA ordination (Figure 6.10). *C. rostratiformis* and *C. ovata* were assigned in the Y functional group (usually small and enriched lakes) (Reynolds et al., 2002; Padisák et al., 2009). *N. lanceolata* has been reported as tolerant to organic/eutrophic conditions (Van Dam et al., 1994; Delgado and Pardo, 2014). According to Hlúbíková et al. (2007), the natural increase in nutrient concentration and organic compounds could lead to the appearance of pollution-tolerant species like *N. lanceolata*. Results of WA (Table 6.8) showed that *A. catenula*, *C. punctulatum*, *S. crenulatum*, *M. varians*, and *M. aeruginosa* preferred warm waters, also clearly supported by CCA (Figure 6.10). *M. aeruginosa* was reported by Katsiapi et al. (2011) to the dominant species the cyanobacterial biovolume when water temperature was high. The dominance of

bloom-forming cyanobacteria including *A. catenula* and *M. aeruginosa* has been related to high temperature (Caraco and Miller, 1998; Becker et al., 2009). With regard to WA, *C. lapponica*, *C. iris*, *E. sorex* and *E. proxima* were associated with alkaline waters, while *D. tenuis*, *F. incognita*, *E. turgida* and *N. slesvicensis* were associated with high conductivity. These species-environmental relationships were found in the ordination (Figure 6.10). *E. turgida* has been indicated with relatively high conductivity by Potapova and Charles (2003).

Multivariate analyses indicated that environmental factors strongly drive phytoplankton assemblages which were found to be dominant at their optimum conditions. This is in accordance with our hypothesis (II) that environmental variables especially pollution parameters (nutrients, organic matter) are the most important structuring factors of phytoplankton composition and confirmed that phytoplankton and adapted biotic indices would be suitable to use as indicators of anthropogenic pressure in lakes and reservoirs. A strong relationship between phytoplankton assemblages and environmental variables has been noted by several previous studies (e.g. Naselli-Flores, 2000; Crossetti and Bicudo, 2005; Romo and Villena, 2005; Porter et al., 2008; Becker et al., 2010; Abonyi et al., 2012; López et al., 2012; Yerli et al., 2012; Moreti et al., 2013; Devercelli and O'Farrell, 2013; Çelekli and Öztürk, 2014). According to Pasztaleniec and Poniewozik (2010), the quality and quantity of the phytoplankton depend in part on the nutrient load. In the present study, the dominant species changed spatial and temporal. Reynolds et al. (2002), Reynolds (2006), and Padisák et al. (2009) reported that the dominant phytoplankton taxa differed among water bodies and seasons due to the difference of species preference in tolerance ranges of water temperature, light, and nutrient limitations. Wherever the phytoplankton assemblage is primarily phosphorus limited, decreases in annual average total phosphorus (TP) contents should induce a decrease in phytoplankton composition, biomass, in contrast to an increase in water transparency and in the proportion and biodiversity of aquatic secondary producers (Jeppesen et al., 2000; Spears et al., 2016). The ecology of phytoplankton reservoir communities plays a pivotal role in their management and in the development of inland fisheries in the water scarce semi-arid regions of Rio Grandedo Norte State, Brazil (Chellappa et al., 2009). The similarity of our results with previously early studies could be explained by the Mediterranean region climate, geographical

position and the similarity in physicochemical variables and the mixing regime supported by human activities in the watersheds.

6.4.3 Seasonality of phytoplankton

Seasonal influences are strong forces in phytoplankton assemblages' succession. Seasonal phytoplankton dynamics are controlled by physicochemical forces and biotic interactions (Reynolds, 2006; Anneville et al., 2002; Chen et al., 2009). The seasonal steps in algal size mainly depends on nutrient and temperature scales (Morabito et al., 2007; Reynolds, 2006; Chen et al., 2011). Phytoplankton react spontaneously to nutrient concentrations through modifications in its structure, biomass and composition (Reynolds et al., 2002; Marchetto et al., 2009). The peaks of phytoplankton biovolume were observed when the water temperature and the nutrients values especially TP and nitrate were optimum (Nöges et al., 2003). In shallow lakes, the impact of temperature is especially adverse when it synchronizes with low water quantity, and the degradation of water quality generally occurs in the warm season (Nöges et al., 2003; Rakočević, 2012). The physical structure of the environment and the presence of nutrients are, spontaneously or indirectly, the great major parameters that affect the pristine phytoplankton community (Reynolds, 1980; Reynolds, 2006; Becker et al., 2010).

The spatial and temporal occurrences of phytoplankton species were strictly associated with the changes of some important environmental factors in lakes and reservoirs in the Western Mediterranean basin of Turkey during the study seasons. The species number changed not only per site but changed also per season. the species number changed from a minimum of 10 at Çavdır reservoir in summer 2014 to a maximum of 49 at Lake Avlan in summer 2015. This species number was generally higher in 2015 than those of 2014. This situation was in accordance with previous study indicated that the peaks of phytoplankton biovolume were observed when the water temperature was optimum (Nöges et al., 2003). The Phytoplankton biovolume per phylum changed during the study period according to the seasonal changes in water characteristics. Peaks of phytoplankton abundance did not appear at the same time at the different lakes and reservoirs, and phytoplankton succession showed distinct spatial heterogeneity during the study in the lakes and reservoirs. The Figure 6.9a,b and c show the seasonal variations in the phytoplankton phyla'

total biovolume ($\times 10^4 \mu\text{m}^3 \cdot \text{L}^{-1}$) in the lakes and reservoirs. The seasonal phytoplankton composition showed a marked change in abundance among studied water bodies. The mainly abundant phylum was Bacillariophyta followed by the Chlorophyta phylum. This situation is in agreement with previous studies in Valparáiso Reservoir (Spain) (Negro et al., 2000) in Lakes Abant and Gököy (Çelekli, 2006) and in Alleben Reservoir (Çelekli and Öztürk, 2014). The phytoplankton composition observed had mainly similarities to those reported for others Mediterranean shallow lakes such as Lake Skadar in Montenegro (Rakočević, 2012), Alleben Reservoir (Çelekli and Öztürk, 2014) and Lake Mogan in Turkey (Demir et al., 2014) and Estonia, Czech Germany and Greece lakes (Erdoğan, 2016).



CHAPTER VII

BIOASSESSMENT OF THE LENTIC ECOSYSTEMS BASED ON PHYTOPLANKTON

ABSTRACT

Lentic freshwaters (especially lakes and reservoirs) are complex aquatic ecosystems which evolution is governed by several natural and anthropogenic factors. The impact of these factors on the ecological state of lakes and reservoirs are often very variable from one environment to another. Aquatic organisms which are sensitive to the changes of their environment could be suitable tool to provide information about the quality state of the ecosystems in which they are found. Due to this reason, Lakes and reservoirs should be assessed at a local, regional and national level not using only physicochemical parameters, but also biological quality elements. Phytoplankton, as one of these biological quality elements are useful tool for the bioassessment surface waters. The aim of this work was to assess the water quality of three lakes and six reservoirs in the Western Mediterranean Basin (Turkey) using phytoplankton indices. Data were sampled seasonally from summer 2014 to summer 2015.

With regard to the water quality, the phytoplankton trophic index (PTI) values varied from 1.96 to 2.48; the Mediterranean phytoplankton trophic index (Med-PTI) ranged from 0.8 to 2.05, while the assemblage index (Q) values were between 3.22 and 2.10. Ecological quality ratio base on PTI ranged between 0.69 and 0.84, while it changed from 0.65 to 0.85 based on Med-PTI. Lakes and reservoirs showed similar ecological status ranged from moderate to high quality according to the three indices.

7.1 Introduction

Lentic fresh waters (especially lakes and reservoirs) are complex aquatic ecosystems whose evolution is governed by several natural and anthropogenic factors. The impact of these factors on the ecological state of lakes and reservoirs are often very variable from one environment to another. The assessment of lakes and reservoirs water quality is based mainly on the major issues associated with these ecosystems. Eutrophication of lakes and reservoirs resulting in algae blooms has receiving more attention during the last centuries. Nutrient enrichment of aquatic systems due to soil erosion, runoff fertilization of croplands and livestock production, and discharges from sewage treatment plants is considered to be one of the major causes of environmental degradation in the last decades. Developing good indicators of anthropogenic impact on aquatic ecosystems related to different purposes become major challenge that could be resolved by the establishment of monitoring programs. Aquatic organisms which are sensitive to the changes of their environment could be suitable tool to provide information about the quality state of the ecosystems in which they are found. Due to this reason, Lakes and reservoirs should be assessed at a local, regional and national level not using only physicochemical parameters, but also biological quality elements. In this regard, the European Union water framework directive (WFD) recommended recently phytoplankton, phytobenthos, macrophytes, macroinvertebrates, and fish as bioindicators to be applied for surface water assessment (Directive, 2000; EC, 2009). However, the value of phytoplankton (algae) as biomonitors for fresh waters has already been recognized in the mid-19th century (Cohn, 1853; Dokulil, 2003). The first attempt to classify aquatic organisms as indicators of water quality was made by Cohn (1870), later modified by Mez (1898), while the relation of organisms to their water quality was more clearly defined by Kolwitz and Marson (1902, 1908, 1909) by creating the term “Saprobic organisms” (Dokulil 2003). According to WFD, watercourses with ecological status classified as moderate to bad should be restored by actions taken in and around the watercourse itself as well as its watershed. The efficiency of any restoring action should be surveyed by examining the amelioration in water quality. To reduce monitoring costs during restoring, the WFD requires that only the most sensitive bioindicators to environmental changes in watercourses should be monitored (EC, 2009, Marchetto et al., 2009; Phillips et al., 2013). If the

major anthropogenic constraint is the load of nutrients, phytoplankton could be the natural choice for biomonitoring. Phytoplankton response quickly to nutrient variations through changes in its abundance and composition (Reynolds, 1984; Rott, 1984; Naselli Flores and Barone, 1998; Reynolds et al., 2002; Schaumburg et al., 2004; Reynolds, 2006; Poikane et al., 2011; Cellamare et al., 2012).

Phytoplankton could be used as a good bioindicator of water quality due to its sensitivity and ability to response in the surrounding environment changes (Reynolds, 1984; Padisák et al., 2003; EC, 2009; Padisák et al. 2006; Poikane et al., 2011; Katsiapi, et al., 2011).

In last decades, a number of phytoplankton metrics and systems based its biovolume and abundance have been developed in order to evaluate ecological quality of aquatic ecosystems especially in lakes and reservoirs: e.g. Brettum index (Brettum, 1989) , Catalán Index (Catalan et al., 2003), ITP or Barbe Index (Philippe et al., 2003), Phytoplankton Trophic Lake Index (PTSI) (Riedmüller et al., 2006), assemblage index (Q) (Padisák et al., 2006), Phytoplankton Trophic Index (PTI) (Salmaso et al., 2006; Phillips et al., 2013), Mediterranean Phytoplankton Trophic Index (MedPTI) (Marchetto et al., 2009), Phytoplankton Metrics for Polish Lakes (PMPL) (Hutorowicz and Pasztaleniec, 2014), New Mediterranean Assessment System for Reservoir's Phytoplankton (NMASRP) (de Hoyos et al., 2014), Danish lake phytoplankton Index (DLPI) (Phillips et al., 2014).

One of the most recent phytoplankton research interests is the establishment of the so-called phytoplankton functional groups (Abonyi et al., 2012). Many limnological studies including the functional groups were carried out in many countries (e.g. Reynolds et al., 2002; Reynolds, 2006; Salmaso and Padisák, 2007; Moreno-Ostos et al., 2008; Padisák et al., 2003, 2006, 2009; Hoyer et al., 2009; Abonyi et al., 2012; Çelekli and Öztürk, 2014; Demir et al., 2014; Sevindik et al., 2017). The phytoplankton functional groups system, firstly proposed for lakes, is also a good tool for reservoirs, since the two ecosystems are widely similar because of their small spatial and temporal scales (Reynolds, 1999, 2002, 2006; Padisák et al., 2003, 2006, 2009; Becker et al., 2010).

Phytoplankton metrics have previously been used for biomonitoring of Turkish freshwater lakes and reservoirs: e.g. Ömerli reservoir (Albay and Akçaalan, 2003), Lakes Abant and Gök  y (  elekli, 2006), lake Manyas (  elekli and Ongun, 2008), Alleben Reservoir (  elekli and   zt  rk, 2014), lakes Mogan, Abant, Karag  l and Poyrazlar (Atıcı and Tokatli, 2014), Lake Mogan (Demir et al., 2014), 47 shallow lakes (Erdoğan, 2016), and İ kizcetepeler reservoir (Sevindik et al., 2017). With regard to basin approach, this work is the first test to monitor lakes and reservoirs in the Western Mediterranean Basin combining three phytoplankton metrics. This chapter aims to i) assess the ecological status of three lakes and six reservoirs in the Western Mediterranean basin by using phytoplankton trophic index (PTI), Mediterranean phytoplankton trophic index (Med-PTI) and the Q assemblage index.

7.2 Material and Methods

From summer 2014 to Summer 2015, we sampled three lakes and six reservoirs and collected phytoplankton data which were served to calculate indices scores for the water quality assessment.

7.2.1 Phytoplankton trophic index (PTI)

The PTI value was determined for each of the lake and reservoir based on the weighted average of the optima from the taxa present in the lakes and reservoirs, with the proportion of total biovolume as weights according to the formula (Eq.5) proposed by Philips et al. (2013) and modified by Çelekli et al. (2016).

$$PTI = \frac{\sum_{j=1}^n a_j * s_j * i_j}{\sum_{j=1}^n a_j * i_j} \quad (\text{Eq. 5})$$

where a_j is proportion of j th taxon in the sample, s_j is the optimum of j th taxon, and i_j is the indicator value of j th taxon in the sample.

7.2.2 Mediterranean phytoplankton trophic index (MedPTI)

The Mediterranean phytoplankton trophic index originally proposed for the monitoring of deep reservoirs was used in this study to evaluate the water ecological status of both lakes and reservoirs. In this order, the trophic and indicator values of the species found in the water bodies were first determined. The value of the MedPTI index in the i -th reservoir or lake is then calculated using the equation (Eq.6) developed by Marchetto et al. (2009).

$$Med - PTI_i = \frac{\sum_{k=1}^m B_{j,k} * v_k * i_k}{\sum_{k=1}^m B_{j,k} * i_k} \quad (\text{Eq. 6})$$

where $(B_{j,k})$, is the biovolume, (v_k) is the trophic value and (i_k) is the indicator values of the m species found in that reservoir or lake.

7.2.3 Phytoplankton assemblage index (Q)

The phytoplankton assemblage index (Q) was calculated for the lakes and reservoir according to the formula (Eq.7) developed by Padisák et al. (2006).

$$Q = \sum_{j=1}^n p_i F \quad (\text{Eq. 7})$$

Where, p_i is the relative proportion ($p_i = n_i / N$; n_i is biovolume of the i^{th} functional group; N : total biovolume) of functional groups in total biovolume. F is a factor value determined for the i^{th} functional class in the given lake or reservoir.

The factor number (F) for each functional group was established considering what would be the reference state of the corresponding reservoir or lake and probable algal community presence in it. In this case, higher factor F scores were attributed to which would be the original communities on the reservoir or lake and lower scores to communities supposed to be undesirable in reference conditions. Q index boundaries and the classification classes are given in Appendix P.

7.2.4 Ecological quality ratios (EQR)

As recommended by the European Union water framework directive (WFD), PTI and Med-PTI were calculated and then transformed as ecological quality ratios (EQR) for ecological status classification. The EQR was used as a numerical score ranged from zero (0) to one (1) according to the WFD, where high water quality is indicated by scores close to one (1) and bad water quality by scores close to zero (0). PTI, Med-PTI class boundaries and EQRs are presented in Appendix A. EQR_{PTI} and EQR_{MedPTI} were calculated using equations (Eq.8) and (Eq.9) respectively.

$$EQR_{PTI} = \left(\frac{PTI_{Obs} - PTI_{Max}}{PTI_{Ref} - PTI_{Max}} \right) \quad (\text{Eq. 8})$$

where PTI_{Obs} is mean sample PTI for each lake and reservoirs, PTI_{Max} is the maximum PTI score for type, PTI_{Ref} is the reference PTI for type.

$$EQR_{MedPTI} = \left(\frac{4 - MedPTI_o}{4 - MedPTI_{ref}} \right) \quad (\text{Eq. 9})$$

Where, $MedPTI_o$ and $MedPTI_{ref}$ are the observed and reference MedPTI, respectively.

7.3 Results

The ecological characterization of the water bodies was estimated using PTI, Med-PTI, and the Q assemblage index.

7.3.1 Phytoplankton trophic index (PTI)

Values of phytoplankton trophic index (PTI) changed from 2.48 in Lake Gölhisar to 1.96 in Çavdır reservoir, while EQR_PTİ ranged between 0.69 and 0.84. EQR-PTI indicated that all studied lakes and reservoirs had moderate water, except Çayboğazı and Yapraklı reservoirs which showed a good water ecological status (Table 7.1).

Table 7.1 Corresponding Lakes and Reservoirs PTI, EQR values and ecological states

Sites		PTI	EQR_PTİ	Ecological status
Lakes	Gölhisar	2.48	0.75	Moderate
	Avlan	2.56	0.69	Moderate
	Yazır	2.54	0.73	Moderate
Reservoirs	Çayboğazı	2.43	0.71	Moderate
	Geyik	2.42	0.69	Moderate
	Çavdır	1.96	0.84	Good
	Toptaş	2.39	0.69	Moderate
	Yapraklı	2.36	0.78	Good
	Osmankalfalar	2.52	0.71	Moderate

Relationships between TP and PTI is shown in figure 7.1. The figure indicated that PTI had well regression with logTP and high determination of correlation value ($R^2 = 0.958$).

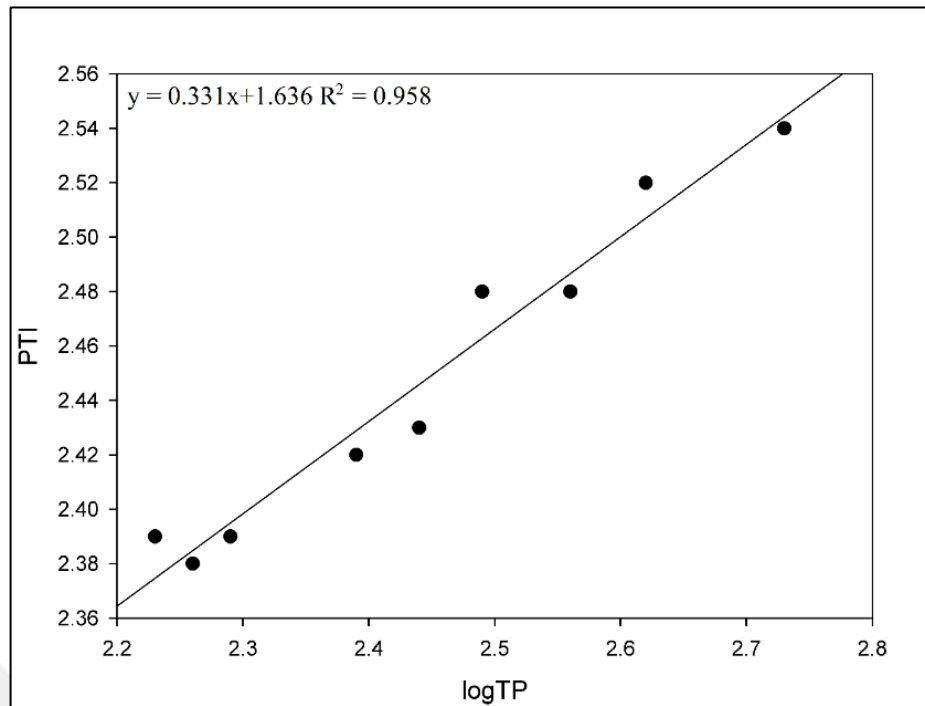


Figure 7.1 Plot of PIT versus TP

7.3.2 Mediterranean phytoplankton trophic index (Med-PTI)

The Mediterranean phytoplankton trophic index (Med-PTI) values ranged from 2.05 in Toptaş reservoir to 0.8 in Lake Yazır. According to these values, the EQR changed from 0.65 to 0.85. EQR-MedPTI showed that high, good, and medium states were found for Lake Gölhisar, Lake Avlan, and Lake Yazır, respectively. With regard to reservoirs, Çayboğazı, Çavdır, Toptaş, and Yapraklı reservoirs had medium status and others had good state (Table 7.2).

Table 7.2 Corresponding Lakes and Reservoirs Med-PTI, EQR values and ecological states

	Sites	Med-PTI	EQR_MedPTI	Status
Lakes	Gölhisar	1.54	0.84	High
	Avlan	1.69	0.76	Good
	Yazır	0.8	0.65	Moderate
Reservoirs	Çayboğazı	1.45	0.85	Good
	Geyik	1.48	0.69	Moderate
	Çavdır	1.81	0.79	Good
	Toptaş	2.05	0.81	Good
	Yapraklı	1.74	0.81	Good
	Osmankalfalar	1.34	0.65	Moderate

Relationship between total phosphorus (TP) and Med-PTI given in Figure 7.2. The figure indicated that Med-PTI showed well regression with logTP ($R^2 = 0.902$).

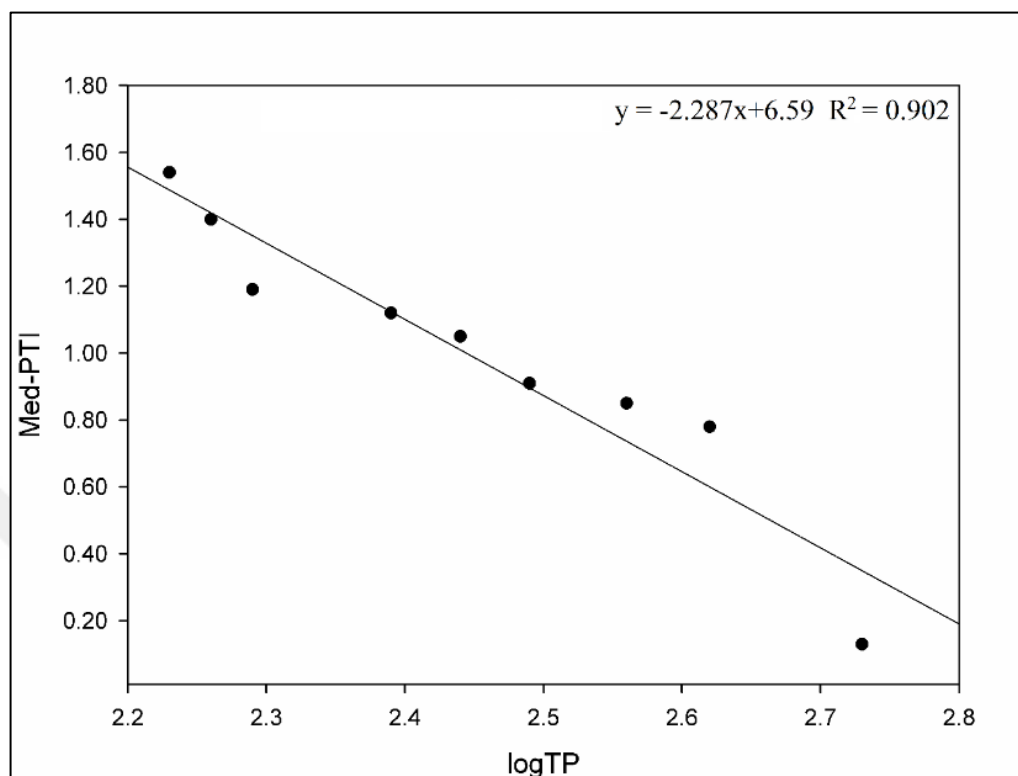


Figure 7.2 Plot of Med-PTI versus TP

7.3.3 Phytoplankton Assemblage Index (Q)

The highest Q index (3.22) was found in Lake Yazır, whereas it's lowest with 2.10 was determined in Yapraklı reservoir. Results of Q assemblage index indicated that all water bodies had medium status, except good state for Lake Yazır (Table 7.3). Relationships between total phosphorus (TP) and Q index is given in Figure 7.3. which indicated that all indices showed well regression with logTP ($R^2 = 0.5$). With Regard to the ecological status observed following the three indices, the variance analysis (one-way ANOVA) indicated a significant difference between the lakes and reservoir. (one-way ANOVA; $P < 0.05$; $F = 255.5$).

Table 7.3 Corresponding Lakes and Reservoirs Q values associated with EQR values and ecological states

	Sites	Q index	State
Lakes	Göhlisar	2.62	Medium
	Avlan	2.59	Medium
	Yazır	3.22	Good
Reservoirs	Çayboğazı	2.37	Medium
	Geyik	2.20	Medium
	Çavdır	2.64	Medium
	Toptaş	2.36	Medium
	Yapraklı	2.10	Medium
	Osmankalfalar	2.40	Medium

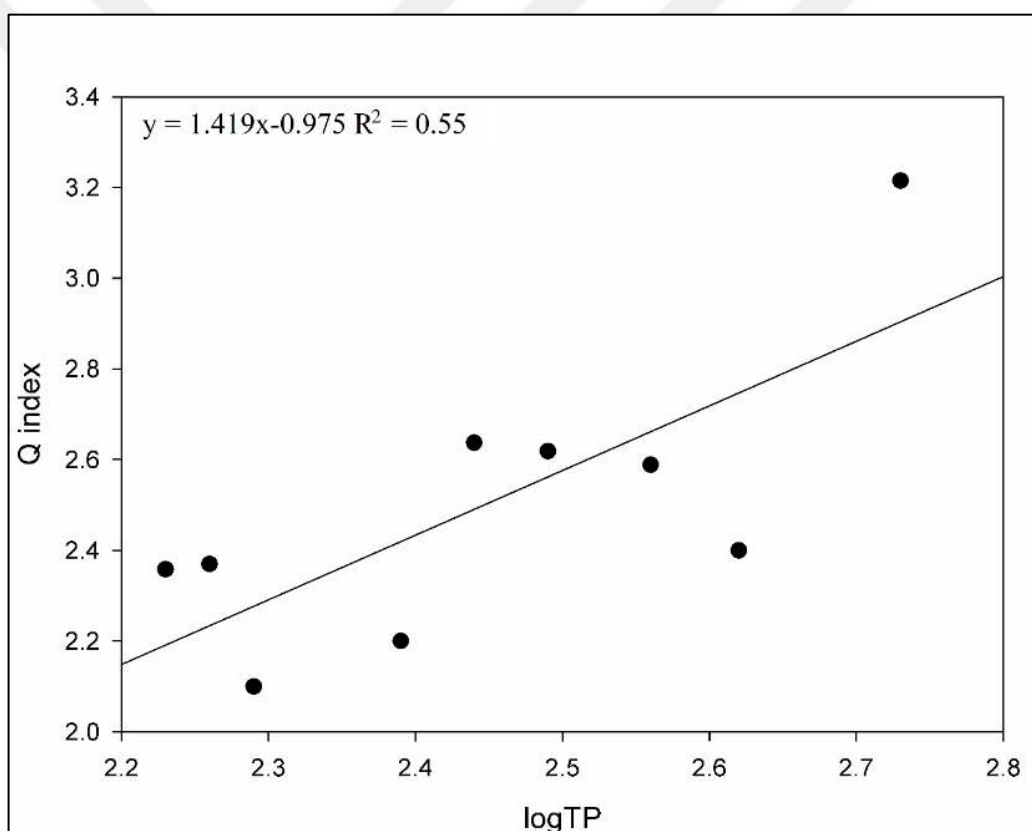


Figure 7.3 Plot of Q index versus TP

7.4 Discussion

Water quality, especially the ecological status of lakes and reservoirs ecosystems could be evaluated using phytoplankton based metrics (Directive, 2000; EC, 2009; Poikane et al., 2011; Phillips et al., 2013). PTI values in this present study changed from 1.96 in Çavdır reservoir to 2.48 in Lake Gölhisar. These values were higher than those found in Alleben Reservoir (Turkey) (Çelekli and Öztürk, 2014) and lower than the values found in Lakes Lugano and Maggiore which are deep lakes in Italy (Salmaso et al., 2006). Med-PTI ranged from 0.8 in Lake Yazır to 2.05 in Toptaş reservoir. These results were different from those found in deep Mediterranean reservoirs (Italy) (Marchetto et al., 2009). The Q index values which ranged from 2.10 in Yapraklı reservoir to 3.2 in Lake Yazır were higher than those found in a deep subtropical Faxinal Reservoir (Brazil) (Becker et al., 2009). But lower than those found in Spain in a deep Mediterranean reservoir, Sau Reservoir (Spain) (Becker et al., 2010).

With regard to ecological status based on EQR-PTI, moderate water quality was observed in all the lakes and some reservoirs during the present study. In accordance with our results, Çelekli and Öztürk (2014) previously reported a moderate ecological status for Alleben Reservoir in Turkey. The values of EQR-PTI indicated that Yapraklı and Çavdır reservoirs had good ecological status, while Çayboğazı, Toptaş, and Çavdır reservoirs were classified in this state by EQR-MedPTI. On the other hand, all reservoirs and lakes had medium ecological status based on Q assemblage index except Lake Yazır which showed good ecological state. The good ecological state obtained during the present study using EQR (based on PTI and Med-PTI) and Q assemblage index was indicated in Mediterranean reservoirs such as Sau Reservoir (Spain) (Becker et al., 2010) and Pareja limno-reservoir (Spain) (Molina-Navarro et al., 2014). The high ecological state in Lake Gölhisar associated to EQR-MedPTI is probably due to moderate human activities on and around this lake especially agriculture practices, fish farming, and organic pollutants input from wastewater which are great sources of organic and inorganic nutrients in freshwater. From this result, the method of Med-PTI firstly developed for evaluating the response of the phytoplankton to changes in nutrient concentration in deep reservoirs by Marchetto et al. (2009) could be reliable for assessing the ecological status of shallow reservoirs and lakes. Among these indices, PTI had more competitive with a

higher coefficient ($R^2 = 0.958$) than others, which was clearly indicated in Figure 3.13, 3.14 and 3.15. Besides, this significant positive relationship between PTI and TP was previously found in European freshwater bodies (Philips et al., 2013). The results of this study revealed that the application of phytoplankton characteristics could be a useful biomonitoring tool for assessing water quality in a basin approach.



CHAPTER VIII

TROPHIC STATES OF THE LAKES AND RESERVOIRS

ABSTRACT

Water quality is the definition of the state or condition of watercourses relative to the requirements of the aquatic living organisms and human needs. The trophic state of the lakes and reservoirs is generally measured using physicochemical and biological parameters including water transparency, total phosphorus, total nitrogen, and chlorophyll-a. The aim of this chapter was to examine the trophic state of 3 lakes and 6 reservoirs in the western Mediterranean basin of Turkey following the Carlson's Trophic State Index and the OECD trophic classification systems. Environmental data were collected during 4 seasons.

Total phosphorus concentrations and the Secchi depth values among stations and seasons. In summer 2014, the highest TP (540 $\mu\text{g/L}$) and SD (2.5 m) were recorded at Lake Gölhisar and Çayboğazı reservoir respectively. Çavdır reservoir showed the highest TP (750 $\mu\text{g/L}$) and SD (1.5 m) in fall 2014, while the lowest TP value, 28 $\mu\text{g/L}$ in summer 2014 was measured at Osmankalfalar reservoir. In spring 2015, the peaks of TP (170 $\mu\text{g/L}$) and SD (5 m) were recorded at Lake Yazır and Yapraklı reservoir respectively, while the lowest TP concentration (32 $\mu\text{g/L}$) in summer was recorded at Lake Avlan. With regard to water quality, 3 trophic categories were found based on Carlson's index, while 4 were found according to OECD criteria. Yapraklı reservoir was mesotrophic according to SD based on Carlson system, while other reservoirs and lakes showed eutrophic status according to TP and SD. OECD criteria classified lakes and reservoirs as hypertrophic based on TP. According to SD, all lakes had hypertrophic status, while Geyik, Toptaş, and Osmankalfalar reservoirs were eutrophic; Çayboğazı and Çavdır reservoirs were recorded as meso-eutrophic, while an oligotrophic status was found at Yapraklı reservoir.

8.1 Introduction

Water quality is characterized based on of the state or condition of water bodies which responses to the requirements of the aquatic organisms and human needs. It is measured as physical, chemical, biological and organoleptic (taste-related) properties of water (Johnson et al., 1997; UN, 2007; Ndungu, 2014). The water state can be affected naturally or through human activities by nutrient enrichment caused which stimulate primary producers. This phenomenon is known as eutrophication. The trophic state of lakes is indicative of their biological productivity, that is, the number of living organisms supported within environmental variables, primarily in the form of algae (Randolph and Wilh, 1984; Denys, 2004; Çelekli, 2006).

The expression eutrophication was wrongly used by limnologists and needs to be clearly defined. Hutchinson (1973) gave a clear history of the evolution and usage of the expression since Weber (1907) first used the expression to characterize the aspect of wetlands. The previously use of the expression was widely descriptive, based on the appearance of lakes, hypolimnetic oxygen reduction, and key taxa of attached macroinvertebrates. Unnecessary to indicate that several freshwater resources seemed to be not easy to classify using these methods. By the mid-1970s, the term eutrophication had appeared to have taken on new definitions in several cases. Thereby it became probable to estimate the primary productivity of water bodies spontaneously, and the principal impact of anthropogenic activities in water bodies' watershed and the concept of water turnover became known. The expression began then to be used in a large dynamic way, involving variations in both within reservoirs functions and landwater relationships (Schindler, 2006). During the later stages of eutrophication, the watercourse is stifled by excessive plant life because of higher concentrations of nutrients such as nitrogen and phosphorus. Anthropogenic activities such as urban construction, sewage discharges, agricultural practices, and industrial development could accelerate the phenomenon (MPCA, 2008).

To evaluate the eutrophication in aquatic systems, limnologists used to determine the relationships between water quality, nutrients, and phytoplankton communities and productivity for establishing a trophic characterization of water bodies. There are numerous systems to use water quality data which depend on mean objectives, the type of samples, and the size of the sampling area. Many studies related to

eutrophication in aquatic ecosystems have been done, as indicated previous methods established for categorization, modeling and interpretations of collected data (Simeonov et al., 2002; Boyacioglu and Boyacioglu, 2007, Alobaidy et al., 2010). One of the most important ways to transmit information on water quality changes is by the development of applicable metrics (Dwivedi and Pathak, 2007, Alobaidy et al., 2010). Metrics are developed using the scores of different physicochemical and biological variables in a water body. The use of metrics in monitoring systems to evaluate ecosystem state has the ability to transmit the general public and decision-makers about the evolution of the ecosystem (Nasirian, 2007; Simoes et al., 2008; Alobaidy et al., 2010). This approach could also help to establish a standart system for assessing successes and failures of management techniques for amelioring water quality (Rickwood and. Carr, 2009; Alobaidy et al., 2010). Water quality indices (WQI) have been applied to control water quality using physicochemical parameters since the 1970s. The first water quality index has been developed by Horton (1965), and since a huge interest has been given to the establishment of 'water quality index' systems with the idea of developing a tool to make easy the interpretation of water quality data (Liou et al., 2004; Boyacioglu 2007), and many water quality indices that have been developed to evaluate water quality in United States, Canada and Europe.

The trophic characterization based on the level of the productivity of the aquatic ecosystems is termed trophic states of the system. The trophic state of a lake is the biological productivity, what could be defined as the quantity of plants and animals in a lake which determined the water quality of aquatic ecosystems and served as water quality indicator (Ndungu, 2014). Freshwater ecosystems can be classified by their step of eutrophication. The trophic state of lakes and reservoirs is generally measured using physicochemical and biological parameters including water transparency (Secchi disk), total phosphorus, total nitrogen and chlorophyll-a. Four great trophic state classes including oligotrophic (few foods), eutrophic (many foods), mesotrophic which falls between these two firs categories, and the hypertrophic state were considered based on the trophic metrics. Transparency as a definition of water clarity can be used to combine an analytical orientation for a watercourse, the localization of transparency differences in a watercourse, and to compare various water resources. Water transparency shows the scale of aquatic

organisms activity and could be taken quickly by the use of a Secchi disk (Preisendorfer 1986; Zaneveld and Pegau, 2004). The Secchi depth of a given sampling stations, which shows the clarity of the water depends on to the level of dissolved solids in water, which in the end gives the quantity of biomass. High water transparency is measured when the occurrence of microalgae and quantity of suspended elements in water is lower. Transparency may also be impacted by the input of nutrients into the water body, especially phosphorus coming from sources such as plantations, wastewaters etc. (Çako et al., 2013) and also by suspended solids generated generally by activities such as urbanization near to aquatic ecosystems, agriculture and livestock rejections, and industrial sewage. Phosphorus is an essential nutrient for plant life and plays a key role for algae growth, it can be found naturally in the environment, but can be a source of pollution of water at a high rate. Anthropogenic activities, however, have resulted in excessive loading of phosphorus into several freshwater ecosystems. This could be a source of water pollution by promoting algae bloom, especially in lakes (MPCA, 2007). While there are a number of ways to measure phosphorus, total phosphorus (TP) provides a good overview of how much phosphorus is in the water as it includes both the dissolved fraction that is available for plant uptake, as well as any particulate phosphorus bound to suspended particles (BPMP, 2017).

Several studies have been carried out throughout Turkey for evaluating water quality of Lakes based on trophic systems: in Lake Manyas and Ömerli Reservoir (Albay and Akçaalan, 2003 a, b), Lake Burdur (Girgin et al., 2004), Manyas Lake (Karafistan and Arık-Çolakoğlu, 2005), in Yeniçağa Lake (Saygı-Başbuğ and Demirkalb, 2004), Lakes Abant and Gököy (Çelekli, 2006). Recently, Erdoğan (2016) studied the water quality of 47 shallow lakes including Hamam, Poyrazlar, Abant, Büyük, Derin, Nazlı, İnce, Serin, Pedina, Eymir, Mogan, Taşkısığı, Küçük Akgöl, Büyük Akgöl, Çubuk, Gölcük Bolu, Yeniçağa, Gölhisar, Mert, Erikli, Saka, Gebekirse, Barutçu, Karagöl İzmir, Gölcük Ödemiş, Emre, Gököl, Karagöl Denizli, Azap, Gölcük Sakarya, Yayla, Saklı, Baldımaz, Gıcı, Tatlı, Sarıkum, Kocagöl, Gerede, Keçi, Karagöl Bolu, Uyuz, Balıklı, Kaya, Eğri, Sarp, Kaz, and Seyfe Göleti using the Carlson's trophic state index (TSI). However, until the present study, there were few or no studies concerning our studied water bodies including lakes Gölhisar,

Avlan and Yazır, Çayboğazı, Geyik, Çavdır, Toptaş, Yapraklı and Osmankalfalar reservoirs.

The measurement of physicochemical and biological parameters such as nutrient contents, water transparency and the amount of chlorophyll a could provides great information for interpreting their relationship with the trophic state in freshwater ecosystems. In this chapter, the trophic states of lakes Gölhisar, Avlan and Yazır, Çayboğazı, Geyik, Çavdır, Toptaş, Yapraklı and Osmankalfalar reservoirs in the Western Mediterranean basin of Turkey were determined and compared based on their water transparency and the concentrations of total phosphorus.



8.2 Material and Methods

8.2.1 Study Area and Data Collection

During this study the trophic state of 3 lakes (Göhlisar, Yazır and Avlan), and 6 reservoirs (Çayboğazı, Geyik, Çavdır, Toptaş, Yapraklı, and Osmankalfalar) in the western Mediterranean basin of Turkey (Figure 8.1) were examined based on Trophic State Index (TSI) Carlson (1977) and Carlson and Simpson (1996) and the OECD trophic classification system (OECD, 1982). Environmental data such as conductivity, pH, total dissolved solids (TDS), salinity, and temperature were measured in situ using an YSI professional plus oxygen–temperature meter from just beneath of the surface in the stations, while water transparency was seasonally taken using a 20-cm Secchi disk during four seasons (from summer 2014 to summer 2015). Altitude, latitude, and longitude of the sampling stations were read from a geographical positioning system (Garmin Vista HCx model GPS). Chemical variables (e.g., TN, N–NH₄, N–NO₃, N–NO₂, TP, and P–PO₄) and biological oxygen demand (BOD₅) were analyzed using a standard method (APHA, 2012).



Figure 8.1 Location of the studied lakes and reservoirs in the West Mediterranean basin from Anonymous (2016)

8.2.2 Trophic States of the Sites

8.2.2.1 Trophic State Index (TSI) carlson (1977)

The trophic state index (TSI) Carlson (1977) based on calculating an index to determine the trophic state of a given water body on a scale of total phosphorus, Secchi depth, and chlorophyll is one of the commonly used classification methods for aquatic ecosystems especially lakes and reservoir. The TSI can be calculated by the following equations (Eq.10, Eq.11 and Eq.12) proposed by Carlson (1977).

$$TSI_{(TP)} = 10(6 - \ln[48 / TP] / \ln[2]) \quad (\text{Eq. 10})$$

$$TSI_{(SD)} = 10(6 - \ln[SD] / \ln[2]) \quad (\text{Eq. 11})$$

$$TSI_{(Cho)} = 10(6 - (2.04 - 0.68 \ln[Cho] / \ln[2])) \quad (\text{Eq. 12})$$

Where $TSI_{(TP)}$ the trophic state referenced to total phosphorus (TP), $TSI_{(SD)}$ is the trophic state referenced to Secchi depth (SD), and $TSI_{(Cho)}$ is the trophic state referenced the concentration of chlorophyll a.

In the present chapter, physicochemical data such as total phosphorus and water transparency were used to calculate the TSI index and to evaluate the water quality according to Carlson (1977). The trophic state classifications were determined based on trophic state index (TSI) as described in Table 8.1.

Table 8.1 Trophic classification for lakes based on TSI (Carlson, 1977)

Class	TSI
Oligotrophic	0–40
Mesotrophic	40–50
Eutrophic	50–70
Hypertrophic	>70

8.2.2.2 OECD (OECD, 1982) Classification System

The OECD system (OECD, 1982) classification method based on total phosphorus and Secchi depth was also used to classified trophic states of the aquatic systems. The trophic state classes of the waters bodies based on the OECD (1982) system were determined as described in Table 8.2.

Table 8.2 Trophic classification system for lakes (OECD, 1982)

Class	Total phosphorus (TP) (µg/L)	Secchi depht (SD) (m)
Ultraoligotrophic	<4	>12
Oligotrophic	4-10	12-6
Mesotrophic	10-35	6-3
Eutrophic	35-100	3-1.5
Hypertrophic	>100	<1.5

8.2.3 Statistical Analyses

To test the hypothesis of no difference trophic states between lakes and reservoir based on Carlson (1977) and OECD (OECD, 1982) trophic classification systems, the variance analysis (one-way ANOVA) was employed at 95% confidence interval using the program SPSS (IBM statistics version 23).

8.3 Results

8.3.1 Seasonal variation of total phosphorus and Secchi depth

The seasonal variation of total phosphorus (TP) and Secchi depth (SD) is given in Table 8.3. The total phosphorus contents and the Secchi depth values changed not only from one sampling station to another but also varied for the same station during the study period. In summer 2014, the highest TP (540 µg/L) and SD (2.5 m) were recorded at Lake Gölhisar and Çayboğazı reservoir respectively. Çavdır reservoir showed the highest TP (750 µg/L) and SD (1.5 m) in fall 2014, while the lowest TP values, 28 and 50 µg/L respectively in summer 2014 and fall 2014 were measured at Osmankalfalar reservoir. In spring 2015, the peaks of TP (170 µg/L) and SD (5 m) were recorded at Lake Yazır and Yapraklı reservoir respectively, while the lowest TP concentration (32 µg/L) in summer was recorded at Lake Avlan. On the other hand, the lowest SD was found at Lake Yazır during all the four seasons.

Table 8.3 Variation of total phosphorus (TP) and Secchi depth (SD) during the study

	Summer2014		Fall 2014		Spring 2015		Summer 2015	
	TP	SD	TP	SD	TP	SD	TP	SD
Lake Gölhisar	540	0.9	520	0.4	50	1.0	128	0.3
Çayboğazı reservoir	95	2.5	480	0.5	100	1.0	40	3.0
Lake Avlan	440	0.8	750	0.2	40	0.9	32	1.8
Geyik reservoir	90	1.2	710	1.5	70	1.5	112	1.5
Çavdır reservoir	95	2.3	750	1.5	50	2.5	175	1.5
Toptaş reservoir	86	0.7	510	0.4	40	1.0	37	0.6
Yapraklı reservoir	95	6.0	550	1.0	50	5.0	44	1.7
Osmankalfalar reservoir	28	1.1	50	0.5	108	0.7	167	1.0
Lake Yazır	70	0.3	70	0.2	170	0.2	130	0.3

Units: TP (µg/L); SD (m)

8.3.2 Trophic State Based on TSI Carlson (1977)

8.3.2.1 Trophic state variation in relation to total phosphorus

Trophic State Index (TSI) Carlson (1977) was used to determine trophic states of lakes and reservoirs in the West Mediterranean basin of Turkey. The result of TSI classification (Table 8.4) indicated that the trophic states of the lakes and reservoirs changed from one station to another and through the seasons. Peaks of TP corresponding to hypertrophic state (>70) were recorded at all stations in fall 2014 except for Osmankalfalar reservoir and Lake Yazır which indicated an eutrophic state (50-70). On another hand, all water bodies showed an eutrophic state in spring 2015 except Osmankalfalar reservoir which was hypertrophic. All the resevoirs (Çayboğazı, Çavdır, Toptaş, Yapraklı and Osmankalfalar) were eutrophic in summer 2014 and 2015, while Lake Gölhisar was hypertrophic. Lake Yazır was eutrophic in summer 2014 and indicated an hypertrophic state in summer 2015 in contast to Lake Avlan which was hypertrophic in summer 2014 and became eutrophic in summer 2015. The one-way ANOVA test between the mean TSI values indicated that there was no significant variation between the stations ($F= 0.34$, $p> 0.05$).

Table 8.4 Trophic state of the stations in relation to total phosphorus (Carlson, 1977)

	Summer 2014	Fall 2014	Spring 2015	Summer 2015
Lake Gölhisar	Hyp	Hyp	Eut	Hyp
Çayboğazı reservoir	Eut	Hyp	Eut	Eut
Lake Avlan	Hyp	Hyp	Eut	Eut
Geyik reservoir	Eut	Hyp	Eut	Eut
Çavdır reservoir	Eut	Hyp	Eut	Eut
Toptaş reservoir	Eut	Hyp	Eut	Eut
Yapraklı reservoir	Eut	Hyp	Eut	Eut
Osmankalfalar reservoir	Eut	Eut	Hyp	Eut
Lake Yazır	Eut	Eut	Eut	Hyp

Hyp = hypertrophic, Eut = eutrophic, Mes = mesotrophic

8.3.2.2 Trophic state variation in relation to Secchi depth

The trophic state of the water bodies and their corresponding SD during the study period are given in Table 8.5. A hypereutrophic state ($TSI >70$, and a Secchi depth <0.5) was observed for G11 all the study period except in spring 2015. An eutrophic

state (TSI ranged between 50 and 70, and a Secchi depth between 0.5 and 1 m) was observed in summer 2014 and spring 2015 for Lake Gölhisar, Lake Avlan, Geyik reservoir, Toptaş reservoir and Osmankalfalar reservoir, while a mesotrophic state (TSI ranged between 40 and 50, and a Secchi depth between 2 and 2.5 m) was observed for Çavdır reservoir and Çayboğazı reservoir. An oligotrophic state for Yapraklı reservoir (TSI <40, and a Secchi depth >5 m) was observed in summer 2014 and spring 2015. During fall 2014, Çavdır reservoir, Geyik reservoir, Osmankalfalar reservoir, Yapraklı reservoir and Çayboğazı reservoir indicated a eutrophic state, when a hypereutrophic state was observed in Lake Gölhisar, Lake Avlan and Toptaş reservoir. A mesotrophic state was observed for Çayboğazı reservoir, while the other water bodies showed a eutrophic state in summer 2015 except for Lakes Gölhisar and Yazır which showed a hypereutrophic state. A one-way ANOVA test between the means indicated that there was significant variation between the stations ($F= 6.35$, $P < 0.001$).

Table 8.5 Trophic state of the stations in relation to Secchi depth (Carlson, 1977)

	Summer 2014	Fall 2014	Spring 2015	Summer 2015
Lake Gölhisar	Eut	Hyp	Hyp	Hyp
Çayboğazı reservoir	Mes	Eut	Eut	Mes
Lake Avlan	Eut	Hyp	Hyp	Eut
Geyik reservoir	Eut	Eut	Hyp	Eut
Çavdır reservoir	Mes	Eut	Hyp	Eut
Toptaş reservoir	Eut	Hyp	Hyp	Eut
Yapraklı reservoir	Oli	Eut	Oli	Eut
Osmankalfalar reservoir	Eut	Eut	Hyp	Eut
Lake Yazır	Hyp	Hyp	Eut	Hyp

Hyp = hypertrophic, Eut = eutrophic, Mes = mesotrophic

8.3.3 Trophic State Based on OECD (1982)

8.3.3.1 Trophic State Variation in Relation to Total phosphorus

The variation of trophic state in relation to TP during the study period is given in Table 8.6. The classification based on OECD system according to the TP values in $\mu\text{g/L}$ classified all lakes and reservoirs in tow classes (hypertrophic and eutrophic) except Osmankalfalar reservoir and Lake Avlan which indicated a mesotrophic state

in summer 2014 and summer 2015 respectively. The commonly found state was the eutrophic class.

Table 8.6 Trophic state of the stations in relation to total phosphorus (OECD, 1982)

	Summer 2014	Fall 2014	Spring 2015	Summer 2015
Lake Gölhisar	Hyp	Hyp	Eut	Hyp
Çayboğazı reservoir	Eut	Hyp	Hyp	Eut
Lake Avlan	Hyp	Hyp	Eut	Mes
Geyik reservoir	Eut	Hyp	Eut	Hyp
Çavdır reservoir	Eut	Hyp	Eut	Hyp
Toptaş reservoir	Eut	Hyp	Eut	Eut
Yapraklı reservoir	Eut	Hyp	Eut	Eut
Osmankalfalar reservoir	Mes	Eut	Hyp	Hyp
Lake Yazır	Eut	Eut	Hyp	Hyp

Hyp = hypertrophic, Eut = eutrophic, Mes = mesotrophic

8.3.3.2 Trophic state variation in relation to Secchi depth

With regard to SD values, four classes were found in the studied stations. Oligotrophic state was observed only at Yapraklı reservoir in summer 2014. This reservoir indicated hypertrophic, mesotrophic and eutrophic states in fall 2014, spring and summer 2015 respectively (Table 8.7). On the other hand, hypertrophic was the dominant class during the four seasons of the study compared to eutrophic and mesotrophic states.

Table 8.7 Trophic state of the station in relation to Secchi depth (OECD, 1982)

	Summer 2014	Fall 2014	Spring 2015	Summer 2015
Lake Gölhisar	Hyp	Hyp	Hyp	Hyp
Çayboğazı reservoir	Eut	Hyp	Hyp	Mes
Lake Avlan	Hyp	Hyp	Hyp	Eut
Geyik reservoir	Hyp	Eut	Eut	Eut
Çavdır reservoir	Eut	Eut	Eut	Eut
Toptaş reservoir	Hyp	Hyp	Hyp	Hyp
Yapraklı reservoir	Oli	Hyp	Mes	Eut
Osmankalfalar reservoir	Hyp	Hyp	Hyp	Hyp
Lake Yazır	Hyp	Hyp	Hyp	Hyp

Hyp = hypertrophic, Eut = eutrophic, Mes = mesotrophic, Oli = oligotrophic

8.3.4 Comparison of the TSI(TP) and TSI(SD)

The two indices indicated similar trophic state (eutrophic) in spring 2015 for Çayboğazı reservoir, Osmankalfalar reservoir, Toptaş reservoir, Geyik reservoir, and Lake Avlan. The significant was observed in summer 2014 at Yapraklı reservoir which indicated a hypertrophic state following $TSI_{(TP)}$ and on the other hand became oligotrophic based on $TSI_{(SD)}$ (Figure 8.2).

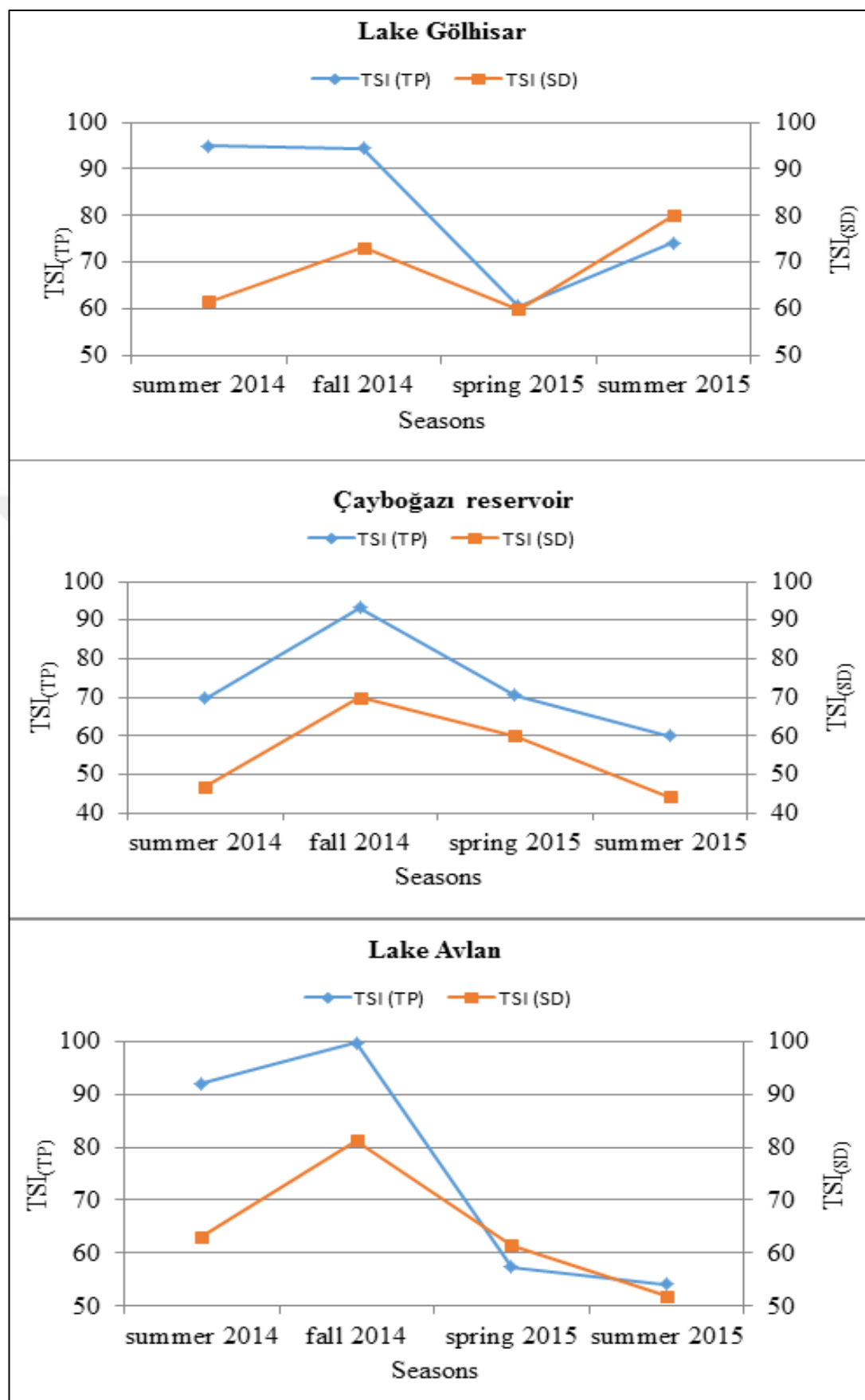


Figure 8.2a variation of $TSI_{(TP)}$ and $TSI_{(SD)}$ in Lake Gölhisar, Çayboğazı reservoir and Lake Avlan

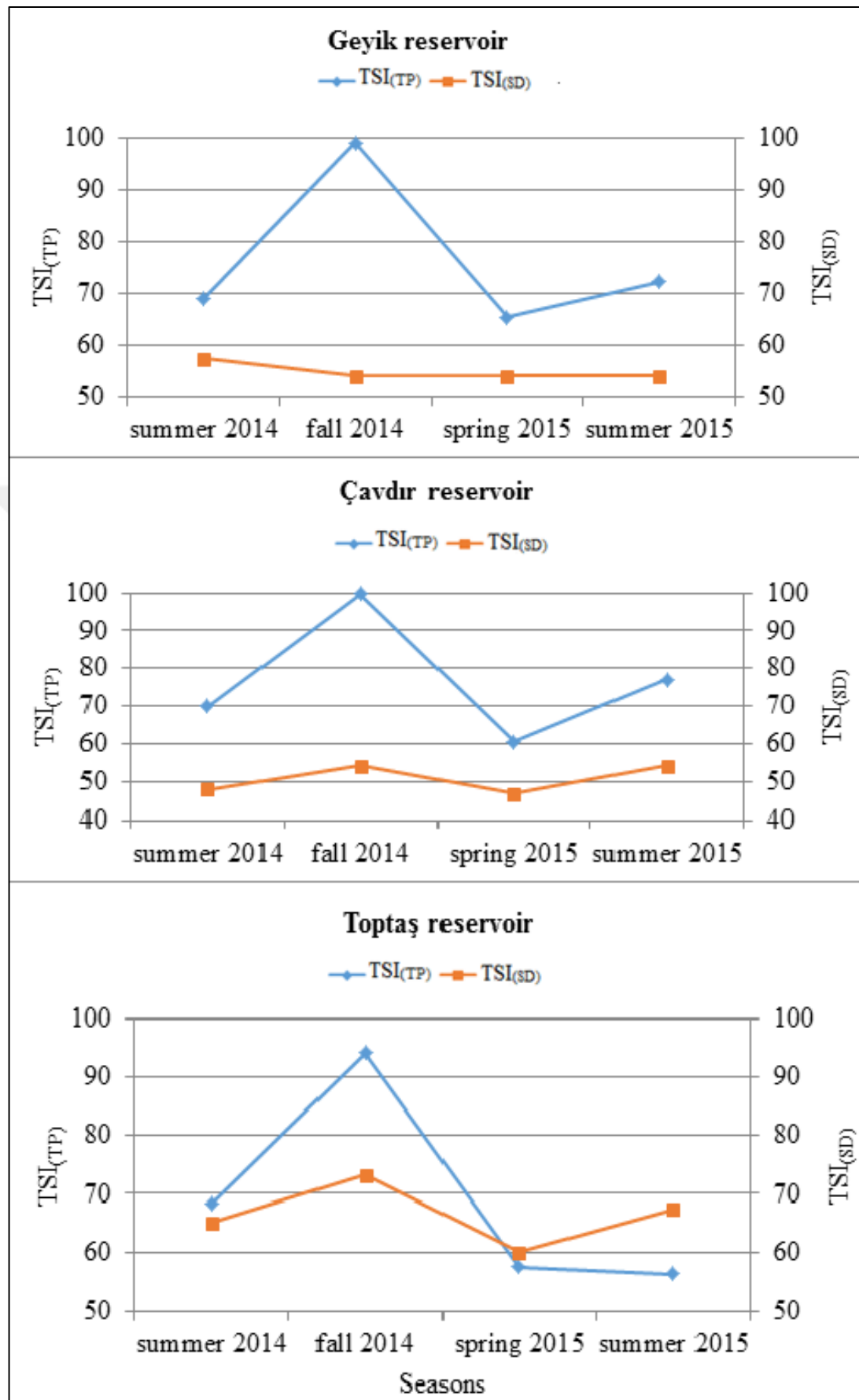


Figure 8.2b variation of $TSI_{(TP)}$ and $TSI_{(SD)}$ in Geyik reservoir, Çavdır reservoir and Toptaş reservoir

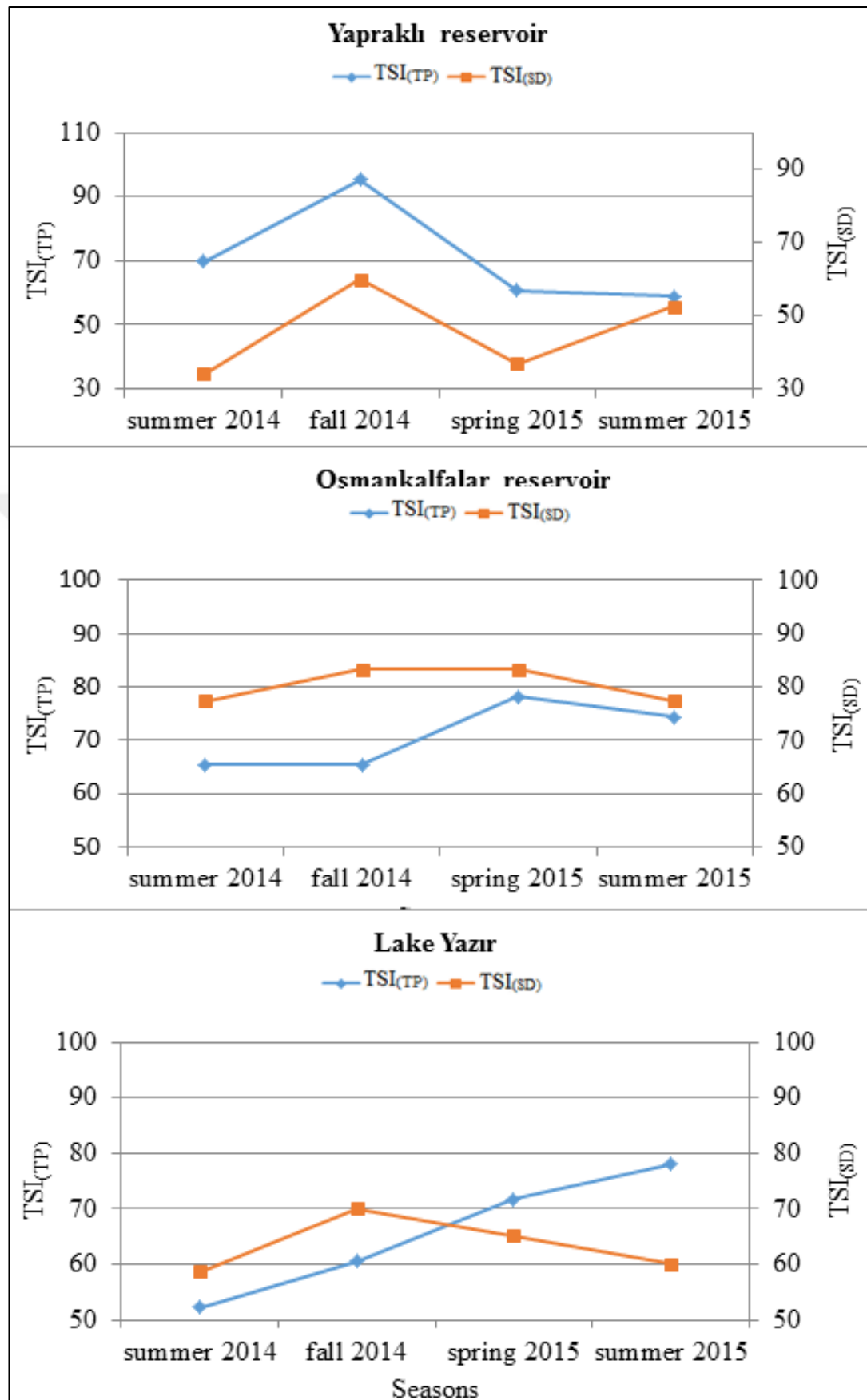


Figure 8.2c variation of $TSI_{(TP)}$ and $TSI_{(SD)}$ in Yapraklı reservoir, Osmankalfalar reservoir and Lake Yazır

8.4 Discussion

8.4.1 Environmental variables

Physicochemical parameter study is very important to get exact idea about the quality of water (Patil et al., 2012). The quality of surface water depends on various nutrients and their concentration, which are mostly derived from the water bodies catchment area.

Total phosphorus (TP) variable was mainly high in the water bodies during the study time. The highest value of TP ($420.8 \mu\text{g l}^{-1}$) was found in Osmankalfalar reservoir (Table 1). This is probably the consequence of excessive fish farming and water seasonal level fluctuations (from 15 m to 3 m) due to irrigation. Sevindik et al., 2017 indicated that the seasonal variation of precipitations and the high use of water in summer for irrigation purposes decrease water level for about 10 m in İkizcetepeler reservoir (Turkey). According to Sgro et al. (2006), local parameters may be more important determinants of water quality and increase variability among sites during low flow periods. In the same idea, Kennedy and Walker (1990) indicated that maximum environmental gradients are associated in reservoirs with high water retention time, high sedimentation rates and flow that are controlled by advective processes. With regard to lakes, Lake Avlan had higher TP value ($365 \mu\text{g l}^{-1}$) than other lakes. This lake was a very shallow and eutrophic ecosystem invaded by macrophytes. Lake Gölhisar, a shallow lake with dense vegetation and turbid water. The lake is near to the District of Gölhisar and surrounded by several villages. This result could be the consequences of anthropogenic activities especially discharge of domestic waste and agriculture fertilizers which can affect the lake through generated organic nutrients. Mancini and Arcà (2000) reported that urbanization and agriculture practices in a study area could affect the basins of watercourses. High TN values were measured in Lake Gölhisar ($1569.3 \mu\text{g l}^{-1}$) and Osmankalfalar reservoir ($1073.8 \mu\text{g l}^{-1}$). These results could be partially due to human activities in the water bodies' catchments. In accordance with our result, Strebel et al. (1989) and Almasri and Kaluarachchi (2004) indicated that the downstream increase of nitrate is a common

human impact by agriculture. Loehr (1974) and Dodds (2002) indicated that human activities that lead to cultural eutrophication include the use of agricultural fertilizers, livestock practices, and release of nutrient-rich sewage into the surface water.

8.4.2 Trophic State

In the present study the trophic states of 3 lakes and 6 reservoirs were examined in the west Mediterranean basin of Turkey based on Carlson (1977) TSI system which was developed on the hypothesis that there is a close relationship between total phosphorus, chlorophyll-a and Secchi depth in phosphorus limited lakes (Matthews et al., 2002) and the OECD classification system (OECD, 1982). Lakes and reservoirs showed different trophic states according to TP and SD variables and the classification system at the different water bodies but indicated similar trophic states based on each of the two systems. This situation confirmed our hypothesis of no difference between the trophic states of the lakes and reservoirs. Three trophic categories were (Mesotrophic, eutrophic and hypertrophic) recorded based on TSI (Carlson, 1977), while four trophic states (Oligotrophic, Mesotrophic, eutrophic and hypertrophic) were found according to the OECD criteria (OECD, 1982). Brenner et al., 1999 and Dodds (2002) reported that lakes may not clearly fall into an individual class in any of the classification methods. For example, TP could be high enough for a lake to be categorized as eutrophic, but water clarity by suspended sediments could keep the chlorophyll amount in the mesotrophic range. Also, TP and phytoplankton composition could be low in a lake with extensive macrophytes biomass and production.

Yapraklı reservoir was mesotrophic according to SD based on Carlson (1977) system, while other reservoirs and lakes showed eutrophic status according to TP and SD. OECD criteria classified lakes and reservoirs as hypertrophic based on TP. According to SD, all lakes had hypertrophic status, while Geyik, Toptaş, and Osmankalfalar reservoirs were eutrophic; Çayboğazı and Çavdır reservoirs were recorded as meso-eutrophic, while an oligotrophic status was observed for Yapraklı reservoir. Similar trophic status related to TP and SD were previously found in early studies (Carlson, 1977; Vollenweider and Kerekes, 1982; Marchetto et al., 2009). These trophic states have also been indicated by Moreno-Ostos et al. (2008) in El Gergal Reservoir (Spain), Becker et al. (2010) in Sau Reservoir (Spain), at Lakes

Sumin and Másłuchowskie in Eastern Poland (Pasztaleniec and Poniewozik, 2010), at Pareja limno-Reservoir (Spain) (Molina-Navarro et al., 2012, 2014) at Alleben Reservoir (Turkey) (Çelekli and Öztürk, 2014), at Lake Lacanau and Lakes Parentis and Hourtin (France) (Cellamare et al., 2012). According to the Carlson's Trophic State Indices and OECD trophic classifications using Secchi depth, and total phosphorous mesotrophic and meso-eutrophic states were indicated at Lake Abant and Gölköy respectively (Çelekli, 2006). In accordance with our finding, Erdoğan (2016) reported a high variability from hypertrophic to oligotrophic for 48 lakes and 12 reservoirs including Gölhisar which showed a hypertrophic state during the present study. The Mediterranean characteristics the geographical properties, climate, land use, and anthropogenic activities which could change environmental factors could be the consequence of the similarity of the trophic states of the reservoirs and lakes.

The mesotrophic, eutrophic and hypertrophic states observed at the lakes and reservoirs could be due to anthropogenic activities. In fact, livestock and agriculture activities have an important place in the Western Mediterranean Basin. The most common crops which grow in the basin are tomatoes, peppers, citrus fruits, apples, pears, quince, wheat, corn, and olives. Fish production and packaging, olive oil production and mining are the main commercial activities. These activities are a great source of organic and inorganic pollutants in a river catchment. Salomoni et al. (2006) reported that high population densities and the large number of industrial and agricultural activities expose most watersheds near to large urban centers to adverse environmental impacts, particularly to pollutions by domestic and industrial wastewaters. This situation was confirmed by Mancini and Arcà (2000) who reported that urbanization and agriculture practices in the study area affect the basins of watercourses.

CHAPTER VIV

CONCLUSION

9.1 Phytoplankton composition and water quality in the lakes and reservoirs

This study revealed that phytoplankton brings complementary information, reflected and integrated unambiguously the ecological status of the lakes and reservoirs. A total of 206 species belonging to 9 classes and 15 functional groups with a dominance of **B** group were identified during the study period. *P. cinctum*, *C. ocellata*, *C. hirundinella*, *P. morum*, *C. ovata*, *M. aeruginosa*, *M. flosaquae* and *Peridinium willei* had important contributions to total phytoplankton biovolume. Phytoplankton functional groups system was used as an indicator of trophic status for lakes and reservoirs in Turkey for the first time as basin wide. The study confirmed the suitable applicability of this system groups for evaluationg the trophic state and provided valuable and complementary knowledge on phytoplankton structures, development, and tolerances to classify and define phytoplankton dynamics in the water bodies. The spatial and temporal occurrences of phytoplankton species were strictly associated with the changes of some important environmental factors in lakes and reservoirs in the West Mediterranean basin. This interactions between predictor factors and the phytoplankton communities in the lakes and reservoirs was elucidated by use of multivariate approaches. Results of the present study indicated that the dominance of phytoplankton assemblages differed among water bodies and seasons due to the difference of species preference in the tolerance range of water temperature and nutrient limitations. Trophic state indices and ecological quality ratio based on PTI and Med-PTI and Q assemblage index were used to evaluate trophic states and ecological quality of lakes and reservoirs in this basin. As basin approach, the present study regarding limnoecology of phytoplankton assemblages in lakes and reservoirs is the first study in Turkey. Further limnoecological studies with multivariate approaches and indices should be conducted to resolve some of the biological deteriorations of different water bodies that this study has pointed out. The use of statistical analysis to examine the impact of various complex predictor factors

on the phytoplankton species composition and distribution is potentially a suitable tool to understand the role of the effective parameters. From that point, the present work is be a contribution to the use of phytoplankton characteristics as a ecological indicators in this region. However, the use of these indices could lead to erroneous interpretation of water quality because ecoregional changes in algal composition, and there is a need to develop a national phytoplankton based metrics as the newly developed diatom-based metric called trophic index Turkey.

9.2 Epithic diatom composition and water quality in the running waters

A total of 102 species of epilithic diatoms from 22 running water (stream and creek) were identified during the present study. *Navicula*, *Cymbella*, *Nitzschia*, *Gomphonema*, *Fragilaria*, and *Cocconeis* were the most dominant genus recorded while *F. capucina*, *C. excisa*, *G. parvulum*, *U. ulna* and *C. placentula* were the common and abundant species observed in the stations. The ecological states obtained for the different sites based on the trophic index relate exactly the state of deterioration of these water bodies due to human activities such as agriculture and fisheries in the basin. The Trophic index in Turkey (TIT) should be appropriate for our studied water bodies and this conclusion is similar to that found in the previous reports of Yusano et al., (1994) and Pongpan and Yuwadee (2014), who suggested that the usefulness of the index should be developed from the organisms in that area. However, this index should be revised to include all phytobenthos communities from lakes, reservoirs and running water to extend its applicability to a number of aquatic systems in Turkey and wherever.

9.3 Trophic state and water quality in the lakes and reservoirs

The trophic state of 3 lakes and 6 reservoirs in the western Mediterranean basin of Turkey based on Carlson's Trophic State Index the OECD trophic classification system. Total phosphorus concentrations and the Secchi depth values among stations and seasons. In summer 2014, the highest TP (540 µg/L) and SD (2.5 m) were recorded at Lake Gölhisar and Çayboğazı reservoir respectively. Çavdır reservoir showed the highest TP (750 µg/L) and SD (1.5 m) in fall 2014, while the lowest TP values, 28 and 50 µg/L respectively in summer 2014 and fall 2014 were measured at Osmankalfalar reservoir. In spring 2015, the peaks of TP (170 µg/L) and SD (5 m)

were recorded at Lake Yazır and Yapraklı reservoir respectively, while the lowest TP concentration (32 µg/L) in summer was recorded at Lake Avlan.

With regard to water quality, 3 trophic categories were recorded based on trophic state index, while 4 trophic states were found according to the OECD criteria. Yapraklı reservoir was mesotrophic according to SD based on Carlson system, while other reservoirs and lakes showed eutrophic status according to TP and SD. OECD criteria classified lakes and reservoirs as hypertrophic based on TP. According to SD, all lakes had hypertrophic status, while Geyik, Toptaş, and Osmankalfalar reservoirs were eutrophic; Çayboğazı and Çavdır reservoirs were recorded as meso-eutrophic, and oligotrophic status was observed for Yapraklı reservoir. From that point, it appeared that they applied systems seem to be suitable to assess the trophic state of the studied water bodies.

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APPENDICES

Appendix A List of diatom species and varieties found in the studied sites

Species	Code
<i>Achnanthes impexa</i> Lange-Bertalot	Acim
<i>Amphora subcapitata</i> (Kisselew) Hustedt	Amsu
<i>Aulocoseria granulata</i> (Ehrenberg) Simonsen	Augr
<i>Bacillaria obtusa</i> (W.Smith) Elmore	Baob
<i>Diatoma vulgare</i> Bory	Divu
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	Clov
<i>Cocconeis disculus</i> (Schumann) Cleve in Cleve & Jentzsch	Codi
<i>Cocconeis placentula</i> Ehrenberg	Copl
<i>Cocconeis placentula</i> var. <i>Placentula</i> Ehrenberg	Coplpl
<i>Cocconeis placentula</i> var. <i>lineata</i> (Ehrenberg) Van Heurck	Coplplv
<i>Cocconeis scutellum</i> Ehrenberg	Cosc
<i>Craticula accomoda</i> (Hustedt) D.G. Mann	Crac
<i>Craticula halophila</i> (Grunow) Cleve	Crha
<i>Cyclotella iris</i> Brun & Héribaud-Joseph	Cyir
<i>Cyclotella meneghiniana</i> Kützing	Cyme
<i>Cyclotella ocellata</i> Pantocsek	Cyoc
<i>Cymatopleura solea</i> (Brébisson) W. Smith	Cyso
<i>Cymbella cistula</i> (Hemprich) Kirchner	Cyci
<i>Cymbella cymbiformis</i> Agardh	Cycy
<i>Cymbella ehrenbergii</i> Kützing	Cyeh
<i>Cymbella excisa</i> Kützing	Cyex
<i>Cymbella hantzschiana</i> Krammer	Cyha
<i>Cymbella helvetica</i> Kützing	Cyhe
<i>Cymbella mesiana</i> Cholnoky	Cymes
<i>Cymbella simonseni</i> Krammer	Cyst
<i>Cymbella tumida</i> (Brébisson) van Heurck	Cytu
<i>Cymboplectra amphicephala</i> Nägeli in Kützing	Cyam
<i>Cymboplectra naviculiformis</i> (Auerswald ex Heiberg) Krammer	Cyna
<i>Denticula elegans</i> Kützing	Deel
<i>Diatoma hyemalis</i> (Roth) Heiberg	Dihy
<i>Diatoma moniliformis</i> (Kützing) D.M. Williams	Dimo
<i>Diatoma tenuis</i> C. Agardh	Dite
<i>Dickieia expecta</i> S.L. VanLandingham	Naex
<i>Diploneis modica</i> Hustedt	Dimo
<i>Encyonema minutum</i> (Hilse) D.G. Mann	Enmi

Appendix A Continue

Species	Code
<i>Encyonema silesiacum</i> (Bleisch) D.G.Mann	Ensi
<i>Epithemia adnata</i> (Kützing) Brébisson	Epad
<i>Epithemia cystula</i> (Ehrenberg) Ralfs	Epar
<i>Epithemia frickei</i> Krammer	Epfr
<i>Eunotia tenella</i> (Grunow) Hustedt	Eute
<i>Fragilaria capucina</i> Desmazières	Frca
<i>Fragilaria crotonensis</i> Kitton	Frcr
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	Frdi
<i>Gomphonema acuminatum</i> Ehrenberg	Goac
<i>Gomphonema angustum</i> C. Agardh	Goan
<i>Gomphonema clavatum</i> Ehrenberg	Gocl
<i>Gomphonema gracile</i> Ehrenberg	Gogr
<i>Gomphonema minutum</i> (C. Agardh) C. Agardh	Gomi
<i>Gomphonema parvulum</i> Kützing	Gopa
<i>Gomphonema pseudoaugur</i> Lange-Bertalot	Gops
<i>Gomphonema truncatum</i> Ehrenberg	Gotr
<i>Gyrosigma acuminatum</i> (Kützing) Rabenhorst	Gyac
<i>Gyrosigma attenuatum</i> (Kützing) Rabenhorst	Gyat
<i>Hantzschia distinctepunctata</i> var. <i>circuligera</i> Compère	Hadi
<i>Hantzschia weyprechtii</i> Grunow	Hawe
<i>Lysigonium lineatum</i> (Dillwyn) Trevisan	Lyli
<i>Lysigonium varians</i> (C.Agardh) De Toni	Lyva
<i>Navicula bryophila</i> J.B.Petersen	Nabr
<i>Navicula capitatoradiata</i> H.Germain	Naca
<i>Navicula cincta</i> (Ehrenberg) Ralfs in Pritchard	Naci
<i>Navicula clementis</i> Grunow	Nacl
<i>Navicula constans</i> Hustedt	Naco
<i>Navicula cryptocephala</i> Kützing	Nacr
<i>Navicula cryptonella</i> Lange-Bertalot	Nacry
<i>Navicula digitoradiata</i> (Gregory) Ralfs	Nadi
<i>Navicula gregaria</i> Donkin	Nagr
<i>Navicula jaagii</i> Meister	Naja
<i>Navicula lanceolata</i> (C.Agardh) Kützing	Nala
<i>Navicula laterostrata</i> Hustedt	Nalat
<i>Navicula leptostriata</i> Jørgensen	Nale
<i>Navicula margalithii</i> Lange-Bertalot	Nama
<i>Navicula oppugnata</i> Hustedt	Naop
<i>Navicula peregrina</i> (Ehrenberg) Kützing	Nape
<i>Navicula phylepta</i> Kützing	Naph
<i>Navicula radians</i> Héribaude-Joseph	Narad
<i>Navicula radiosa</i> Kützing	Nara
<i>Navicula recens</i> (Lange-Bertalot) Lange-Bertalot	Nare

Appendix A Continue

Species	Code
<i>Navicula schoenfeldii</i> Hustedt	Nasc
<i>Navicula slesvicensis</i> Grunow in van Heurck	Nasl
<i>Navicula tripunctata</i> (O.F.Müller) Bory	Natr
<i>Navicula trivialis</i> Lange-Bertalot	Natri
<i>Navicula tuscula</i> Ehrenberg	Natus
<i>Navicula veneta</i> Kützing	Nave
<i>Navicula vitabunda</i> Hustedt	Navi
<i>Nitzschia angustata</i> (W.Smith) Grunow	Nian
<i>Nitzschia brevissima</i> Grunow in Van Heurck	Nibr
<i>Nitzschia dissipata</i> (Kützing) Grunow	Nidi
<i>Nitzschia elongata</i> Hassal	Niel
<i>Nitzschia flexa</i> Schumann	Nifl
<i>Nitzschia recta</i> Hantzsch ex Rabenhorst	Nire
<i>Nitzschia sigmoidea</i> (Nitzsch) W.Smith	Nisi
<i>Nitzschia umbonata</i> (Ehrenberg) Lange-Bertalot	Nium
<i>Stauroneis phoenicenteron</i> (Nitzsch) Ehrenberg	Stph
<i>Surirella amphioxys</i> W.Smith	Suam
<i>Surirella angusta</i> Kützing	Suan
<i>Surirella brebissonii</i> Krammer & Lange-Bertalot	Subr
<i>Surirella minuta</i> Brébisson	Sumi
<i>Surirella ovalis</i> Brébisson	Suov
<i>Surirella subsalsa</i> W.Smith	Susu
<i>Ulnaria biceps</i> (Kützing) Compère	Ulbi
<i>Ulnaria ulna</i> (Nitzsch) Compère	Ulul

Appendix B List of diatom species recorded in the running water stations in summer 2014

Species	A2	A3	A5	A6	A7	A8	A9	A10	A11	A12	A15	A17	A19	A21
<i>Achnanthes impexa</i> Lange-Bertalot	-	-	-	-	-	-	-	-	-	+	-	-	-	+
<i>Aulacoseira granulata</i> Simonsen	+	-	-	-	+	-	-	-	-	+	+	+	+	-
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	-	+	+	+	+	+	-	-	-	-	-	-	-	-
<i>Cocconeis placentula</i> Ehrenberg	+	+	+	-	+	+	+	+	+	+	+	+	+	+
<i>Cyclotella iris</i> Brun & H-Joseph	-	-	+	+	+	-	-	-	+	+	-	-	-	-
<i>Cyclotella meneghiniana</i> Kützing	-	-	-	+	+	-	+	-	-	-	-	-	-	-
<i>Cymatopleura solea</i> Smith	-	-	-	-	-	+	-	-	-	-	-	-	-	-
<i>Cymbella affinis</i> Kützing	-	+	-	-	-	-	-	-	-	-	+	-	+	+
<i>Cymbella amphicephala</i> Näegeli	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Cymbella cistula</i> Kirchner	-	-	+	-	-	-	-	-	-	-	+	-	-	-
<i>Cymbella cymbiformis</i> Agardh	-	+	-	-	-	-	-	-	+	-	-	-	-	-
<i>Cymbella ehrenbergii</i> Kützing	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Cymbella excisa</i> Kützing	-	-	+	+	-	+	+	+	+	-	-	-	-	-
<i>Cymbella hantzschiana</i> Krammer	-	-	-	-	-	+	-	-	-	-	-	-	-	-
<i>Cymbella minuta</i> Hilse	-	-	+	+	-	+	-	+	-	-	-	+	+	-
<i>Cymbella tumida</i> Heurck	+	-	-	-	-	+	-	-	-	-	-	-	-	-
<i>Denticula elegans</i> Kützing	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Diatoma vulgare</i> Bory	-	-	+	+	+	-	+	+	+	-	-	+	+	-
<i>Fragilaria biceps</i> Ehrenberg	-	-	-	+	+	-	-	-	+	-	-	-	-	-
<i>Fragilaria capucina</i> Desmazières	-	-	-	-	-	-	-	-	-	-	+	-	-	-
<i>Ulnaria ulna</i> (Nitzsch) Compère	-	+	+	+	+	+	+	+	+	-	-	-	-	+
<i>Gomphonema minutum</i> Agardh	-	-	-	+	-	+	+	+	+	-	+	-	-	-
<i>Gomphonema parvulum</i> Kützing	-	+	+	+	+	-	-	+	-	-	+	+	-	+
<i>Gomphonema truncatum</i> Ehrenberg	-	+	+	+	-	-	-	-	-	-	+	-	-	-
<i>Gyrosigma acuminatum</i> Rabenhorst	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Gyrosigma attenuatum</i> Rabenhorst	-	-	+	-	-	-	+	-	-	-	-	-	-	-

Appendix B Continue

Species	A2	A3	A5	A6	A7	A8	A9	A10	A11	A12	A15	A17	A19	A21
<i>Navicula bryophila</i> J.B.Petersen	-	+	-	-	-	-	-	-	-	-	-	-	-	+
<i>Navicula cryptocephala</i> Kützing	+	-	-	-	-	+	+	-	+	-	-	-	-	-
<i>Navicula cryptonella</i> Lange-Bertalot	-	-	-	+	-	-	-	-	-	-	-	-	-	+
<i>Navicula laterostrata</i> Hustedt	-	-	-	-	-	-	+	-	-	+	-	-	-	-
<i>Navicula margalithii</i> Lange-Bertalot	-	-	-	-	-	-	-	-	+	-	-	-	-	-
<i>Navicula oppugnata</i> Hustedt	+	-	+	-	-	+	+	-	+	-	+	+	+	+
<i>Navicula phyllepta</i> Kützing	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Navicula radiosa</i> Kützing	-	-	-	-	-	-	-	-	+	-	-	-	-	-
<i>Navicula trivialis</i> Lange-Bertalot	-	-	-	-	+	+	-	-	-	-	-	-	-	-
<i>Navicula tuscula</i> Ehrenberg	-	-	+	-	-	-	-	-	-	-	-	-	-	-
<i>Nitzschia dissipata</i> Rabenhorst	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Nitzschia elongata</i> Hassal	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Nitzschia obtusa</i> W.Smith	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Nitzschia recta</i> H. ex Rabenhorst	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Nitzschia sigmoidea</i> (Nitzsch) Smith	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Nitzschia umbonata</i> L-Bertalot	-	-	-	-	-	-	+	-	-	-	+	-	-	-
<i>Surirella minuta</i> Brébisson	-	-	-	-	-	-	-	-	-	-	+	-	-	-

+ and – indicate presence and absence of species. For station codes, see Table 5.1

Appendix C List of diatom species recorded in the running water stations in fall 2014

Species	A2	A3	A5	A6	A7	A8	A9	A1-	A11	A12	A15	A16	A17	A19	A21
<i>Aulacoseira granulata</i> Simonsen	+	-	-	-	-	-	-	-	-	-	-	+	-	+	-
<i>Clevamphora ovalis</i> (Kützing) Mereschkowsky	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-
<i>Cocconeis placentula</i> Ehrenberg	+	+	-	+	-	-	-	+-	+	+	-	-	+	-	-
<i>Cocconeis placentula</i> var. <i>lineata</i> Heurck	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Cyclotella iris</i> Brun & H-Joseph	-	-	-	-	-	-	-	-	-	-	-	+	-	+	-
<i>Cyclotella ocellata</i> Pantocsek	+	+	-	+	-	-	-	+	-	+	-	-	+	-	-
<i>Cymbella affinis</i> Kützing	+	+	-	+	-	+	-	-	+	+	-	+	+	-	+
<i>Cymbella cymbiformis</i> Agardh	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-
<i>Cymbella helvetica</i> Kützing	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-
<i>Cymbella minuta</i> Hilse	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-
<i>Cymbella simonsenii</i> Krammer	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Diatoma moniliformis</i> Williams	-	-	-	-	-	+	-	-	-	-	-	-	-	-	+
<i>Diatoma vulgare</i> Bory	-	+	-	-	-	-	-	+	+	-	-	-	-	-	+
<i>Diploneis modica</i> Hustedt	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-
<i>Fragilaria capucina</i> Desmazières	+	-	-	+	-	-	-	+	+	+	-	-	+	-	-
<i>Fragilaria crotonensis</i> Kitton	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-
<i>Ulnaria ulna</i> (Nitzsch) Compère	-	+	-	+	-	+	-	-	-	+	-	+	-	-	+
<i>Gomphonema angustum</i> Agardh	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
<i>Gomphonema clavatum</i> Ehrenberg	-	+	-	+	-	-	-	-	+	+	-	-	-	-	-
<i>Gomphonema minutum</i> Agardh	-	-	-	-	-	+	-	-	-	-	-	+	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	+	-	+	-	+	-	-	+	+	-	-	-	-	+
<i>Melosira lineata</i> Agardh	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula clementis</i> Grunow	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-
<i>Navicula constans</i> Hustedt	-	-	-	+	-	-	-	-	-	+	-	-	-	-	-
<i>Navicula cryptocephala</i> Kützing	-	-	-	-	-	+	-	-	-	-	-	-	-	-	+
<i>Navicula lanceolata</i> Kützing	+	-	-	-	-	-	-	+	-	-	-	-	-	-	-

Appendix C Continue

Species	A2	A3	A5	A6	A7	A8	A9	A1-	A11	A12	A15	A16	A17	A19	A21
<i>Navicula leptostriata</i> Jørgensen	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Navicula oppugnata</i> Hustedt	-	-	-	-	-	+	-	-	-	-	-	-	-	-	+
<i>Navicula peregrina</i> Kützing	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-
<i>Navicula phyllepta</i> Kützing	-	-	-	-	-	-	-	-	+	-	-	-	-	-	+
<i>Navicula recens</i> Lange-Bertalot	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula trivialis</i> Lange-Bertalot	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-
<i>Navicula vitabunda</i> Hustedt	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Nitzschia angustata</i> Grunow	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-
<i>Nitzschia dissipata</i> Rabenhorst	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+
<i>Nitzschia umbonata</i> Lange-Bertalot	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-
<i>Surirella angusta</i> Kützing	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-
<i>Surirella brebissonii</i> Krammer & L-Bertalot	-	-	-	-	-	+	-	-	-	-	-	-	-	-	+

+ and – indicate presence and absence of species. For station codes, see Table 5.1

Appendix D List of diatom species recorded in the running water stations in spring 2015

Species	A2	A4	A5	A6	A8	A11	A14	A15	A16	A17	A18	A19	A20	A21	Bak4	Bak9	Bak12	R1
<i>Amphora inariensis</i>	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Amphora subcapitata</i>	-	-	+	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Aulacoseira granulata</i>	-	-	-	-	-	-	+	-	-	+	-	-	-	-	-	-	-	-
<i>Clevamphora ovalis</i>	+	-	-	+	-	-	-	-	-	-	-	-	+	+	-	+	-	-
<i>Cocconeis disculus</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Cocconeis placentula</i>	-	+	+	-	-	-	-	+	-	+	-	-	+	-	-	+	+	-
<i>Cocconeis scutellum</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Cyclotella ocellata</i>	+	+	-	-	-	+	+	-	+	-	+	-	-	-	+	-	+	+
<i>Cyclotella radiosia</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Cymatopleura solea</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Cymbella affinis</i>	+	-	+	+	+	+	+	+	+	-	+	+	+	+	+	+	+	+
<i>Cymbella tumida</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-
<i>Denticula elegans</i>	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-
<i>Diatoma tenuis</i>	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-
<i>Diatoma vulgaris</i>	-	-	+	+	+	+	-	-	+	-	-	-	+	+	+	-	+	-
<i>Epithemia frickei</i>	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-
<i>Fragilaria biceps</i>	-	-	-	-	-	-	-	-	-	-	+	-	-	+	-	-	-	-
<i>Fragilaria capucina</i>	+	-	+	+	+	+	-	+	+	+	+	+	+	+	+	+	+	+
<i>Fragilaria dilatata</i>	+	-	-	-	-	-	+	+	+	-	+	-	-	-	+	-	-	-
<i>Fragilaria fasciculata</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Ulnaria ulna</i>	+	-	+	+	+	+	+	+	+	+	+	+	+	+	-	+	+	+
<i>Gomphonema angustum</i>	-	-	+	+	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Gomphonema parvulum</i>	+	-	-	-	+	-	-	+	+	+	+	-	+	+	+	+	+	+
<i>Gomphonema pseudoaugur</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Gomphonema truncatum</i>	-	-	+	-	+	-	-	-	+	-	-	-	+	+	+	+	-	-
<i>Hantzschia weyprechtii</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-

Appendix D Continue

Species	A2	A4	A5	A6	A8	A11	A14	A15	A16	A17	A18	A19	A20	A21	Bak4	Bak9	Bak12	R1
<i>Melosira lineata</i>	-	-	-	-	-	+	+	-	-	-	+	-	-	-	-	-	-	-
<i>Melosira varians</i>	+	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula capitatoradiata</i>	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-
<i>Navicula cincta</i>	-	-	-	-	-	-	-	+	+	-	-	-	-	-	-	+	-	-
<i>Navicula clementis</i>	-	-	-	-	-	+	-	-	-	-	+	-	-	-	-	-	-	-
<i>Navicula digitoradiata</i>	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula expecta</i>	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula lanceolata</i>	-	-	+	-	-	+	-	+	-	-	-	-	+	-	-	-	-	-
<i>Navicula pusilla</i>	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula radiosa</i>	-	-	-	+	+	-	-	-	-	-	-	+	-	-	-	-	-	-
<i>Navicula schoenfeldii</i>	-	-	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula tripunctata</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Navicula trivialis</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Navicula veneta</i>	-	-	-	-	-	-	-	+	-	-	-	-	-	+	-	-	-	-
<i>Nitzschia brevissima</i>	-	-	-	-	-	-	+	-	+	-	-	-	-	-	-	-	-	-
<i>Nitzschia dissipata</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-
<i>Nitzschia flexa</i>	-	-	+	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-
<i>Nitzschia recta</i>	-	-	+	+	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Surirella angusta</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-
<i>Surirella ovalis</i>	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Surirella subsalsa</i>	+	-	+	-	-	-	-	-	-	-	-	-	-	+	-	-	-	-

+ and – indicate presence and absence of species. For station codes, see Table 5.1

Appendix E List of diatom species recorded in the running water stations in summer 2015

Species	A3	A5	A6	A8	A9	A10	A11	A12	A13	A15	A16	A17	A18	A19	A21	R1	Bak 14
<i>Aulacoseira granulata</i>	-	-	-	-	+	-	-	-	-	+	+	-	-	+	-	-	-
<i>Clevamphora ovalis</i>	+	+	-	-	+	-	-	-	-	-	-	+	+	-	-	-	-
<i>Cocconeis placentula</i>	+	+	+	+	+	+	+	+	-	+	+	+	+	+	+	-	+
<i>Cyclotella ocellata</i>	-	+	-	-	-	+	+	+	-	-	-	+	+	-	+	-	+
<i>Cymatopleura solea</i>	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Cymbella affinis</i>	+	+	+	+	+	+	+	+	-	+	-	+	-	+	+	+	+
<i>Cymbella cistula</i>	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	+	-
<i>Cymbella mesiana</i>	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Cymbella minuta</i>	-	-	-	-	-	+	-	-	-	-	+	-	-	+	-	-	-
<i>Cymbella naviculiformis</i>	-	-	-	-	-	-	-	+	-	-	-	-	+	-	+	-	-
<i>Cymbella silesiaca</i>	+	+	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Denticula elegans</i>	-	-	-	-	+	-	+	-	-	-	-	-	-	-	-	-	-
<i>Diatoma hyemalis</i>	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Diatoma tenuis</i>	+	+	-	-	-	-	+	+	-	-	-	+	-	-	+	-	-
<i>Diatoma vulgaris</i>	-	+	+	+	-	+	-	+	-	-	+	+	-	+	+	+	-
<i>Epithemia adnata</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-
<i>Epithemia cistula</i>	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Eunotia tenella</i>	-	-	+	-	+	-	-	+	-	-	-	-	-	-	+	+	-
<i>Fragilaria capucina</i>	+	+	+	+	+	-	+	+	-	+	-	+	+	-	+	+	+
<i>Fragilaria dilatata</i>	+	-	-	-	-	-	-	+	-	-	-	-	-	-	+	-	+
<i>Ulnaria ulna</i>	+	+	-	+	+	+	-	+	-	-	-	+	+	-	+	-	+
<i>Gomphonema gracile</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-
<i>Gomphonema minutum</i>	-	-	-	-	-	+	-	-	-	+	-	-	-	-	-	-	-
<i>Gomphonema parvulum</i>	-	-	+	+	+	+	+	-	-	+	+	+	+	-	-	+	+
<i>Gomphonema pseudoaugur</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-
<i>Gomphonema truncatum</i>	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-

Appendix E Continue

Species	A3	A5	A6	A8	A9	A10	A11	A12	A13	A15	A16	A17	A18	A19	A21	R1	Bak 14
<i>Melosira lineata</i>	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula capitatoradiata</i>	-	-	-	-	+	-	+	+	-	-	-	+	+	-	-	+	-
<i>Navicula cryptonella</i>	+	+	-	+	+	-	-	-	-	-	-	-	-	-	-	+	-
<i>Navicula halophila</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-	-
<i>Navicula jaagii</i>	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	+	-
<i>Navicula lanceolata</i>	-	-	-	+	-	+	-	-	-	-	-	+	-	-	-	+	-
<i>Navicula oppugnata</i>	-	-	-	-	-	-	-	-	-	+	+	-	-	+	-	-	-
<i>Navicula phyllepta</i>	-	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-
<i>Navicula radians</i>	-	-	-	-	-	+	-	-	-	-	-	-	+	-	-	-	+
<i>Navicula slesvicensis</i>	-	-	-	-	-	+	+	-	-	-	-	-	+	-	-	-	-
<i>Nitzschia umbonata</i>	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-
<i>Stauroneis phoenicenteron</i>	+	+	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Surirella amphioxys</i>	-	-	-	+	-	-	-	-	-	-	-	-	-	-	-	-	-
<i>Surirella minuta</i>	-	-	-	-	-	-	-	-	-	+	-	-	-	-	-	-	-

+ and – indicate presence and absence of species. For station codes, see Table 5.1

Appendix F List of all phytoplankton taxa recorded during the study period

Bacillariophyta (Bac)	
Species	Code
<i>Amphipleura lindheimeri</i> Grunow	Amli
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	Amov
<i>Amphora subcapitata</i> (Kisselew) Hustedt	Amsu
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	Augr
<i>Cocconeis placentula</i> Ehrenberg	Copl
<i>Cocconeis placentula</i> var. <i>lineata</i> (Ehrenberg) v	Copll
<i>Cocconeis placentula</i> var. <i>Placentula</i> Ehrenberg	Coplv
<i>Cyclotella cyclopuncta</i> Håkansson & J.R.Carter	Cycy
<i>Cyclotella dallasiana</i> W.Smith	Cyda
<i>Cyclotella fottii</i> Hustedt in Huber-Pestalozzi	Cyfr
<i>Cyclotella iris</i> Brun & Héribaude-Joseph	Cyir
<i>Cyclotella krammeri</i> Håkansson	Cykr
<i>Cyclotella meneghiniana</i> Kützing	Cyme
<i>Cyclotella ocellata</i> Pantocsek	Cyoc
<i>Cyclotella plitvicensis</i> Hustedt	Cypl
<i>Cyclotella radiosa</i> (Grunow) Lemmermann	Cyra
<i>Cymatopleura solea</i> (Brébisson) W.Smith	Cyso
<i>Cymbella affinis</i> Kützing	Cyaf
<i>Cymbella amphicephala</i> Nägeli in Kützing	Cyam
<i>Cymbella aspera</i> (Ehrenberg) Cleve	Cyas
<i>Cymbella caespitosa</i> (Kützing) Brun	Cyca
<i>Cymbella cistula</i> (Ehrenberg) O.Kirchner	Cyci
<i>Cymbella cymbiformis</i> C.Agardh	Cycym
<i>Cymbella gracilis</i> (Hustedt) Krammer	Cygr
<i>Cymbella hantzschiana</i> Krammer	Cyha
<i>Cymbella laevis</i> Nägeli in Rabenhorst	Cyla
<i>Cymbella minuta</i> Hilse in Rabenhorst	Cymi
<i>Cymbella naviculiformis</i> (Auerswald ex Heiberg) Cleve	Cyna
<i>Cymbella tumida</i> (Brébisson) van Heurck	Cytu
<i>Denticula elegans</i> Kützing	Deel
<i>Denticula valida</i> (Pedicino) Grunow in van Heurck	Deva
<i>Diatoma tenue</i> C.Agardh	Dite
<i>Diatoma vulgare</i> Bory	Divu
<i>Epithemia frickei</i> Krammer	Epfr
<i>Epithemia sorex</i> Kützing	Epso
<i>Epithemia turgida</i> (Ehrenberg) Brun	Eptu
<i>Eunotia glacialis</i> Meister	Eugl
<i>Eunotia tenella</i> (Grunow) Hustedt	Eute
<i>Ulnaria biceps</i> (Kützing) Compère	Frbi
<i>Fragilaria capucina</i> Desmazières	Frca
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	Frdi

Appendix F continue

Bacillariophyta (Bac)	
Species	Code
<i>Fragilaria fasciculata</i> (C.Agardh) Lange-Bertalot	Frfa
<i>Fragilaria incognita</i> Reichardt	Frin
<i>Fragilaria leptostauron</i> (Ehrenberg) Hustedt	Frle
<i>Fragilaria parasitica</i> (W.Smith) Grunow	Frpa
<i>Fragilaria acus</i> Lange-Bertalot	Frac
<i>Ulnaria biceps</i> (Kützing) Compère	Frbi
<i>Gomphonema acuminatum</i> Ehrenberg	Goac
<i>Gomphonema angustatum</i> (Kützing) Rabenhorst	Goan
<i>Gomphonema angustum</i> C.Agardh	Goang
<i>Gomphonema augur</i> Ehrenberg	Goau
<i>Gomphonema gracile</i> Ehrenberg	Gogr
<i>Gomphonema mammilla</i> Ehrenberg	Goma
<i>Gomphonema minutum</i> (C.Agardh) C.Agardh	Gomi
<i>Gomphonema parvulum</i> Kützing	Gopa
<i>Gomphonema pseudoaugur</i> Lange-Bertalot	Gops
<i>Gomphonema subtile</i> Ehrenberg	Gosu
<i>Gomphonema truncatum</i> Ehrenberg	Gotr
<i>Gyrosigma acuminatum</i> (Kützing) Rabenhorst	Gyac
<i>Gyrosigma attenuatum</i> (Kützing) Rabenhorst	Gyat
<i>Hantzschia distinctepunctata</i> var. <i>circuligera</i> Compère	Hadi
<i>Melosira lineata</i> (Dillwyn) C.Agardh	Meli
<i>Melosira varians</i> C.Agardh	Meva
<i>Navicula angusta</i> Grunow	Naan
<i>Navicula bryophila</i> J.B.Petersen	Nabr
<i>Navicula capitatoradiata</i> H.Germain	Naca
<i>Navicula cari</i> Ehrenberg	Nacar
<i>Navicula cincta</i> (Ehrenberg) Ralfs in Pritchard	Naci
<i>Navicula cryptocephala</i> Kützing	Nacr
<i>Navicula cryptonella</i> Lange-Bertalot	Nacry
<i>Navicula expecta</i> S.L.VanLandingham	Naex
<i>Navicula gregaria</i> Donkin	Nagr
<i>Navicula halophila</i> (Grunow) Cleve	Naha
<i>Navicula lanceolata</i> (C.Agardh) Kützing	Nala
<i>Navicula phyllepta</i> Kützing	Naph
<i>Navicula porifera</i> Hustedt	Napo
<i>Navicula pseudomutica</i> Hustedt	Naps
<i>Navicula pupula</i> Kützing	Napu
<i>Navicula radians</i> Héribaude-Joseph	Nara
<i>Navicula radiosa</i> Kützing	Narad
<i>Navicula slesvicensis</i> Grunow	Nasl
<i>Navicula soehrensensis</i> Krasske	Naso

Appendix F continue

Bacillariophyta (Bac)	
Species	Code
<i>Navicula tripunctata</i> (O.F.Müller) Bory	Natr
<i>Navicula trivialis</i> Lange-Bertalot	Natri
<i>Navicula veneta</i> Kützing	Nave
<i>Nitzschia angustata</i> (W.Smith) Grunow	Nian
<i>Nitzschia calida</i> Grunow	Nica
<i>Nitzschia flexa</i> Schumann	Nifl
<i>Nitzschia palea</i> (Kützing) W.Smith	Nipa
<i>Nitzschia recta</i> Hantzsch ex Rabenhorst	Nire
<i>Nitzschia sigma</i> (Kützing) W.Smith	Nisi
<i>Nitzschia solita</i> Hustedt	Niso
<i>Rhopalodia gibba</i> (Ehrenberg) Otto Müller	Rhgi
<i>Rhopalodia gibba</i> var. <i>minuta</i> Krammer	Rhgiv
<i>Stauroneis anceps</i> Ehrenberg	Stan
<i>Stauroneis anceps</i> var. <i>siberica</i> Grunow	Stanv
<i>Stauroneis nobilis</i> Schumann	Stno
<i>Surirella minuta</i> Brébisson	Sumi
<i>Surirella ovalis</i> Brébisson	Suov
<i>Ulnaria ulna</i> (Nitzsch) Compère	Ulul
Chlorophyta (Chl)	
<i>Botryococcus braunii</i> Kützing	Bobr
<i>Botryosphaerella sudetica</i> (Lemmermann) P.C.Silva	Bosu
<i>Chlamydomonas crassa</i> H.R.Christen	Chcr
<i>Chlamydomonas debaryana</i> Goroschankin	Chde
<i>Chlamydomonas globosa</i> J.W.Snow	Chgl
<i>Chlamydomonas lapponica</i> Skuja	Chla
<i>Coelastrum astroideum</i> De Notaris	Coas
<i>Coelastrum microporum</i> Nägeli	Comi
<i>Eudorina elegans</i> Ehrenberg	Euel
<i>Geminella interrupta</i> Turpin	Gein
<i>Geminella ordinate</i> (West & G.S.West) Heering	Geor
<i>Monoraphidium arcuatum</i> (Korshikov) Hindák	Moar
<i>Monoraphidium contortum</i> (Thuret) Komárková	Moco
<i>Mougeotia boodlei</i> (West & G.S West) Collins	Mobo
<i>Mougeotia nummuloides</i> (Hassall) De Toni	Monu
<i>Mougeotia parvula</i> Hassall	Mopa
<i>Mougeotia pulchella</i> (Wittrock) De Toni	Mopu
<i>Mougeotia quadrangulata</i> Hassall	Moqu
<i>Mougeotia scalaris</i> Hassall	Mosc
<i>Oocystis borgei</i> J.W.Snow	Oobo
<i>Oocystis parva</i> West & G.S.West	Oopa
<i>Pandorina morum</i> (O.F.Müller) Bory	Pamo

Appendix F continue

Chlorophyta (Chl)	
Species	Code
<i>Pediastrum boryanum</i> (Turpin) Meneghini	Pebo
<i>Pediastrum duplex</i> Meyen	Pedu
<i>Pediastrum simplex</i> Meyen	Pesi
<i>Pediastrum tetras</i> (Ehrenberg) Ralfs	Pete
<i>Scenedesmus abundans</i> (O.Kirchner) Chodat	Scab
<i>Scenedesmus communis</i> E.Hegewald	Scco
<i>Scenedesmus dimorphus</i> (Turpin) Kützing	Scdi
<i>Scenedesmus ellipticus</i> Corda	Scel
<i>Scenedesmus falcatus</i> Chodat	Scfa
<i>Scenedesmus longispina</i> R.Chodat	Sclo
<i>Scenedesmus obliquus</i> (Turpin) Kützing	Scob
<i>Spirogyra dubia</i> Kützing	Spdu
<i>Spirogyra elongate</i> (Vaucher) Kützing	Spel
<i>Spirogyra grevilleana</i> (Hassall) Kützing	Spgr
<i>Spirogyra inflate</i> (Vaucher) Dumortier	Spin
<i>Spirogyra weberi</i> Kützing	Spwe
<i>Tetraëdron minimum</i> (A.Braun) Hansgirg	Temi
<i>Treubaria setigera</i> (W.Archer) G.M.Smith	Trse
<i>Ulothrix subconstricta</i> G.S.West	Ulsu
Cyanobacteria (Cya)	
<i>Anabaena catenula</i> Kützing ex Bornet & Flahault	Anca
<i>Anabaena planctonica</i> Brunnthaler	Anpl
<i>Anabaena verrucosa</i> J.B.Petersen	Anve
<i>Aphanizomenon issatschenkoi</i> (Usacev) Proshkina-Lavrenko	Apis
<i>Calothrix parietina</i> Thuret ex Bornet & Flahault	Capa
<i>Chroococcopsis chroococcoides</i> (F.E.Fritsch) K & Anagnostidis	Chch
<i>Chroococcus disperses</i> (Keissler) Lemmermann	Chdi
<i>Chroococcus turgidus</i> (Kützing) Nägeli	Chtu
<i>Chroodactylon ornatum</i> (C.Agardh) Basson	Chor
<i>Merismopedia glauca</i> (Ehrenberg) Kützing	Megl
<i>Microcystis aeruginosa</i> (Kützing) Kützing	Miae
<i>Microcystis flosaquae</i> (Wittrock) Kirchner	Mifl
<i>Nostoc linckia</i> Bornet ex Bornet & Flahault	Noli
<i>Oscillatoria agardhii</i> Gomont	Osag
<i>Oscillatoria brevis</i> Kützing ex Gomont	Osbr
<i>Oscillatoria cortiana</i> Meneghini ex Gomont	OscO
<i>Oscillatoria limosa</i> C.Agardh ex Gomont	Osli
<i>Oscillatoria tenuis</i> C.Agardh ex Gomont	Oste
<i>Pseudanabaena catenata</i> Lauterborn	Psca
<i>Snowella lacustris</i> (Chodat) Komárek & Hindák	Snla

Appendix F continue

Miozoa (Mio)	
<i>Ceratium furcoides</i> (Levander) Langhans	Cefu
<i>Ceratium hirundinella</i> (O.F.Müller) Dujardin	Cehi
<i>Peridiniopsis cunningtonii</i> Lemmermann	Pecu
<i>Peridinium cinctum</i> (O.F.Müller) Ehrenberg	Peci
<i>Peridinium lomnickii</i> Woloszynska	Pelo
<i>Peridinium willei</i> Huitfeldt-Kaas	Pewi
Charophyta (Cha)	
<i>Cosmarium punctulatum</i> Brébisson	Copu
<i>Cosmarium subcrenatum</i> Hantzsch in Rabenhorst	Cosu
<i>Closterium acerosum</i> Ehrenberg ex Ralfs	Clac
<i>Closterium aciculare</i> T.West	Claci
<i>Closterium diana</i> Ehrenberg ex Ralfs	Cldi
<i>Closterium littorale</i>	Clli
<i>Closterium moniliferum</i> Ehrenberg ex Ralfs	Clmo
<i>Closterium parvulum</i> Nägeli	Clpa
<i>Staurostrum chaetoceras</i> (Schröder) G.M.Smith	Stch
<i>Staurostrum cingulum</i> (West & G.S.West) G.M.Smith	Stci
<i>Staurostrum crenulatum</i> (Nägeli) Delponte	Ster
<i>Zygnema pectinatum</i> (Vaucher) C.Agardh	Zype
Euglenozoa (Eug)	
<i>Euglena acus</i> (O.F.Müller) Ehrenberg	Euac
<i>Euglena anabaena</i> Mainx	Euan
<i>Euglena chlamydomorpha</i> Mainx	Euch
<i>Euglena geniculata</i> Dujardin	Euge
<i>Euglena oblonga</i> F.Schmitz	Euob
<i>Euglena oxyuris</i> Schmarda	Euox
<i>Euglena proxima</i> P.A.Dangeard	Eupr
<i>Euglena viridis</i> (O.F.Müller) Ehrenberg	Euvi
<i>Phacus caudatus</i> Hübner	Phca
<i>Phacus longicauda</i> (Ehrenberg) Dujardin	Phlo
<i>Phacus parvulus</i> G.A.Klebs	Phpa
<i>Trachelomonas cervicula</i> Stokes	Trce
<i>Trachelomonas granulosa</i> Playfair	Trgr
<i>Trachelomonas hispida</i> (Perty) F.Stein	Trhi
<i>Trachelomonas pseudofelix</i> Deflandre	Trps
<i>Trachelomonas scabra</i> Playfair	Trsc
<i>Trachelomonas zorensis</i> Deflandre	Trzo
Cryptophyta (Cry)	
<i>Campylomonas reflexa</i> (M.Marsson) D.R.A.Hill	Care
<i>Cryptomonas erosa</i> Ehrenberg	Cyer
<i>Cryptomonas marssonii</i> Skuja	Cyma

Appendix F continue

<i>Cryptomonas ovata</i> Ehrenberg	Cyov
<i>Cryptomonas rostratiformis</i> Skuja	Cyro
Ochrophyta (Och)	
Species	Code
<i>Dinobryon divergens</i> O.E.Imhof	Didi



Appendix G List of species found at Lake Gölhisar

Species	Su14	Fa14	Sp15	Su15
<i>Clevamphora ovalis</i> (Kützing) Mereschkowsky	-	+	+	+
<i>Anabaena catenula</i> Kützing ex Bornet & Flahault	-	-	-	+
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	-	-	-	+
<i>Botryococcus braunii</i> Kützing	-	-	+	+
<i>Ceratium furcoides</i> (Levander) Langhans	-	-	-	+
<i>Chlamydomonas globosa</i> J.W.Snow	-	-	+	+
<i>Chroococcus dispersus</i> (Keissler) Lemmermann	-	-	+	-
<i>Chroococcus turgidus</i> (Kützing) Nägeli	-	-	-	+
<i>Cocconeis placentula</i> Ehrenberg	-	+	+	+
<i>Coelastrum astroideum</i> De Notaris	-	-	-	+
<i>Cryptomonas marssonii</i> Skuja	-	-	-	+
<i>Cryptomonas ovata</i> Ehrenberg	+	+	+	+
<i>Cyclotella ocellata</i> Pantocsek	-	+	+	+
<i>Cymbella excisa</i> Kützing	-	+	+	+
<i>Denticula elegans</i> Kützing	-	-	-	+
<i>Diatoma tenuis</i> C.Agardh	-	+	+	-
<i>Dinobryon divergens</i> O.E.Imhof	-	-	+	-
<i>Epithemia sorex</i> Kützing	-	-	-	+
<i>Epithemia turgida</i> var. <i>granulata</i> (Ehrenberg) Brun	-	-	-	+
<i>Eudorina elegans</i> Ehrenberg	-	-	+	-
<i>Euglena acus</i> (O.F.Müller) Ehrenberg	+	-	+	+
<i>Euglena chlamydophora</i> Mainx	-	-	-	+
<i>Euglena oblonga</i> F.Schmitz	-	-	-	+
<i>Euglena proxima</i> P.A.Dangeard	+	-	-	-
<i>Fragilaria capucina</i> Desmazières	-	+	+	-
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	+	-	+	-
<i>Fragilaria famelica</i> (Kützing) Lange-Bertalot	-	+	-	-
<i>Fragilaria incognita</i> Reichardt	-	+	-	-
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) Lange-Bertalot	-	+	+	+
<i>Gomphonema gracile</i> Ehrenberg	-	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	+	+	-
<i>Gomphonema truncatum</i> Ehrenberg	-	-	+	+
<i>Merismopedia glauca</i> (Ehrenberg) Kützing	+	-	-	-
<i>Microcystis aeruginosa</i> (Kützing) Kützing	+	-	+	-
<i>Monoraphidium arcuatum</i> (Korshikov) Hindák	-	-	-	+
<i>Monoraphidium contortum</i> (Thuret) K-Legnerová	-	-	-	+
<i>Mougeotia boodlei</i> (West & G.S West) Collins	-	-	-	+
<i>Mougeotia pulchella</i> (Wittrock) De Toni	-	+	-	-
<i>Mougeotia quadrangulata</i> Hassall	-	-	-	+
<i>Mougeotia scalaris</i> Hassall	-	-	-	+
<i>Navicula cincta</i> (Ehrenberg) Ralfs	-	-	+	-
<i>Navicula cryptocephala</i> Kützing	+	-	-	-

Appendix G Continue

Species	Su14	Fa14	Sp15	Su15
<i>Navicula lanceolata</i> (C.Agardh) Kützing	-	-	+	+
<i>Navicula oblonga</i> (Kützing) Kützing	-	-	+	-
<i>Navicula radians</i> Héribaude-Joseph	-	-	-	+
<i>Navicula radiosa</i> Kützing	+	-	+	-
<i>Navicula slesvicensis</i> Grunow in van Heurck	-	+	-	-
<i>Navicula tripunctata</i> (O.F.Müller) Bory	-	-	+	-
<i>Nitzschia brevissima</i> Grunow in van Heurck	-	-	+	-
<i>Nitzschia palea</i> (Kützing) W.Smith	-	-	+	-
<i>Oocystis parva</i> West & G.S.West	-	-	-	+
<i>Oscillatoria limosa</i> C.Agardh ex Gomont	-	-	-	+
<i>Oscillatoria tenuis</i> C.Agardh ex Gomont	-	-	-	+
<i>Pandorina morum</i> (O.F.Müller) Bory	-	-	+	+
<i>Pediastrum tetras</i> (Ehrenberg) Ralfs	-	-	-	+
<i>Peridiniopsis cunningtonii</i> Lemmermann	+	-	-	+
<i>Peridinium willei</i> Huitfeldt-Kaas	-	-	-	+
<i>Phacus longicauda</i> (Ehrenberg) Dujardin	-	-	+	+
<i>Pseudanabaena catenata</i> Lauterborn	-	-	-	+
<i>Rhopalodia gibba</i> (Ehrenberg) Otto Müller	-	-	-	+
<i>Rhopalodia gibba</i> var. <i>minuta</i> Krammer	-	-	-	+
<i>Scenedesmus communis</i> E.Hegewald	+	-	+	+
<i>Scenedesmus falcatus</i> Chodat	-	-	-	+
<i>Scenedesmus longispina</i> R.Chodat	-	-	-	+
<i>Scenedesmus obliquus</i> (Turpin) Kützing	-	-	-	+
<i>Spirogyra grevilleana</i> (Hassall) Kützing	-	+	-	-
<i>Spirogyra inflata</i> (Vaucher) Dumortier	-	+	-	-
<i>Spirogyra weberi</i> Kützing	-	+	-	+
<i>Staurastrum crenulatum</i> (Nägeli) Delponte	-	+	-	-
<i>Stauroneis anceps</i> Ehrenberg	-	-	+	-
<i>Stauroneis anceps</i> var. <i>siberica</i> Grunow	-	-	+	-
<i>Stauroneis nobilis</i> Schumann	-	-	+	-
<i>Surirella ovalis</i> Brébisson	-	-	+	-
<i>Tetraëdron minimum</i> (A.Braun) Hansgirg	+	-	-	-
<i>Trachelomonas cervicula</i> Stokes	-	-	-	+
<i>Trachelomonas pseudofelix</i> Deflandre	-	-	+	-

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix H List of species found at Çayboğazı reservoir

Species	Su14	Fa14	Sp15	Su15
<i>Clevamphora ovalis</i> (Kützing) Mereschkowsky	-	-	+	
<i>Botryococcus braunii</i> Kützing	-	-	+	-
<i>Campylomonas reflexa</i> (M.Marsson) D.R.A.Hill	-	-	+	-
<i>Ceratium furcoides</i> (Levander) Langhans	-	+	-	-
<i>Ceratium hirundinella</i> (O.F.Müller) Dujardin	+	+	-	-
<i>Chlamydomonas globosa</i> J.W.Snow	-	-	-	+
<i>Chlamydomonas lapponica</i> Skuja	+	-	-	-
<i>Chroococcus turgidus</i> (Kützing) Nägeli	-	-	+	-
<i>Cocconeis placentula</i> Ehrenberg	-	-	+	+
<i>Cryptomonas marssonii</i> Skuja	-	-	+	-
<i>Cryptomonas ovata</i> Ehrenberg	-	-	+	-
<i>Cyclotella dallasiana</i> W.Smith	-	-	+	-
<i>Cyclotella iris</i> Brun & Héribaud-Joseph	+	+	-	+
<i>Cyclotella ocellata</i> Pantocsek	-	-	+	+
<i>Cyclotella plitvicensis</i> Hustedt	-	-	+	-
<i>Cymbella affinis</i> Kützing	-	-	+	+
<i>Cymbella caespitosa</i> (Kützing) Brun	-	-	-	+
<i>Cymbella hantzschiana</i> Krammer	-	-	-	+
<i>Diatoma vulgare</i> Bory	-	-	+	+
<i>Euglena chlamydotheca</i> Mainx	-	-	+	-
<i>Euglena oxyuris</i> Schmarda	-	-	-	+
<i>Euglena viridis</i> (O.F.Müller) Ehrenberg	-	+	-	-
<i>Eunotia tenella</i> (Grunow) Hustedt	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	-	+	+
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	-	-	+	+
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) L-Bertalot	+	+	+	+
<i>Gomphonema augur</i> Ehrenberg	-	-	+	-
<i>Gomphonema gracile</i> Ehrenberg	-	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	-	+	+
<i>Gomphonema subtile</i> Ehrenberg	-	-	-	+
<i>Melosira lineata</i> (Dillwyn) C.Agardh	-	-	+	-
<i>Monoraphidium contortum</i> (Thuret) K-Legnerová	-	-	-	+
<i>Navicula cryptocephala</i> Kützing	-	-	-	+
<i>Navicula cryptonella</i> Lange-Bertalot	+	-	-	+
<i>Navicula radians</i> Héribaud-Joseph	-	-	-	+
<i>Nitzschia angustata</i> (W.Smith) Grunow	+	-	-	+
<i>Nitzschia flexa</i> Schumann	-	-	-	+
<i>Peridiniopsis cunningtonii</i> Lemmermann	-	-	+	-
<i>Peridinium willei</i> Huitfeldt-Kaas	-	-	-	+
<i>Scenedesmus communis</i> E.Hegewald	-	-	+	-
<i>Scenedesmus ellipticus</i> Corda	-	+	-	-
<i>Scenedesmus falcatus</i> Chodat	-	-	+	-

Appendix H Continue

Species	Su14	Fa14	Sp15	Su15
<i>Surirella minuta</i> Brébisson in Kützing	-	-	+	-
<i>Trachelomonas pseudofelix</i> Deflandre	-	-	+	-
<i>Trachelomonas scabra</i> Playfair	-	-	-	+
<i>Ulothrix subconstricta</i> G.S.West	-	-	-	+

Su14=summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix I List of species found at Lake Avlan

Sepecies	Su14	Fa14	Sp15	Su15
<i>Clevamphora ovalis</i> (Kützing) Mereschkowsky	-	-	+	+
<i>Anabaena planctonica</i> Brunnthaler	-	-	-	+
<i>Botryococcus braunii</i> Kützing	-	-	+	+
<i>Ceratium furcoides</i> (Levander) Langhans	-	-	+	+
<i>Chlamydomonas globosa</i> J.W.Snow	-	-	+	-
<i>Chlamydomonas lapponica</i> Skuja	+	-	-	-
<i>Chroococcus turgidus</i> (Kützing) Nägeli	-	+	-	+
<i>Chroodactylon ornatum</i> (C.Agardh) Basson	-	-	-	+
<i>Closterium aciculare</i> T.West	-	+	-	-
<i>Cocconeis placentula</i> Ehrenberg	-	-	+	+
<i>Cocconeis placentula</i> var. <i>placentula</i> Ehrenberg	-	-	-	+
<i>Coelastrum astroideum</i> De Notaris	-	-	-	+
<i>Cosmarium punctulatum</i> Brébisson	-	-	-	+
<i>Cryptomonas ovata</i> Ehrenberg	+	+	+	+
<i>Cryptomonas rostratiformis</i> Skuja	-	+	-	-
<i>Cyclotella iris</i> Brun & Héribaude-Joseph	+	-	-	+
<i>Cyclotella ocellata</i> Pantocsek	-	-	-	+
<i>Cymbella excisa</i> Kützing	-	+	+	+
<i>Cymbella cymbiformis</i> C.Agardh	-	+	-	-
<i>Cymbella gracilis</i> (Rabenhorst) Cleve	-	-	+	-
<i>Cymbella laevis</i> Nägeli in Rabenhorst	-	-	+	-
<i>Denticula elegans</i> Kützing	-	-	+	-
<i>Diatoma vulgare</i> Bory	-	-	+	+
<i>Epithemia sorex</i> Kützing	+	-	-	+
<i>Euglena acus</i> (O.F.Müller) Ehrenberg	-	-	-	+
<i>Euglena anabaena</i> Mainx	-	-	+	-
<i>Euglena chlamydomphora</i> Mainx	-	+	+	+
<i>Euglena oblonga</i> F.Schmitz	-	-	-	+
<i>Euglena oxyuris</i> Schmarda	-	-	-	+
<i>Euglena proxima</i> P.A.Dangeard	+	-	-	-
<i>Euglena viridis</i> (O.F.Müller) Ehrenberg	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	+	+	+
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	-	-	-	+
<i>Fragilaria parasitica</i> (W.Smith) Grunow	-	-	-	+
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) L-Bertalot	+	+	-	+
<i>Geminella ordinata</i> (West & G.S.West) Heering	-	-	-	+
<i>Gomphonema angustum</i> C.Agardh	-	+	-	-
<i>Gomphonema minutum</i> (C.Agardh) C.Agardh	-	-	+	-
<i>Gomphonema parvulum</i> Kützing	-	+	+	+
<i>Gomphonema pseudoaugur</i> Lange-Bertalot	-	+	-	-
<i>Melosira lineata</i> (Dillwyn) C.Agardh	-	-	-	+
<i>Microcystis aeruginosa</i> (Kützing) Kützing	+	-	+	+

Appendix I Continue

Species	Su14	Fa14	Sp15	Su15
<i>Monoraphidium arcuatum</i> (Korshikov) Hindák	-	-	-	+
<i>Mougeotia boodlei</i> (West & G.S West) Collins	-	-	-	+
<i>Navicula capitatoradiata</i> H.Germain	-	-	-	+
<i>Navicula cryptocephala</i> Kützing	+	+	-	-
<i>Navicula lanceolata</i> (C.Agardh) Kützing	-	+	-	-
<i>Navicula ordinaria</i> Hustedt	-	+	-	-
<i>Navicula phyllepta</i> Kützing	-	-	-	+
<i>Navicula radiosa</i> Kützing	-	-	+	-
<i>Navicula slesvicensis</i> Grunow	-	-	-	+
<i>Navicula tripunctata</i> (O.F.Müller) Bory	-	-	+	-
<i>Nitzschia flexa</i> Schumann	-	-	-	+
<i>Nitzschia recta</i> Hantzsch ex Rabenhorst	-	+	-	-
<i>Nitzschia sigma</i> (Kützing) W.Smith	-	+	-	-
<i>Nostoc linckia</i> Bornet ex Bornet & Flahault	-	-	-	+
<i>Oocystis parva</i> West & G.S.West	-	-	-	+
<i>Pandorina morum</i> (O.F.Müller) Bory	-	-	-	+
<i>Pediastrum duplex</i> Meyen	-	+	-	-
<i>Pediastrum simplex</i> Meyen	-	+	-	-
<i>Pediastrum tetras</i> (Ehrenberg) Ralfs	-	-	-	+
<i>Peridiniopsis cunningtonii</i> Lemmermann	-	-	-	+
<i>Phacus longicauda</i> (Ehrenberg) Dujardin	-	-	+	+
<i>Phacus parvulus</i> G.A.Klebs	-	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	-	-	-	+
<i>Scenedesmus communis</i> E.Hegewald	-	+	+	-
<i>Scenedesmus ellipticus</i> Corda	-	-	-	+
<i>Snowella lacustris</i> (Chodat) Komárek & Hindák	-	-	-	+
<i>Staurostrum cingulum</i> (West & G.S.West) Smith	-	-	-	+
<i>Trachelomonas cervicula</i> Stokes	-	-	-	+
<i>Trachelomonas pseudofelix</i> Deflandre	-	-	+	-
<i>Zygnema pectinatum</i> (Vaucher) C.Agardh	-	-	-	+

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix J List of species found at Geyik reservoir.

Species	Su14	Fa14	Sp15	Su15
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	-	-	-	+
<i>Anabaena catenula</i> Kützing ex Bornet & Flahault	-	-	-	+
<i>Anabaena verrucosa</i> J.B.Petersen	-	-	+	-
<i>Aphanizomenon issatschenkoi</i> (U.) P-Lavrenko	-	-	-	+
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	+	+	-	+
<i>Botryococcus braunii</i> Kützing	-	-	-	+
<i>Calothrix parietina</i> Thuret ex Bornet & Flahault	-	-	-	+
<i>Ceratium furcoides</i> (Levander) Langhans	-	-	-	+
<i>Ceratium hirundinella</i> (O.F.Müller) Dujardin	+	-	-	-
<i>Chlamydomonas globosa</i> J.W.Snow	-	-	+	+
<i>Chroococcus turgidus</i> (Kützing) Nägeli	-	-	+	+
<i>Closterium acerosum</i> Ehrenberg ex Ralfs	-	-	-	+
<i>Cocconeis placentula</i> Ehrenberg	-	-	+	+
<i>Cocconeis placentula</i> var. <i>placentula</i> Ehrenberg	-	-	+	-
<i>Coelastrum astroideum</i> De Notaris	-	-	+	+
<i>Cosmarium punctulatum</i> Brébisson	-	-	-	+
<i>Cryptomonas erosa</i> Ehrenberg	+	-	-	-
<i>Cryptomonas ovata</i> Ehrenberg	-	-	+	+
<i>Cyclotella meneghiniana</i> Kützing	-	+	-	-
<i>Cyclotella ocellata</i> Pantocsek	-	-	+	+
<i>Cylindrospermum muscicola</i> K ex B & Flahault	-	-	+	-
<i>Cymbella affinis</i> Kützing	-	-	-	+
<i>Diatoma vulgare</i> Bory	-	-	-	+
<i>Euglena chlamydotheca</i> Mainx	-	-	+	-
<i>Euglena oxyuris</i> Schmarda	-	-	-	+
<i>Eunotia glacialis</i> Meister	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	-	+	+
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	-	-	+	-
<i>Fragilaria incognita</i> Reichardt	-	-	-	+
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) L-Bertalot	+	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	-	-	+
<i>Melosira lineata</i> (Dillwyn) C.Agardh	-	-	+	-
<i>Melosira varians</i> C.Agardh	-	-	-	+
<i>Microcystis aeruginosa</i> (Kützing) Kützing	-	+	-	+
<i>Monoraphidium arcuatum</i> (Korshikov) Hindák	-	-	-	+
<i>Mougeotia boodlei</i> (West & G.S West) Collins	-	-	-	+
<i>Mougeotia nummuloides</i> (Hassall) De Toni	-	-	+	-
<i>Navicula capitatoradiata</i> H.Germain	-	-	-	+
<i>Navicula cryptonella</i> Lange-Bertalot	+	-	-	-
<i>Navicula lanceolata</i> (C.Agardh) Kützing	-	-	+	-
<i>Navicula oppugnata</i> Hustedt	+	-	-	-
<i>Navicula slesvicensis</i> Grunow in van Heurck	-	-	-	+

Appendix J Continue

Species	Su14	Fa14	Sp15	Su15
<i>Navicula trivialis</i> Lange-Bertalot	+	-	-	-
<i>Navicula veneta</i> Kützing	-	-	+	-
<i>Nitzschia flexa</i> Schumann	-	-	-	+
<i>Oscillatoria limosa</i> C.Agardh ex Gomont	-	-	+	+
<i>Oscillatoria tenuis</i> C.Agardh ex Gomont	+	+	+	-
<i>Pandorina morum</i> (O.F.Müller) Bory	+	-	-	-
<i>Pediastrum duplex</i> Meyen	-	+	-	+
<i>Pediastrum simplex</i> Meyen	-	+	-	-
<i>Pediastrum tetras</i> (Ehrenberg) Ralfs	-	-	+	-
<i>Peridiniopsis cunningtonii</i> Lemmermann	-	-	-	+
<i>Peridinium lomnickii</i> Woloszynska	+	-	-	-
<i>Peridinium willei</i> Huitfeldt-Kaas	-	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	+	-	+	-
<i>Scenedesmus communis</i> E.Hegewald	+	-	+	+
<i>Scenedesmus dimorphus</i> (Turpin) Kützing	-	-	+	-
<i>Scenedesmus obliquus</i> (Turpin) Kützing	-	-	+	-
<i>Snowella lacustris</i> (Chodat) K & Hindák	-	-	-	+
<i>Spirogyra dubia</i> Kützing	+	-	-	-
<i>Spirogyra weberi</i> Kützing	-	-	-	+
<i>Staurostrum crenulatum</i> (Nägeli) Delponte	-	-	-	+
<i>Tetraëdron minimum</i> (A.Braun) Hansgirg	-	+	-	-
<i>Trachelomonas scabra</i> Playfair	-	-	-	+

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix K List of species found at Çavdır reservoir

Species	Su14	Fa14	Sp15	Su15
<i>Amphipleura lindheimeri</i> Grunow	-	-	+	-
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	-	-	+	+
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	-	-	+	-
<i>Botryococcus braunii</i> Kützing	-	-	+	+
<i>Ceratium furcoides</i> (Levander) Langhans	-	-	+	+
<i>Ceratium hirundinella</i> (O.F.Müller) Dujardin	+	-	+	+
<i>Chlamydomonas globosa</i> J.W.Snow	-	-	-	+
<i>Chroococcus turgidus</i> (Kützing) Nägeli	-	-	+	+
<i>Cocconeis placentula</i> Ehrenberg 1838	+	-	-	-
<i>Cocconeis placentula</i> var. <i>placentula</i> Ehrenberg	-	-	-	+
<i>Cryptomonas marssonii</i> Skuja	-	-	+	-
<i>Cryptomonas ovata</i> Ehrenberg	-	-	+	+
<i>Cryptomonas rostratiformis</i> Skuja	-	-	+	-
<i>Cyclotella iris</i> Brun & Héribaud-Joseph	+	+	-	-
<i>Cyclotella ocellata</i> Pantocsek	-	-	+	+
<i>Cymbella affinis</i> Kützing	-	-	-	+
<i>Denticula elegans</i> Kützing	-	-	-	+
<i>Diatoma vulgaris</i> Bory	-	-	+	-
<i>Dinobryon divergens</i> O.E.Imhof	-	-	+	-
<i>Eudorina elegans</i> Ehrenberg	-	-	-	+
<i>Euglena acus</i> (O.F.Müller) Ehrenberg	-	-	-	+
<i>Eunotia tenella</i> (Grunow) Hustedt	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	-	+	+
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) L-Bertalot	-	+	+	+
<i>Geminella interrupta</i> Turpin	+	+	-	+
<i>Gomphonema gracile</i> Ehrenberg	-	-	-	+
<i>Gomphonema mammilla</i> Ehrenberg	-	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	-	-	+
<i>Gomphonema truncatum</i> Ehrenberg	-	-	+	-
<i>Gyrosigma acuminatum</i> (Kützing) Rabenhorst	-	-	-	+
<i>Gyrosigma attenuatum</i> (Kützing) Rabenhorst	-	-	+	-
<i>Hantzschia distinctepunctata</i> Compère	-	-	+	-
<i>Melosira lineata</i> (Dillwyn) C.Agardh	-	-	+	-
<i>Melosira varians</i> C.Agardh	-	-	+	-
<i>Merismopedia glauca</i> (Ehrenberg) Kützing	+	-	-	+
<i>Microcystis aeruginosa</i> (Kützing) Kützing	-	+	-	-
<i>Microcystis flosaquae</i> (Wittrock) Kirchner	-	-	-	+
<i>Navicula phyllepta</i> Kützing	+	-	-	-
<i>Navicula radiosa</i> Kützing	-	-	+	-
<i>Navicula slesvicensis</i> Grunow in van Heurck	-	-	-	+

Appendix K Continue

Species	Su14	Fa14	Sp15	Su15
<i>Navicula trivialis</i> Lange-Bertalot	-	-	+	-
<i>Nitzschia flexa</i> Schumann	-	-	+	-
<i>Nitzschia solita</i> Hustedt	-	-	+	-
<i>Oscillatoria limosa</i> C.Agardh ex Gomont	-	-	-	+
<i>Pandorina morum</i> (O.F.Müller) Bory	-	-	-	+
<i>Pediastrum simplex</i> Meyen	-	+	-	-
<i>Peridinium willei</i> Huitfeldt-Kaas	-	-	-	+
<i>Phacus longicauda</i> (Ehrenberg) Dujardin	-	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	-	-	+	-
<i>Scenedesmus communis</i> E.Hegewald	+	-	+	+
<i>Spirogyra elongata</i> (Vaucher) Kützing	-	-	-	+
<i>Trachelomonas scabra</i> Playfair	-	-	-	+
<i>Trachelomonas zorensis</i> Deflandre	-	-	+	-
<i>Zygnema pectinatum</i> (Vaucher) Agardh	-	-	-	+

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix L List of species found at Toptaş reservoir

Species	Su14	Fa14	Sp15	Su15
<i>Botryococcus braunii</i> Kützing	-	-	-	+
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	+	+	+	+
<i>Ceratium furcoides</i> (Levander) Langhans	-	-	-	+
<i>Chlamydomonas debaryana</i> Goroschankin	-	-	-	+
<i>Cocconeis placentula</i> Ehrenberg	-	+	+	+
<i>Coelastrum microporum</i> Nägeli in A.Braun	-	-	-	+
<i>Cryptomonas marssonii</i> Skuja	-	-	+	-
<i>Cryptomonas ovata</i> Ehrenberg	+	+	+	+
<i>Cryptomonas rostratiformis</i> Skuja	-	-	+	-
<i>Cyclotella ocellata</i> Pantocsek	-	+	+	+
<i>Cymatopleura solea</i> (Brébisson) W.Smith	-	-	+	-
<i>Cymbella affinis</i> Kützing	-	+	-	+
<i>Cymbella amphicephala</i> Nägeli in Kützing	+	-	-	-
<i>Cymbella naviculiformis</i> (A ex Heiberg) Cleve	-	-	-	+
<i>Denticula elegans</i> Kützing	-	-	-	+
<i>Diatoma vulgare</i> Bory	-	+	-	-
<i>Euglena chlamydotheca</i> Mainx	-	-	-	+
<i>Euglena oblonga</i> F.Schmitz	-	-	-	+
<i>Euglena oxyuris</i> Schmarida	-	+	+	-
<i>Euglena proxima</i> P.A.Dangeard	+	-	-	-
<i>Eunotia glacialis</i> Meister	-	-	-	+
<i>Eunotia tenella</i> (Grunow) Hustedt	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	-	+	+
<i>Fragilaria dilatata</i> (Brébisson) L-Bertalot	-	-	+	+
<i>Fragilaria ulna</i> (Kützing) L-Bertalot	+	-	-	-
<i>Gomphonema parvulum</i> Kützing	-	+	+	-
<i>Gyrosigma attenuatum</i> (Kützing) Rabenhorst	-	-	+	-
<i>Microcystis aeruginosa</i> (Kützing) Kützing	+	+	-	-
<i>Navicula capitatoradiata</i> H.Germain	-	-	-	+
<i>Navicula cryptonella</i> Lange-Bertalot	-	-	-	+
<i>Navicula expecta</i> S.L.VanLandingham	-	-	+	-
<i>Navicula lanceolata</i> (C.Agardh) Kützing	-	+	-	-
<i>Navicula pupula</i> Kützing	-	-	-	+
<i>Navicula slesvicensis</i> Grunow	-	-	-	+
<i>Navicula veneta</i> Kützing	-	-	+	-
<i>Nitzschia calida</i> Grunow	+	-	-	-
<i>Nitzschia flexa</i> Schumann	-	-	-	+
<i>Oscillatoria cortiana</i> Meneghini ex Gomont	-	-	-	+
<i>Pediastrum boryanum</i> (Turpin) Meneghini	-	-	-	+
<i>Pediastrum duplex</i> Meyen	-	+	-	+
<i>Pediastrum simplex</i> Meyen	+	+	+	+
<i>Peridinium cinctum</i> (O.F.Müller) Ehrenberg	+	-	-	+

Appendix L Continue

Species	Su14	Fa14	Sp15	Su15
<i>Phacus longicauda</i> (Ehrenberg) Dujardin	-	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	-	-	-	+
<i>Scenedesmus communis</i> E.Hegewald	-	-	+	-
<i>Scenedesmus dimorphus</i> (Turpin) Kützing	-	-	+	-
<i>Scenedesmus ellipticus</i> Corda	-	-	-	+
<i>Staurastrum chaetoceras</i> (Schröder).Smith	-	+	-	-
<i>Staurastrum cingulum</i> (West & West) Smith	-	-	-	+
<i>Stauroneis nobilis</i> Schumann	-	-	+	-
<i>Trachelomonas cervicula</i> Stokes	-	-	-	+
<i>Trachelomonas pseudofelix</i> Deflandre	-	-	+	-

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix M List of species found at Yapraklı reservoir

Species	Su14	Fa14	Sp15	Su15
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	+	-	-	-
<i>Botryococcus braunii</i> Kützing 1849	-	-	-	+
<i>Botryosphaerella sudetica</i> (Lemmermann) Silva	-	+	-	-
<i>Clevamphora ovalis</i> (Kützing) Mereschowsky	-	-	-	+
<i>Ceratium hirundinella</i> (O.F.Müller) Dujardin	-	+	-	-
<i>Chroococcopsis chroococcoides</i> (Fritsch) K & A	-	+	-	-
<i>Chroococcus turgidus</i> (Kützing) Nägeli	-	+	-	+
<i>Closterium aciculare</i> T.West	-	+	-	-
<i>Closterium diana</i> Ehrenberg ex Ralfs	-	+	-	-
<i>Cocconeis placentula</i> Ehrenberg	-	+	+	+
<i>Cryptomonas marssonii</i> Skuja	-	-	+	-
<i>Cryptomonas ovata</i> Ehrenberg	+	+	+	+
<i>Cyclotella cyclopuncta</i> Håkansson & Carter	-	+	+	-
<i>Cyclotella fottii</i> Hustedt in Huber-Pestalozzi	-	-	-	+
<i>Cyclotella iris</i> Brun & Héribaude-Joseph	+	-	-	-
<i>Cyclotella ocellata</i> Pantocsek	-	+	+	+
<i>Cymatopleura solea</i> (Brébisson) W.Smith	-	-	-	+
<i>Cymbella affinis</i> Kützing	-	-	-	+
<i>Cymbella aspera</i> (Ehrenberg) Cleve	-	-	-	+
<i>Cymbella gracilis</i> (Rabenhorst) Cleve	-	-	-	+
<i>Cymbella tumida</i> (Brébisson) van Heurck	-	-	+	-
<i>Diatoma tenuis</i> C.Agardh	-	-	-	+
<i>Diatoma vulgare</i> Bory	-	-	-	+
<i>Dinobryon divergens</i> O.E.Imhof	-	-	-	+
<i>Euglena oxyuris</i> Schmarda	-	+	-	-
<i>Eunotia tenella</i> (Grunow) Hustedt	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	+	-	+
<i>Fragilaria leptostauron</i> (Ehrenberg) Hustedt	-	-	-	+
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) L-Bertalot	+	-	+	+
<i>Geminella interrupta</i> Turpin	-	-	-	+
<i>Gomphonema gracile</i> Ehrenberg	-	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	-	+	+
<i>Gomphonema truncatum</i> Ehrenberg	+	-	-	-
<i>Merismopedia glauca</i> (Ehrenberg) Kützing	+	-	-	+
<i>Microcystis flosaquae</i> (Wittrock) Kirchner	-	-	-	+
<i>Monoraphidium arcuatum</i> (Korshikov) Hindák	-	-	-	+
<i>Mougeotia boodlei</i> (West & G.S West) Collins	-	-	-	+
<i>Mougeotia parvula</i> Hassall	-	-	-	+
<i>Navicula cryptocephala</i> Kützing	+	-	-	-
<i>Navicula halophila</i> (Grunow) Cleve	-	+	-	-
<i>Navicula lanceolata</i> (C.Agardh) Kützing	-	+	-	+
<i>Navicula radians</i> Héribaude-Joseph	-	-	-	+

Appendix M Continue

Species	Su14	Fa14	Sp15	Su15
<i>Navicula radiosa</i> Kützing	-	-	+	-
<i>Navicula slesvicensis</i> Grunow	-	-	-	+
<i>Navicula tripunctata</i> (O.F.Müller) Bory	-	-	-	+
<i>Oocystis borgei</i> J.W.Snow	+	-	-	-
<i>Pediastrum boryanum</i> (Turpin) Meneghini	-	-	-	+
<i>Pediastrum duplex</i> Meyen	-	+	-	+
<i>Pediastrum simplex</i> Meyen	-	+	-	-
<i>Peridiniopsis cunningtonii</i> Lemmermann	-	-	+	+
<i>Peridinium cinctum</i> (O.F.Müller) Ehrenberg	+	-	-	-
<i>Peridinium willei</i> Huitfeldt-Kaas	-	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	-	-	-	+
<i>Rhopalodia gibba</i> (Ehrenberg) Otto Müller	-	-	-	+
<i>Scenedesmus communis</i> E.Hegewald	-	+	-	+
<i>Scenedesmus dimorphus</i> (Turpin) Kützing	-	-	-	+
<i>Scenedesmus ellipticus</i> Corda	+	-	-	-
<i>Staurastrum cingulum</i> (West & G.S.West) Smith	-	-	-	+
<i>Staurastrum crenulatum</i> (Nägeli) Delponte	-	+	-	-
<i>Stephanodiscus rotula</i> (Kützing) Hendey	-	-	+	-
<i>Tetraëdron minimum</i> (A.Braun) Hansgirg	+	-	-	-
<i>Trachelomonas hispida</i> (Perty) F.Stein	-	-	-	+

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix N List of species found at Osmankalfalar reservoir

Species	Su14	Fa14	Sp15	Su15
<i>Achnanthes impexa</i> Lange-Bertalot	-	-	-	-
<i>Aulacoseira granulata</i> (Ehrenberg) Simonsen	-	-	-	+
<i>Botryococcus braunii</i> Kützing	-	-	+	+
<i>Ceratium furcoides</i> (Levander) Langhans	-	-	-	+
<i>Ceratium hirundinella</i> (O.F.Müller) Dujardin	-	-	-	+
<i>Chlamydomonas crassa</i> H.R.Christen	-	-	-	+
<i>Cocconeis placentula</i> Ehrenberg	-	-	+	+
<i>Coelastrum microporum</i> Nägeli in A.Braun	+	-	-	-
<i>Cyclotella iris</i> Brun & Héribaud-Joseph	-	+	-	+
<i>Cyclotella krammeri</i> Håkansson	-	-	+	-
<i>Cyclotella ocellata</i> Pantocsek	+	-	+	+
<i>Cyclotella radiosa</i> (Grunow) Lemmermann	-	-	-	+
<i>Cymbella affinis</i> Kützing	+	-	-	+
<i>Cymbella minuta</i> Hilse in Rabenhorst	-	+	-	-
<i>Diatoma vulgare</i> Bory	-	-	-	+
<i>Dinobryon divergens</i> O.E.Imhof	-	+	-	-
<i>Eudorina elegans</i> Ehrenberg	-	-	-	+
<i>Euglena acus</i> (O.F.Müller) Ehrenberg	+	-	-	-
<i>Euglena chlamydophora</i> Mainx	-	-	+	+
<i>Euglena oxyuris</i> Schmarda 1846	-	-	-	+
<i>Euglena viridis</i> (O.F.Müller) Ehrenberg	-	+	-	-
<i>Ulnaria biceps</i> (Kützing) Compère	-	-	+	-
<i>Fragilaria capucina</i> Desmazières	-	-	-	+
<i>Fragilaria dilatata</i> (Brébisson) L-Bertalot	-	-	+	-
<i>Fragilaria parasitica</i> (W.Smith) Grunow	-	-	-	+
<i>Geminella interrupta</i> Turpin	-	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	-	-	+
<i>Microcystis aeruginosa</i> (Kützing) Kützing	+	-	+	-
<i>Navicula angusta</i> Grunow	-	-	-	+
<i>Navicula cari</i> Ehrenberg	-	-	+	-
<i>Navicula cryptocephala</i> Kützing	+	+	-	-
<i>Navicula exspecta</i> S.L.VanLandingham	-	-	-	+
<i>Navicula phyllepta</i> Kützing	-	-	-	+
<i>Navicula porifera</i> Hustedt	-	-	+	-
<i>Navicula veneta</i> Kützing	-	-	-	+
<i>Peridiniopsis cunningtonii</i> Lemmermann	-	-	+	+
<i>Phacus caudatus</i> Hübner	+	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	-	-	-	+
<i>Scenedesmus communis</i> E.Hegewald	+	+	-	+
<i>Scenedesmus dimorphus</i> (Turpin) Kützing	+	-	+	-
<i>Scenedesmus ellipticus</i> Corda	-	+	-	-
<i>Scenedesmus falcatus</i> Chodat	-	-	+	-

Appendix N Continue

Species	Su14	Fa14	Sp15	Su15
<i>Tetraëdron minimum</i> (A.Braun) Hansgirg	-	+	-	-
<i>Trachelomonas hispida</i> (Perty) F.Stein	-	-	-	+
<i>Trachelomonas pseudofelix</i> Deflandre	-	-	+	-

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix O List of species found at Lake Yazır

Species	Su14	Fa14	Sp15	Su15
<i>Botryococcus braunii</i> Kützing	-	-	+	+
<i>Chlamydomonas lapponica</i> Skuja	+	-	-	-
<i>Closterium moniliferum</i> E. ex Ralfs	-	-	-	+
<i>Cocconeis placentula</i> Ehrenberg	-	-	+	+
<i>Cocconeis placentula</i> var. <i>placentula</i>	-	-	-	+
<i>Cryptomonas marssonii</i> Skuja	-	-	+	-
<i>Cryptomonas ovata</i> Ehrenberg	+	-	+	+
<i>Cyclotella iris</i> Brun & Héribaud-Joseph	+	-	-	-
<i>Cyclotella ocellata</i> Pantocsek	-	-	+	+
<i>Cymatopleura solea</i> (Brébisson) W.Smith	-	-	+	-
<i>Cymbella affinis</i> Kützing	-	+	-	+
<i>Cymbella cistula</i> (Ehrenberg) O.Kirchner	-	-	-	+
<i>Cymbella cymbiformis</i> C.Agardh	-	-	-	+
<i>Cymbella elliptica</i> Prudent	-	-	-	-
<i>Cymbella gracilis</i> (Rabenhorst) Cleve	-	+	-	-
<i>Denticula elegans</i> Kützing	-	-	+	+
<i>Dinobryon divergens</i> O.E.Imhof	-	-	+	-
<i>Epithemia sorex</i> Kützing	-	-	-	+
<i>Epithemia turgida</i> (Ehrenberg) Brun	-	-	-	+
<i>Eudorina elegans</i> Ehrenberg	-	-	+	-
<i>Euglena chlamydophora</i> Mainx	-	-	+	-
<i>Euglena oxyuris</i> Schmarda	-	-	-	+
<i>Fragilaria capucina</i> Desmazières	-	-	+	+
<i>Fragilaria crotonensis</i> Kitton	-	-	-	-
<i>Fragilaria dilatata</i> (Brébisson) Lange-Bertalot	-	-	+	+
<i>Fragilaria ulna</i> var. <i>acus</i> (Kützing) L-Bertalot	+	+	+	+
<i>Geminella interrupta</i> Turpin	-	-	+	-
<i>Gomphonema angustatum</i> (Kützing) Rabenhorst	-	-	+	-
<i>Gomphonema gracile</i> Ehrenberg	-	-	-	+
<i>Gomphonema parvulum</i> Kützing	-	-	+	+
<i>Gomphonema truncatum</i> Ehrenberg	-	-	-	+
<i>Microcystis aeruginosa</i> (Kützing) Kützing	-	-	+	-
<i>Monoraphidium arcuatum</i> (Korshikov) Hindák	-	-	-	+
<i>Navicula bryophila</i> J.B.Petersen	-	+	-	-
<i>Navicula cryptocephala</i> Kützing	+	-	-	-
<i>Navicula gregaria</i> Donkin	-	-	-	+
<i>Navicula lanceolata</i> (Agardh) Kützing	-	-	+	+
<i>Navicula pseudomutica</i> Hustedt	-	+	-	-
<i>Navicula radiosa</i> Kützing	+	-	+	-
<i>Nitzschia flexa</i> Schumann	-	-	-	+

Appendix O Continue

Species	Su14	Fa14	Sp15	Su15
<i>Oscillatoria limosa</i> Agardh ex Gomont	-	-	-	+
<i>Oscillatoria tenuis</i> Agardh ex Gomont	-	-	-	+
<i>Pandorina morum</i> (O.F.Müller) Bory	-	-	+	-
<i>Pediastrum boryanum</i> (Turpin) Meneghini	-	-	-	-
<i>Pediastrum duplex</i> Meyen	-	-	-	-
<i>Pediastrum integrum</i> Nägeli	-	-	-	-
<i>Pediastrum simplex</i> Meyen	-	-	-	-
<i>Pediastrum tetras</i> (Ehrenberg) Ralfs	-	-	-	-
<i>Peridiniopsis cunningtonii</i> Lemmermann	-	-	+	-
<i>Peridinium cinctum</i> (O.F.Müller) Ehrenberg	+	-	-	+
<i>Peridinium lomnickii</i> Woloszynska	-	-	-	-
<i>Peridinium willei</i> Huitfeldt-Kaas	-	-	+	-
<i>Pleurosigma angulatum</i> (Queckett).Smith	-	-	-	+
<i>Pseudanabaena catenata</i> Lauterborn	+	-	+	-
<i>Rhopalodia gibba</i> (Ehrenberg) Otto Müller	-	-	-	+
<i>Scenedesmus abundans</i> (O.Kirchner) Chodat	-	-	+	-
<i>Scenedesmus communis</i> E.Hegewald	-	-	+	-
<i>Scenedesmus dimorphus</i> (Turpin) Kützing	-	-	+	-
<i>Scenedesmus falcatus</i> Chodat	-	-	+	+
<i>Scenedesmus obliquus</i> (Turpin) Kützing	-	-	+	-
<i>Staurastrum cingulum</i> (West & West) Smith	-	-	+	-
<i>Trachelomonas pseudofelix</i> Deflandre	-	-	+	-

Su14= summer 2014; Fa14=fall 2014; Sp15= spring 2015; Su15= summer 2015; + and - represent occurrence and absence respectively

Appendix P PTI, Med-PTI and Q index classes and boundaries

PTI based EQR class boundaries (Philips et al., 2013)					
Class limit	high – good	good – moderate	moderate – poor		poor – bad
EQR	0.89	0.70	0.45		0.23
Med-PTI class boundaries and EQRs (Marchetto et al., 2009)					
Class limit	high – good	good – moderate	moderate – poor		poor – bad
Med-PTI	2.77	2.45	2.13		1.81
EQR	0.89	0.79	0.69		0.59
Ecological status and Q index boundaries (Padisák et al., 2006)					
Status	Poor	Tolerable	Medium	Good	Excellent.
Q index	0–1	1–2	2–3	3–4	4–5

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EDUCATION

	Graduate school	Year
Master	University of Maradi (Niger)	2013
Bachelor	University of Niamey (Niger)	2011
High School	CES Rive droite (Niamey/Niger)	2005

Work experience

Year	Place	Enrollment
2014-2016	Turkey. Project (188.02.01)	Scholar-Research Assist.
2012-2013	Niger. High School Fogasso Maradi	Lecturer
2012-2013	Niger. High School Annur Maradi	Lecturer
2011-2013	Niger. University of Maradi	National service (Assistant)
2011-2013	Niger. High School LDB Maradi	Lecturer

PUBLICATIONS

Toudjani, A. A., Issiaka, Y., Moustapha, A. M. (2013). Typologie de l'avifaune des zones humides de Madarounfa, région de Maradi (Niger). *Journal des Sciences de l'Environnement*, **2**, 1-9.

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FOREIGN LANGUAGES

- French : both writing and oral ;
- English : both writing and oral ;
- Turkish : both writing and oral ;
- Arabic: Writing and reading;

HOBBIES

- Hobby 1: Sport ;
- Hobby 2: reading.