TRACKING THE RECENT HISTORICAL CHANGES IN TURKISH SHALLOW LAKES BASED ON A PALAEOLIMNOLOGICAL APPROACH USING DIATOMS

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ABSTRACT

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Current study provides information about the factors that influence the diatom composition of shallow lakes in Turkey. Thirty two shallow lakes were sampled with snap-shot sampling. Moreover, short cores from four small shallow lakes and three large lakes were retrieved.

The study showed that conductivity had a significant impact on shaping the fossil diatom composition in surface sediment samples and the effect of eutrophication was captured in short core samples especially after 1970s for altering the past status of Turkish shallow lakes. The complex interactions of biological remains and environmental variables were also revealed.The comparison of instrumental water-level data (40-100 years) with fossil diatom samples showed the reflection of short term water level changes in sediment cores was rather poor but the negative impact of eutrophication was hinted especially for Lake Uluabat and Lake Marmara.

The current study suggested that diatom fossils are good indicators for capturing the environmental variations and can be used in the absence of historical data for tracking the historical changes of shallow lakes in Turkey. However, to achieve clear signs of the composition alteration, a long-term perspective is needed and the complexity of multiple drivers for shaping the community structure of the diatom flora should be considered. In addition, the significant influence of conductivity and nutrients emphasizes the expected impacts of climate change on Mediterranean region.

Keywords: Diatom, palaeolimnology, shallow lakes, conductivity, eutrophication

ÖZ

TÜRKİYE SIĞ GÖLLERİNİN YAKIN GEÇMİŞ ZAMANDAKİ DURUMLARININ DİATOM FOSİLLERİ KULLANILARAK BELİRLENMESİ

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Bu tez Türkiye sığ göllerinin diatom kompozisyonunu etkileyen faktörler hakkında bilgi sunmaktadır. Veri seti oluşturmak için toplam otuziki sığ göl örneklenmiştir. Ayrıca dört küçük sığ ve üç büyük sığ göldenkısa karot örnekleri alınmıştır.

Çalışma sonucunda iletkenliğin yüzey veri setindeki diatom dağılımını anlamlı olarak etkilediği görülmüştür. Bunun yanında, kısa karot örneklerinde özellikle 1970lerden sonra ötrofikasyonun etkisi saptanmıştır. Ayrıca biyolojik kalıntılar ve çevresel değişkenler arasındaki kompleks ilişkiler ortaya çıkarılmıştır. Su seviyesi verilerinin (40-100 yıl) çökeldeki diatom fosillerinin dağılımı ile karşılaştırılması sonucu kısa dönemli su seviyesi değişimlerinin fosillerin dağılımı üzerindeki etkilerinin zayıf olduğu görülmüştür. Fakat özellikle Uluabat ve Marmara gölleri için ötrofikasyonun negatif etkisi saptanmıştır.

Bu çalışma çevresel değişimlerin tespit edilmesinde diatomların başarılı bir indikator tür olduğunu göstermiştir ve uzun dönemli izleme verilerinin olmadığı

durumlarda göllerin geçmiş durumlarının belirlenmesinde diatom fosillerinin kullanılabileceğini işaret etmektedir. Fakat diatom florasındaki kompozisyonal değişimin açıklanması sırasında uzun dönemli bir bakış açısı gereklidir ve çoklu etkenlerin kompozisyon yapısı üzerindeki etkisi göz önünde bulundurulmalıdır. Ayrıca iletkenlik ve besin tuzlarının anlamlı etkisi, iklim değişikliğinin Akdeniz iklim kuşağında yaratması beklenen olumsuz etkilerine dikkat çekmektedir.

Anahtar kelimeler: Diatom, palaeolimnology, sığ göller, iletkenlik, ötrofikasyon.

To my family and friends

*

We're all mad here

(Lewis Carroll, Alice in Wonderland)

*

In the world we live in, what we know and what we don't know are like Siamese twins, inseparable, existing in a state of confusion.

(Haruki Murakami, Sputnik Sweetheart)

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INTRODUCTION

1.1.Shallow Lakes

Freshwater ecosystems significantly influence human being and wildlife as they provide a diverse habitat for organisms and serve as a transition zone between terrestrial and aquatic ecosystems (Moss, 1998; Beklioglu et al., 2011). Additionally, they can be used for recreational, industrial, transportation and agricultural activities but foremost among these, their role as a drinking water supply is vital (Moss, 1998). Shallow waters, especially small shallow lakes are the main contributors of the world's freshwater ecosystems (Wetzel, 2001; Moss, 1998). In definition, a lake with <3 m mean depth and a water column that exposed to mixing in a continuous cycle, avoiding a stable thermal stratification, is categorised as a shallow lake (Moss, 1998). A wider littoral area covered with macrophytes due to higher light penetration is a distinct feature of the shallow lake ecosystems which distinguish them from deep lakes (Moss, 1998; Beklioglu et al., 2011). Due to their close system structure, shallow lakes are more vulnerable to the effects of human activities and climate change than deeper lakes (Padisák and Reynolds, 2003; Beklioglu et al., 2011; Gottschalk and Kahlert, 2012).

1.2. Eutrophication and Salinization in Mediterranean Freshwater Ecosystems

Freshwater ecosystems have been exposed to several threats (e.g. environmental pollution, overfishing, climate change, etc.) altering their chemistry, hydrology and community structure over the decades (Ormerod et al., 2010; Tockner et al., 2010). Among those threats, anthropogenic impacts have had the most significant negative effects on lake ecosystems (Harper, 1992) mainly b causing eutrophication (Hasler, 1947; Ulen et al., 2007), salinization (Beklioglu and Tan, 2008) and biodiversity loss (Baastrup-Spohr et al., 2013).

Eutrophication is a well-known phenomenon for shallow lake ecosystems, occurring when the nutrient levels, especially phosphorus and nitrogen are over the threshold level (Wetzel, 2001; Elser et al., 2007). A lake can have eutrophic characteristics naturally (Hall and Smol, 1993) but more often excessive inorganic inputs from agriculture, sewage disposal and pollution (Hall and Smol, 1999; Smol and Stoermer, 2010a) are the main sources of eutrophication in freshwater ecosystems. Additionally, climate change induced impacts (less precipitation, reduced flushing rates, droughts) enhance the nutrient concentrations in lakes as well, thus triggering the trophic state to shift towards eutrophication (Smol and Stoermer, 2010a; Özen et al., 2010; Coppens et al., 2016). During the eutrophication process, the structure and the composition of the community completely alters (Brucet et al., 2010; Jeppesen et al., 2010; Meerhoff et al., 2012) and the appearance of anoxic conditions due to enhanced growth of cyanobacteria are triggered (Wagner and Adrian 2009; Kosten et al., 2012). The recovery from eutrophication is rather difficult since the loss of quality and diversity of the environment is mostly irreversible. During the years, several methods (diversion of excess nutrients, mixing, oxygenation, etc.) were developed to reduce the impacts of this phenomenon butgenerally, the application and sustainability of these methods were limited for complex freshwater environments (Chislock et al., 2013).

Increasing salinity levels in freshwater ecosystems cause a significant biodiversity loss (Søndergaard, Bjerring, and Jeppesen, 2012; Jeppesen et al., 2014) and the impacts of temperature and eutrophication might be overridden if the salinity increase is high enough to change the lake status (e.g. freshwater to brackish) (Brucet et al., 2009). The negative impact of salinization especially for lakes in Anatolian plateau is well documented (Beklioglu and Tan 2008; Beklioglu et al., 2011; Özen et al., 2010; Beklioğlu et al., 2017). Vander Laan et al. (2013) suggested that conductivity is one of the most important environmental variable that alters the biodiversity of streams in the USA and Ballot et al. (2009) revealed that Lake Naivasha, the largest freshwater lake in the Kenyan Rift Valley, suffered from cyanobacteria blooms after a significant increase in salinity levels. Moreover, increase in mortality and decrease in reproduction and growth caused by osmotic stress are other negative impacts of salinization on freshwater ecosystems (Nielsen et al., 2003; Bezirci et al., 2012).

On the other hand, the negative impact of climate change on freshwater ecosystems is crucial (Jeppesen et al., 2011). Climate change caused changes like increase in the number of cyanobacteria blooms, enhanced eutrophication due to longer growth seasons and increased temperatures are the major concerns that may affect the structure and community composition of lakes completely (Jeppesen et al., 2010; Moss, 2011; Kosten et al., 2012; Jeppesen et al., 2014). According to IPCC reports (IPCC, 2007; IPCC, 2014), Mediterranean region will be affected by global warming significantly. A 25–30% reduction in precipitation and increased evaporation together with up to 30–40% decrease in runoff waters are expected for Turkey and Spain (A. Erol and Randhir, 2012; IPCC, 2007; IPCC, 2014). Additionally, climate change enhanced salinity and hydraulic retention due to changes in water levels may cause a significantly impact on the structure of lake ecosystems in the region (Brucet et al., 2010; Zohary and Ostrovsky, 2011; Brucet et al., 2012; Jeppesen et al., 2014).

Shallow lakes in Turkey are economically valuable and they hold a good variety of species biodiversity due their topographic and climatic variability (Çakıroğlu et al., 2014; Levi et al., 2014; Şekercioğlu et al., 2011). Turkish Lakes commonly have eutrophic characteristics because of the negative impacts of water loss due to climate change, agriculture and land use (Beklioglu, Altinayar & Tan, 2006; Bucak et al., 2012; Çakıroğlu et al., 2014; Levi et al., 2014). Salinization is another important factor that shaping the community structure of the water bodies in Turkey (Beklioglu et al., 2011; Çakiroglu et al., 2014; Levi et al., 2014). According to Önol and Unal (2014), there will be a 3–4 °C temperature increase in Central Anatolian region in Turkey by 2071–2100 together with at least 10% decrease in precipitation which will significantly affect the water and nutrient balance of lakes in the area. Several studies revealed that salinization and eutrophication has already caused severe problems in Turkish shallow lakes (Beklioglu and Tan, 2008; Özen et al., 2010; Beklioglu et al., 2011; Çakıroğlu et al., 2014; Levi et al., 2014; Bucak et al., 2017). The expected impacts of the climate change may exacerbate the impacts of those stressors and cause more drastic changes for the lakes in Anatolian plateau.

1.3. Palaeolimnology

Palaeolimnology is a discipline that investigates the historical changes of lake ecosystems by combining limnology, ecology and geology (Whitmore and Riedinger-Whitmore, 2014) and it creates a link between past and present status of the lake environment. Dead material from the lake itself (autochthonous) and surrounding habitat (allochthonous) chronologically accumulate at the lake basin and it forms a stratigraphic archive (Smol, 2008) that can be used for determining the response of the lake to the changes in its environment, like hydrology, climate, vegetation etc., (Frey, 1988; Last et al., 2012).

First palaeolimnological studies started around 1920's on lake sedimentation and lamination iof sediments (Lundqvist, 1927; Nipkow, 1920) .Later on, Hutchinson

& Wollack (1940) and Pennington, (1943) contributed to the area and improved the knowledge significantly. During the 1960's, palaeolimnological studies focused on pollution caused by anthropogenic activities (Stockner and Benson, 1967; Digerfeldt 1972; Battarbee, 1973; Battarbee, 1978; Bradbury, 1975; Birks et al., 1976; Pennington et al., 1977) and acid rain became an important issue around 1980's (Battarbee, 1991; Last and Smol, 2001). Thereafter, new palaeolimnological approaches like transfer function approach has been used to quantitatively reconstruct the past environmental status of the freshwater ecosystems and it was one of the first attempts to bind a bond between past, present and future conditions of the freshwater ecosystems (Birks, 1998; Hall and Smol, 1999; Last and Smol, 2001). Moreover, while single proxies studies are still hold a significant value for palaeolimnological studies, the utilization of multi-proxy approach become more acknowledgedrecently especially for reconstructing past environments and climate change (Birks and Birks, 2006; Lotter and Birks, 2003).

Currently, palaeolimnologists have the ability of making accurate environmental reconstructions for answering questions about past and present conditions of freshwater environments. The information gathered can be used for lake management issues as well (Smol, 2008; Battarbee, 1999; Last and Smol, 2001). Moreover, palaeolimnological methods are involved for defining baselines and restoration targets, assessing, determining natural and ecological status, and identifying causes of change of lake environments (Battarbee, 1999; Smol, 2002). In palaeolimnological studies, there are three types of data sets namely calibration or training data set, environmental data set and core data set (Charles and Smol, 1994; Birks, 1995). Calibration or training set consists of the recent individuals of the selected sediment proxy collected from surface sediment samples. Preferably, calibration sets should include not less than 40 lakes, reflecting the limnological conditions of the present environment (Birks, 2012). Environmental data set consist of the environmental factors, which influence the abundance of the selected proxy and can include physical, chemical and biological variables of lake (e.g. depth, zooplankton biomass, salinity etc.) (Birks, 2012). The core data set contain the down core examination of the selected proxy. Finally, the aim of palaeolimnology is to reconstruct and interpret past environmental conditions using the combination of these three data sets (Smol, 2002; Berlung, 1986).

For palaeolimnological issues, shallow lakes contain a variety of physical, geochemical and biological proxies in their sediment accumulated in lake bottom that enables past ecosystem reconstructions (Beniston et al., 2007; Frei et al., 2006) (Table 1.1). Sediment samples are taken vertically from the lake bottom using sediment corers (gravity corers, piston corers, freeze coring) (Figure 1.1). According to the law of superposition the age of the sediment increases with depth which can be determined by dating techniques, such as ^{210}Pb and ^{14}C (Last et al., 2012).

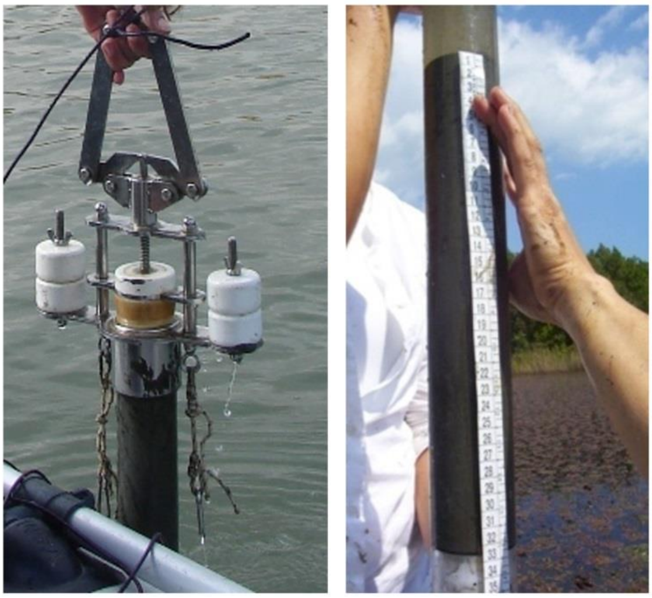

Figure 1.1 An example of gravity type Kajak Corer.

Loss on ignition (LOI) is one of the widely used chemical proxies enabling the determination of carbonate and organic contents of the sediment (Heiri, Lotter, and Lemcke, 2001) . Higher organic carbon in the sediment might be the sign of nutrient enrichment as well as water level fluctuations (Kauppila and Valpola, 2003; Korhola, Tikkanen, and Weckström, 2005). Magnetic susceptibility and X-ray diffractometry (XRD) expose the mineralogy of the sediment (Last and Smol, 2001) and the information can be later used for palaeoclimatic reconstructions (Geiss, Umbanhowar, Camill, & Banerjee, 2003). Isotopic carbon content of the sediment gives information about past productivity and nutrient changes of the surface sediments (Last and Smol, 2001). Additionally the ratio of $^{13}C/^{12}C$, $^{18}O/^{16}O$ and D/H isotopes are used in palaeoclimate reconstructions (Last and Smol, 2001)and Spheroidal Carbonaceous Particles (fly-ash components) provide information about atmospheric pollution histories of lakes (Clymo et al., 1990).

Table 1.1 Physical, geochemical and biological proxy groups in lake sediment (modified from Douglas, 2007).

Group	***Application***
Physical	
Grain Size	Sediment supply, sedimentation, and turbulance
Loss-on-ignition	Inorganic and organic composition of sediment matrix
Mineralogy and elemental composition	Sediment supply, water chemistry
Magnetic properties	Sedimentation, dating, erosion
Fluid inclusions	Aquatic paleoclimate and paleochemisitry
Fly ash and charcoal	Industrialization, fires
Geochemical	
Organic matter	Paleoproductivity
C:N	Paleoproductivity
Stable isotopes	Paleoclimate, paleoproductivity, paleotemperature
Biological	
Algae	Aquatic chemistry and microhabitats
Pigments	Aquatic chemistry and microhabitats
Insects	Aquatic chemistry and microhabitats
Zooplankton	Aquatic chemistry and microhabitats
Sponges	Aquatic chemistry and microhabitats
Pollen	Terrestrial vegetation, aquatic macrophytes
Phytoliths	Terrestrial catchment vegetation, especially grasses

Algal remains, mainly represented by diatoms, biogeochemical fossils and pigments, are commonly used for determining the impacts of pollution, eutrophication and salinization on freshwater ecosystems (Gasse, Juggins, and Khelifa, 1995; Reed, 1998; Ryves, Mcgowan, and Anderson, 2002). Sedimentary pigments (carotenoids, chlorophylls, their derivatives and other lipid-soluble pigments) have a longer preservation than other remains in the sediment and they are good indicators of primary producer community of the past environments (Last and Smol, 2001). Plant remains (plant macrofossils, pollen and conifer stomata) give information about past aquatic and terrestrial vegetation of the area as well as past water level changes and past atmospheric concentrations reconstructions(Last and Smol, 2001). Additionally, spores and fungi represent past terrestrialization and erosion (Last and Smol, 2001) while cladocera and other branchiopod crustaceans are very useful tools for palaeoenvironmental reconstructions of several environmental variables (salinity, climatic changes, trophic oscillations, acidification, and water-level changes) due to their high dispersal capacity and short reproduction cycle (Last and Smol, 2001).

1.4. Diatoms and their role in palaeolimnology

Diatoms are unicellular organisms with siliceous cell walls and yellow-brown pigments. They are categorized as photosynthesizing algae from the Division Chrysophyta, Class Bacillariophyceae. 25 % of the primary production of earth is constituted by diatoms (Werner, 1977). Since most of the diatoms do not have any specialised organelles for moving, most of them are non-motile or they have limited movement activities. Diatoms can be found both in both solidarity and colonised forms. The earliest evidence of a diatom is from early Jurassic (Rothpletz, 1986; Rothpletz, 1990) and the oldest, well preserved flora is from early Cretaceous (Hardwood andGersonde, 1990). The first diatom genus was described in 1791 as *Bacillaria* Gmelin, with *Vibrio (Bacillaria) paxallifer* used as the type by O. F. Müller.

Diatoms are divided into two orders as ascentric (Centrales or Biddulphiales), showing radial symmetry and as pennate (Pennales or Bacillariales), which are bilaterally symmetric (Figure 1.2). Each diatom cell (frustule) consists of two identical silica shells called thecae, forming a petri-dish like connection that contains an upper larger valve (epitheca) and a smaller lower valve (hypotheca). The epitheca is composed of a relatively flat upper part (valve face) with downturned edges (valve mantle), called the epivalve, and one or more hoop like girdle bands called the epicingulum (Figure 1.3)(Last and Smol, 2001). Each thecae is uniquely patterned and these pattern characteristics are mainly used for identification of each organism to species level. They reproduce with both sexual and asexual production. While sexual reproduction, which occurs when the diatom needs restoration for the cell size, results in same sized daughter cells (Mann, 1993), asexual reproduction that takes place with cell division occuring by mitosis, results in two smaller size daughter cells (Mann, 1993).

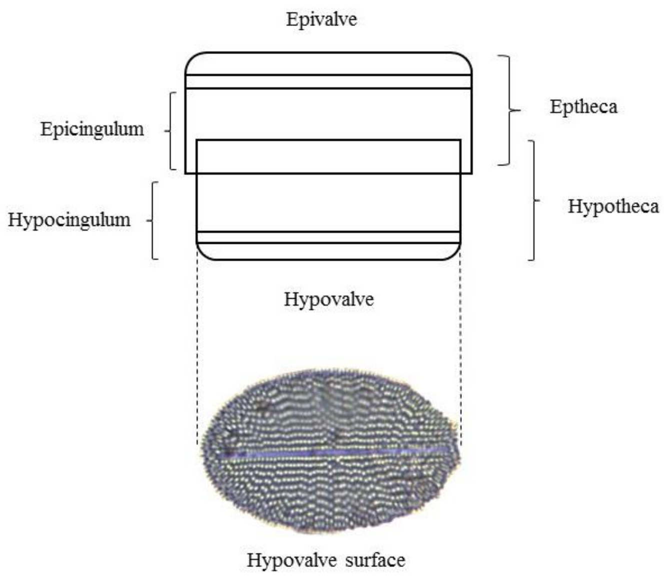

Figure 1.2 The main parts of the diatom frustule (Modified from http://www.ucl.ac.uk/GeolSci/micropal/diatom.html**)**

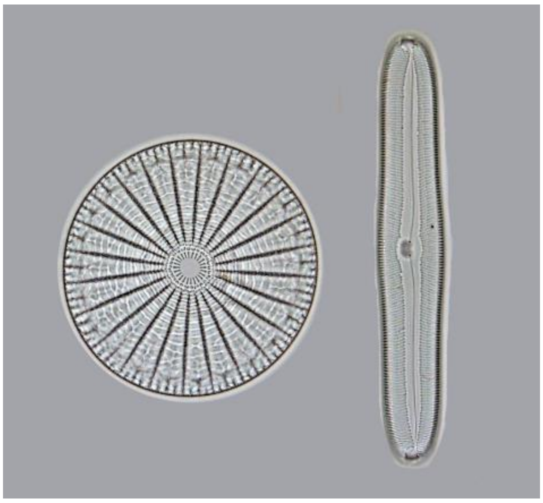

Figure 1.3 Centric diatom (left) and pennate diatom (right). (From http://botit.botany.wisc.edu).

Diatom valves can be found in many environments but they are largely present in fresh and marine waters. There are several factors controlling the presence and abundance of diatoms (Last and Smol, 2001). Increasing temperatures, for instance, have a positive impact on benthic diatom fauna (Hillebrand et al., 2007) . Furthermore, especially in lake ecosystems with higher light availability (since diatoms are light dependent) and mixing events influence the biomass and composition of the species and may even completely alter the fauna (Reynolds, 1984; Smol and Stoermer, 2010). Tilman et al. (1982) found out that diatom species are in competition for phosphorus and silicon uptake so the availability of nutrients and silica directly affect the diatom abundance (Kilham, Theriot, and Fritz, 1996). Since phosphorus concentration has a significant impact on diatom productivity and it triggers eutrophication, diatom-phosphorus relationship is commonly used for detecting the impacts of eutrophication in palaeolimnological studies (Anderson and Rippey, 1994; Bennion, Juggins, and Anderson, 1996; Hall and Smol 1999; Bennion et al., 2011; Dong et al., 2012). pH is another important factor directly influence the presence of diatoms since the silica composition of the cell wall is resistant to acidic conditions which has a positive impact on the preservation of diatom valves (Charles, 1985; Flower, 1986; Battarbee and Charles, 1987). Each diatom species has a different salinity tolerance and they are also commonly used for salinity reconstruction studies (Fritz et al., 1991; Kashima, 2003; Reed, Mesquita-Joanes, and Griffiths, 2012). However, it has alsobeen documented that salinization in freshwater ecosystems enhance the dissolution of diatom cells (Barker, Fontes, and Gasse, 1994; Reed, 1998; Ryves, 1994).

In palaeolimnological research, diatoms are the most-preferred biological proxy due to their wide range of taxonomic distinction, abundance, accurate preservation in the sediment and most importantly their rapid response to the changes in environmental conditions (Lotter, 1998; Smol and Stoermer, 2010b; Reid, 2005). Each diatom species has its own niche preferences and ecological optima (Reavie and Smol, 2001; Fritz, Juggins, and Battarbee, 1993). Accordingly, diatom valves found in sediment have the ability of representing the ecological status of the local habitat (Dixit, Dixit, and Smol, 1992). Diatoms play an important role as being indicators of water quality and environmental change starting from early ages (Nipkow, 1920; Cleve-Euler A., 1922). In twentieth century, they became a major tool for the reconstruction of past water acidification (Flower and Battarbee, 1983; Birks et al., 1990; Ginn, Cumming, and Smol, 2007; Battarbee and Bennion, 2010), eutrophication (Anderson and Rippey, 1994; Bennion, Fluin, and Simpson, 2004; Hall and Smol, 2010; Tremblay, Pienitz, and Legendre, 2014), palaeoenvironment (Marchetto, Colombaroli, and Tinner, 2007; Quillen, Gaiser, and Grimm, 2011; Gomes et al., 2014; Leira, Filippi, and Cantonati, 2015), salinization (Siver, 1999; Reed, Mesquita-Joanes, and Griffiths, 2012; Stenger-Kovács et al., 2014) as well as detecting the impacts of climate change (Fritz et al., 1991; Smol et al., 1995; Laird et al., 1996; Smol and Cumming, 2000)

1.5. Objectives

i. To determine significant environmental variables that structured the diatom community from 32 Turkish Shallow Lakes and by using ordination techniques and information about ecological preferences of diatom taxa, reconstructing the historical changes related with hydology and nutrients for two shallow lakes (Lake Karagöl-İzmir in Western Anatolia and Lake Karagöl-Beypazarı) in Central Anatolia (Chapter 2),

ii. To determine the hydrological changes in the short cores using a multi proxy approach and define the past environmental status of two shallow lakes in Turkey. (Chapter 3),

iii. To investigate the impact of water level changes on lake ecosystem structure using diatoms fossils in three large shallow lakes from Turkey for the last 100 years. (Chapter 4).

CHAPTER 2

INVESTIGATION THE PAST STATUS OF TWO TURKISH SHALLOW LAKES USING A PALEOLIMNOLOGICAL APPROACH

2.1. Introduction

In recent years, eutrophication of freshwater ecosystems with the increase in nutrient levels especially phosphorus (P) and nitrogen (N) became a significant problem worldwide (Hobaek et al., 2012). Eutrophication occurs when the nutrient levels especially phosphorus (P) and nitrogen (N) are increased in freshwater ecosystems (Wetzel, 2001). Eutrophication might be appeared naturally (Hall and Smol, 1993) but for the past 50 years due to uncontrolled discharge of organic matters and nutrients into fragile freshwater ecosystems (Mainstone and Parr, 2002) the main source is anthropogenic(Smol and Stoermer, 2010). Increased nutrient levels had significant impacts on primary producers and this caused the uncontrolled growth of toxic algae in freshwater ecosystems (Martin Søndergaard et al., 2007). As a result, loss of water clarity, biodiversity, and water quality are the main effects of eutrophication process (Burkholder, 2001; Jeppesen et al., 2000; Wassen et al., 2005). Additionally, climate change might exacerbate the occurrence and severity of eutrophication by causing a temperature increase initiating longer growth season, droughts, reduced flushing rates or increased floodings (Smol and Stoermer, 2010; Bennion et al., 2015).

According to Erol and Randhir (2012), Mediterranean region will face 25-30% decrease in precipitation and significant increase in evaporation rates for the next few decades which will lead further salinization of freshwater through drought induced prolonged hydraulic residence (Beklioğlu et al., 2017; Kaushal et al., 2005; Ramakrishna and Viraraghavan, 2005). Consequence of the effects on aquatic organisms would be higher mortality, reducing reproduction and growth rate (Bezirci et al., 2012; Nielsen et al., 2003; Søndergaard et al., 2007). Moreover, there was a positive relationship between the occurrence of toxic algae blooms and the increase in salinity (Sellneret al., 1988) and in some cases, enhanced salinity may cause a shift from clear water to turbid water stage (Davis et al., 2003; James, Fisher, and Moss, 2003). Additionally, a synergistic effect between eutrophication and salinization were detected that saline lakes are prone to eutrophication compared to freshwater lakes containing the same amount of nutrients (Jeppesen et al., 2004).

Turkey, had many shallow lakes with high ecological and economical values that hold a wide variety of species biodiversity due to its topographic and climatic variability (Çakıroğlu et al., 2014; Levi et al., 2014; Şekercioğlu et al., 2011; Boll et al., 2016). Most of the lakes in the area commonly have eutrophic characteristics with frequent salinization owing to uncontrolled land use (e.g. urbanization, agriculture), excessive water withdrawal for agriculture along with prolonged drought (Beklioglu, Altinayar, and Tan, 2006; Bucak et al., 2012; Beklioğlu et al., 2011; Beklioğlu et al., 2017). Despite all these critical problems, there was lack of long term monitoring programme for the Turkish shallow lakes with some exceptions (Meryem Beklioğlu et al., 2017). Consequently, this reduce the detailed information on status of the lakes (Reed et al., 2008) and would prevent to develop management approaches. In such circumstances, palaeolimnological methods can be very valuable and they could provide the missing information and would help enlightening the alteration of past lake environment (Bennion et al., 2011). The sediment of a lake basin serves a great archive for geochemical, physical and biological proxies that can be used to reveal the responses of freshwater ecosystems to environmental to changes and stressors (Keller, 2009; Smol, 2010). Among all proxies, diatom remains in sediment are commonly used in palaeolimnological studies due to their fast responses to ecosystem shifts (Bennion and Simpson, 2011).

Diatoms (Bacillariophyceae) are a common and abundant group of algae in aquatic environments (Werner, 1977). Their contribution to primary production is 20-25% and their role in silica and carbon cycle is crucial (David G Mann, 1999). Since they are so diverse and each taxa had specific preferences for the environment, the abundance of taxa in sediment can reflect past environmental conditions in an effective way (Dixit, Dixit, and Smol, 1992; Gell et al., 2007; Reid and Ogden, 2009). Thus, several studies showed that they have been used to gather information about past eutrophication (Bennion, Juggins, and Anderson, 1996; Bradshaw and Anderson, 2001; Hall and Smol, 2010; Kauppila et al., 2012; Lotter, Birks, Hofmann, and Marchetto, 1998), salinization (Dixit et al., 1992; Gasse, Juggins, and Khelifa, 1995; Reed et al., 2008; Reed, Mesquita-Joanes, and Griffiths, 2012) and climate change (Laird et al., 2003; Marchetto, Colombaroli, and Tinner, 2008).

In this study, surface sediments together with wide environmental variables were taken from 32 shallow Turkish lakes were used to determine the most important variables that control the diatom assemblages. The information on interaction between the diatom community and environmental variables were applied to the short sediment cores taken from two shallow lakes (Lake Karagöl-İzmir in Western Anatolia and Lake Karagöl-Beypazarı, in Central Anatolia to investigate historical changes took place.

The aims of the study are (i) to pin down the environmental changes that influence the diatom distribution in lakes32 Turkish Shallow Lakes (ii) by using ordination techniques and information about ecological preferences of diatom taxa, elucidate the historical changes in hydrology and nutrients in two shallow lakes (Lake Karagöl-İzmir in Western Anatolia and Lake Karagöl-Beypazarı) in Central Anatolia.

2.2. Material and Methods

2.2.1 Field Sampling and Laboratory Analyses

Environmental variables were collected from 32 Turkish shallow lakes during the peak of growing season (August–September) using a snapshot sampling protocol (Moss et al., 2003) (Figure 2.1). Depth-integrated water samples from the deepest point of the lakes were collected and stored frozen for further chemical analyses. Twenty to forty litres of water from the depth-integrated sample was filtered through a 20 μm mesh size filters for zooplankton collection and 4 % Lugol's solution was employed to fix the remains. Total phosphorous (TP) was detected by acid hydrolysis and molybdate blue reaction methods (Mackereth andTalling, 1979). Total nitrogen (TN) was analysed with Scalar Autoanalyzer (San++Automated Wet Chemistry Analyzer, Skalar Analytical, B.V., Breda, The Netherlands). Ethanol extraction method was used for the determination of chlorophyll-a concentration (Jespersen andChristoffersen, 1987) and Golterman et al., (1978 was employed to detect silicate concentrations in samples. YSI 556MPS multi-probe was used to collect in situ data for conductivity (μS), pH, salinity (g/l), temperature (oC) and dissolved oxygen (mg/l) from the same point where Secchi depth and maximum depth were also recorded. Overnight fishing (at least 12 h) with gill nets (12 multiple mesh sizes between 5 to 55 mm) method was used to determine the fish composition and fauna (catch per unit effort, CPUE, number net-1) per lake. Parallel transects with even intervals were set for each lake to collect the aquatic plants samples and percent plant volume inhabited (PVI%) was calculated (Canfield Jr. et al., 1984).

2.2.2 Two Study Sites

Lake Karagöl-İzmir (hereafter Karagöl-İ) (38° 33.48' N, 27° 13.08' E), a natural tectonic lake, is located in İzmir province, near Yamanlar Mountain, Western Anatolia (Edremit, 2011) (Figure 2.1). The lake area is around 2 ha and the elevation is 800 meters (Mermer, Parlak, & Çevik, 1995; Ustaoglu, 1983) and it has a ‘’Natural Park’’ status (Republic of Turkey Ministry of Forestry and Water Affairs). The maximum depth was 6.8 m during the sampling year and the Secchi depth was measured as 1.80 m. The lake is fed by a small stream towards eastern part but the main water resource is the precipitation (Edremit, 2011). The average temperature is 18.5 ºC for the area and the average precipitation is 600 ± 30 mm (Turkish State Meteorological Service, 2011).

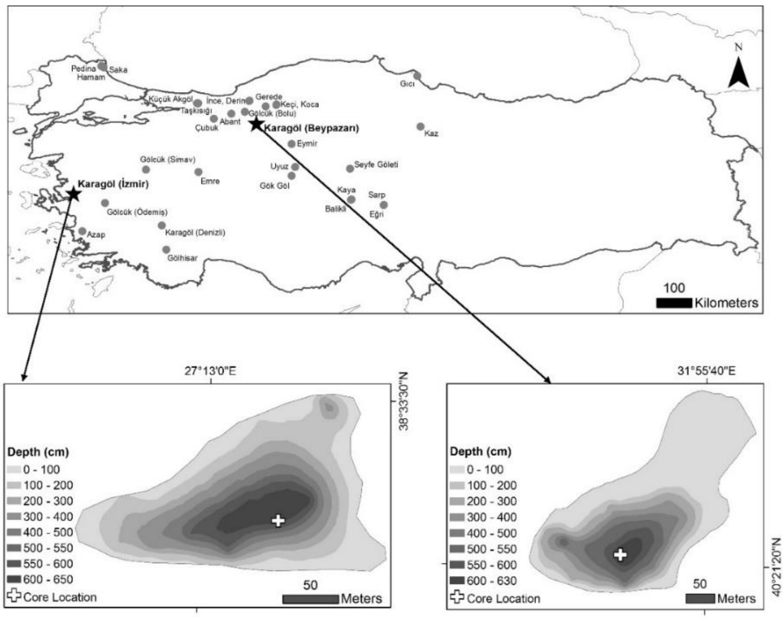

Figure 2.1 Map showing the distribution of 32 shallow lakes (upper part) and the bathymetric maps of Lake Karagöl-İzmir (bottom left) and Lake Karagöl-Beypazarı (bottom right). Coring locations from two lakes were represented with ‘’+’’ sign on each bathymetry map.

Lake Karagöl-Beypazarı (herafter Karagöl-B) is located in Beypazarı, (40°21.35' N, 031°55.57' E), and it has also a ''natural park'' status since 1976 (Figure 2.1). The natural park reserve covers 15 ha while the lake area is 9 ha and the elevation is 1500 m. The lake is surrounded by a forest and utilized for recreational use only. The maximum depth is 6.5 m and the Secchi depth was measured as 1.5 m during the sampling. The average temperature is 13 ºC and the average precipitation is 380 mm (Turkish State Meteorological Service, 2011).

2.2.3 Palaeolimnological Sampling and Analyses

Surface sediment samples (7-10 from each lake) were collected from the deepest point of the lakes with KC-Denmark Kajak Corer, the samples then were mixed and kept frozen in sealed plastic bags until further analyses. A Kajak corer was also used to collect short core samples from the deepest points of Lake Karagöl-İzmir and Lake Karagöl-Beypazarı (Figure 2.1).Each core sample then was sliced into 1 cm intervals on the shore and kept frozen in the sealed plastic bags.Diatom subsamples for taxonomical analyses and counting were prepared for each cm usingBattarbee, (1986). During the application of the method, 10 ml 30% H_2O_2 (hydrogen peroxide) were added onto 0.1 gram of wet sediment and the mixture was put into a water bath (70-80 ºC) until the fizzing activity stopped. After that, the samples were topped with distilled water and centrifuged at 1200 rpm for 4 minutes at least three times. The supernatant part was removed during each centrifuge session and finally, the suspended material was used for the preparation of diatom slides. At least 500 valves were counted from each slide sample and both Leica DMI4000B inverted microscope and Leica DM750 light microscope at Limnology Laboratory, Middle East Technical University were employed for the identification and necessary literature was employed. (Cleve-Euler A., 1922; Hustedt, 1966, 1976, Krammer andLange-Bertalot, 1986, 1988, 1991a, 1991b). Heiriet al., (2001) was used to determine loss on ignition (LOI) at 550 °C and 925 °C, respectively.

2.2.4 Dating

The cores were dated using ^{210}Pb method. ^{210}Pb (half-life is 22.3 year) is a naturally-produced radionuclide, derived from atmospheric fallout (termed unsupported ^{210}Pb). ^{137}Cs (half-life is 30 years) and ^{241}Am are artificially produced radionuclides, introduced to the study area by atmospheric fallout from nuclear weapons testing and nuclear reactor accidents. They have been extensively used in the dating of recent sediments. Samples from Lake Karagöl-Beypazarı core was analyzed for ^{210}Pb, ^{226}Ra, ^{137}Cs and ^{241}Am by direct gamma assay in the Environmental Radiometric Facility at University College London, using ORTEC HPGe GWL series well-type coaxial low background intrinsic germanium detector. ^{210}Pb was determined via its gamma emissions at 46.5keV, and ^{226}Ra by the 295keV and 352keV gamma rays emitted by its daughter isotope ^{214}Pb following 3 weeks storage in sealed containers to allow radioactive equilibration. Cesium-137 and ^{241}Am were measured by their emissions at 662keV and 59.5keV (Appleby et al, 1986). The Lake Karagöl-İzmir core, on the other hand, was analyzed for the activity of ^{210}Pb, ^{226}Ra and ^{137}Cs via gamma spectrometry at the Gamma Dating Centre, Institute of Geography, University of Copenhagen. The measurements were carried out on a Canberra low-background Germanium well-detector. ^{210}Pb was measured via its gamma-peak at 46,5 keV, ^{226}Ra via the granddaughter ^{214}Pb (peaks at 295 and 352 keV) and ^{137}Cs via its peak at 661 keV.

2.2.5 Data Analyses

All environmental parameters were tested for normality using Kolmogorov–Smirnov test in SigmaStat 3.5 (Justel, Peña, and Zamar, 1997) and appropriate transformations were applied (log, logx+1, square root, ln, etc.) if necessary. Species data were also transformed with square root transformation (Legendre and Gallagher, 2001). A series of ordination analyses were applied to the biological proxy and environmental variable data to detect the patterns and causes of variation in multivariate datasets (Ter Braak andPrentice, 1988). Detrended correspondence analysis (DCA) was applied to species data to determine if a linear or a unimodal model was appropriate for the data set of the study. The gradient length was 4.4 so a unimodal approach canonical correspondence analysis (CCA) was chosen for the further analysis (ter Braak, 1995). Moreover, during the analyses, using the variance inflation factors (VIFs), highly correlated environmental variables were detected and the ones had VIF > 5 were removed from the data set (DeSellas et al., 2008). A series of CCA were applied to the subset of environmental variables and species data sets. Next, Monte Carlo permutation tests with 999 unrestricted permutations was applied to the data set to achieve the significant environmental variables impacted the species distribution in the data set. Moreover, CCA was also used to estimate the historical changes in environmental variables. Samples from two short cores were passively plotted onto CCA graph and the positions of the historical samples were used to establish information about the past environmental conditions of two shallow lakes. CCA passive plotting analysis were applied individually for each site and the lake where the reconstruction was applied was removed from the calibration set. CANOCO version 4.5 were applied during the ordination analyses. Finally, stratigraphic diagrams for two study lakes were constructed for diatom short core data using C2 programme (Juggins, 2007).

2.3 Results

2.3.1 Environmental Data

The general characteristics of the 32 study lakes are given in Table 2.1. The size of the lakes varied between 1 and 400 ha. Most of the lakes were shallow except Lake Abant and Derin (Maximum depth: 17.5 m and 9.6 m, respectively). The average oxygen concentration was 6.8 mg L^{-1} while the average pH was 8.1 **(**Table 2.1**).** In general, lakes showed eutrophic characteristics (average TP 134.3 µg L^{-1}, TN 1174.6 µg L^{-1}, Chl-a 33.9 µg L^{-1}, Secchi depth 1.4 m) (Table 2.1) and the mean conductivity was 700.5 µS cm^{-1} for 32 lakes. The average silicate concentration was 5645.5 mg L^{-1} (Table 2.1). The mean CPUE was 320.5 and PVI% was 18.8 %.

Table 2.1 Summary characteristics of environmental variables from the 32 Turkish shallow lakes.

Variables	Unit	Median	Mean	Range
Altitude	**m**	1060.5	858.3	0.0-1423.0
Area	**ha**	12.0	48.4	1.0-400.0
Dissolved Oxygen	**mg L^{-1}**	6.5	6.8	0.6-12.9
pH		7.8	8.1	6.8-9.2
Conductivity	**µS cm^{-1}**	470.5	700.5	101.8-2949.0
Secchi depth	**m**	0.8	1.4	0.2-9.0
Maximum depth	**m**	3.4	3.9	0.6-17.4
Total Phosphorus	**µg L^{-1}**	88.9	134.3	15.0-632.0
Total Nitrogen	**µg L^{-1}**	992.7	1174.6	264.1-2340.0
Silicate	**mg L^{-1}**	5253.7	5645.5	1613.4-15651.0
Chlorophyll-a	**µg L^{-1}**	14.9	33.9	1.8-181.9
Percent Volume Infested	**%**	6.2	18.8	0.0-79.9
Total Fish	**CPUE**	894.2	385.3	0.2-1425.0

2.3.2 Ordination

Species Distribution

A total of 55 species were identified in the surface sediments of 32 shallow lakes. The detailed table of the identified diatom species per lake was given in Appendix A. The data set was mainly consisted of benthic taxa and the planktonic flora were represented with eleven species namely *Aulacoseria granulata* (Grunow), *Cyclostephanos dubius* (Hustedt), *Cyclotella atomus* (Hustedt), *Cyclotella comensis* (Grunow), *Cyclotella meneghiniana* (Kützing), *Cyclotella kuetzingiana* (Fricke), *Cyclotella ocellata* (Pantocsek), *Cyclotella radiosa* (Grunow), *Cyclostephanos tholiformis* (Stoermer), *Stephanodiscus hantzschii* (Grunow)and *Stephanodiscus parvus* (Stoermer & Håkansson) (Appendix A). Plant attach taxa like *Epithemia sorex* (Kützing), *Epithemia turgida (*Ehrenberg) *andEpithemia adnata* (Kützing) were positively associated with PVI%. While planktonic freshwater taxa *C. ocellata* was negatively correlated with conductivity and nutrients, *C. meneghiniana* and *Bacillaria paradoxa* (J.F.Gmelin) were positively correlated with conductivity. Most of the high nutrient tolerant species were positioned along TN-TP gradient namely *Amphora pediculus* (Kützing), *Cocconeis placentula* (Ehrenberg), *Amphora libyca* (Ehrenberg), *Ulnaria ulna* (Nitzsch), *Nitzschia palea* (Kützing) and *Nitzschia amphibia* (Grunow) (Figure 2.1).

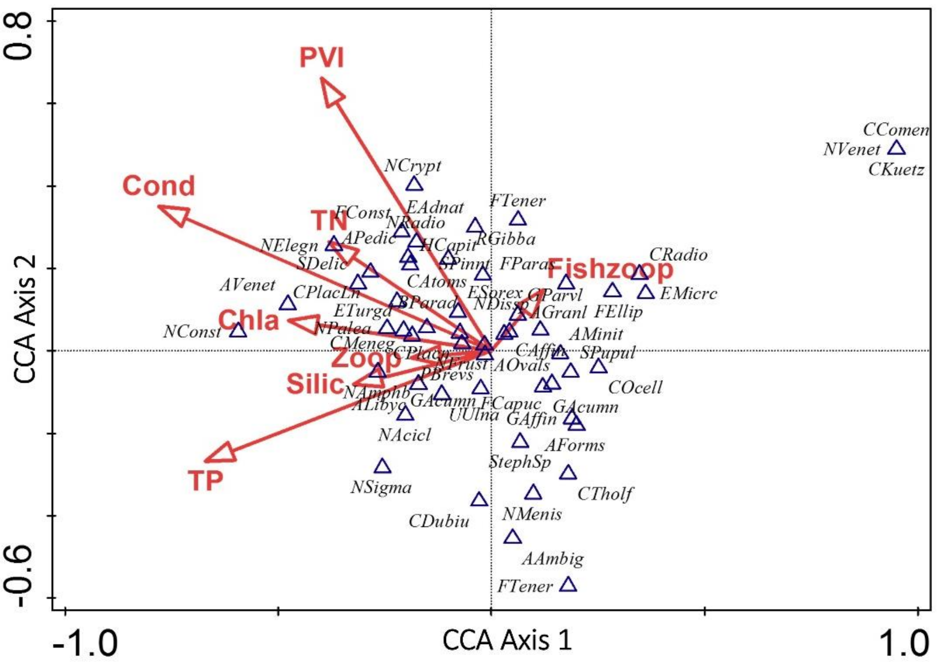

Figure 2.2 Distribution of diatom species in Canonical Correspondence Analysis (CCA) plot with environmental variables namely: PVI (percent plant volume inhabited), Cond (Conductivity), TP (Total Phosphorus), TN (Total Nitrogen), Chla (Chlorophyll-a), Silic (Silicate), Zoop (Zooplankton), Fishzoop (CPUE). For species abbreviations, see Appendix A.

32 sampled lakes were included the ordination analyses. According to CCA analyses results, conductivity, TP, TN, Silicate, Chl-a, Zooplankton, fish abundance and PVI% were explained 30% of the total variation in the distribution of diatoms (Figure 2.3). The eigenvalues of the first and second axis were $k1$ = 0.344 and $k2$ = 0.295. While the first axis was positively correlated with fish, it was negatively correlated with TP, zooplankton biomass and silicate. The second axis was positively correlated with TN, conductivity and Chl-a concentration (Figure 2.2). Additionally, Monte Carlo permutation test with 999 permutations was revealed that conductivity *(p*=0.004*)*, PVI% *(p*=0.12*)* and TP *(p*=0.23*)* are the main

environmental variables that influence the diatom composition and biomass for the lakes in the data set and together they explained 13.1 % of the variation.

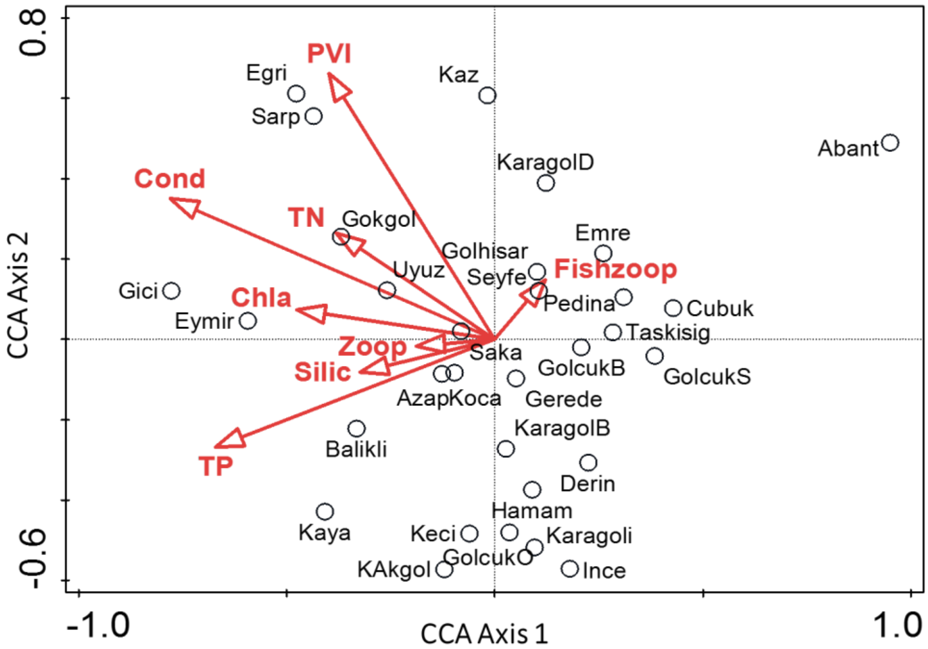

Figure 2.3 Canonical Correspondence Analysis (CCA) plot of the 32 study lakes constrained with environmental variables namely: PVI (percent plant volume inhabited), Cond (Conductivity), TP (Total Phosphorus), TN (Total Nitrogen), Chla (Chlorophyll-a), Silic (Silicate), Zoop (Zooplankton), Fishzoop (CPUE).

Moreover, while lakes with high conductivity (Gıcı, Eymir) were grouped around conductivity gradient, freshwater lakes (Pedina, Çubuk, Taşkısığı, GölcükB, GölcükS, KaragölB, Derin, Hamam, Karagölİ, Gölcük) with were negatively correlated with conductivity gradient (Figure 2.3). Lakes with high PVI % (Kaz, Gökgöl, Eğri, Sarp, Uyuz, Gıcı, Saka) were found around the PVI gradient, lakes with no/low plant coverage (GölcükB, Gerede, Derin, Hamam, Karagölİ, İnce, GölcükÖ, Taşkısığı) were gathered the opposite side of the Figure 2.3. Moreover, lakes with high TP levels (KaragölB, KAkgöl, Kaya, Balıklı, GölcükÖ, Karagölİ, Azap) were positioned closer to TP gradient.

2.3.3 Core Chronology and Loss on Ignition (LOI) Results

2.3.3.1 Lake Karagöl-İzmir

According to 550 ºCresults, organic matter levels showed a linear decrease from top to the bottom parts of Lake Karagöl-İ core (Figure 2.4). Except from 26-27 cm, inorganic carbon levels (LOI 925 ^{o}C) were showing the same pattern as organic carbon throughout the core (Figure 2.4). The unsupported ^{210}Pb content of the surface was about 200 Bq kg^{-1} for Lake Karagöl-İ and the activity decreased exponentially with depth. The calculated flux of unsupported 210Pb is 809 Bq m^{-2} y^{-1}. Theamount was about eight times higher than the estimated local atmospheric supply (Appleby, 2001). Significant activities of ^{137}Cs were found throughout the core and no distinct peaks were present. ^{137}Cs was found in layers dated to well before the initial release of this isotope into nature in the mid-1950 which indicated a mixing event or mobilization of the isotope within the sediment for that period. The average sedimentation rate was 0,31 g cm^{-2} yr $^{-1}$.Overall, the general decline in the activity of unsupported ^{210}Pb with depth was revealed a reasonable chronology for the sample and the bottom part of the core (36 cm) was dated back to 1932.

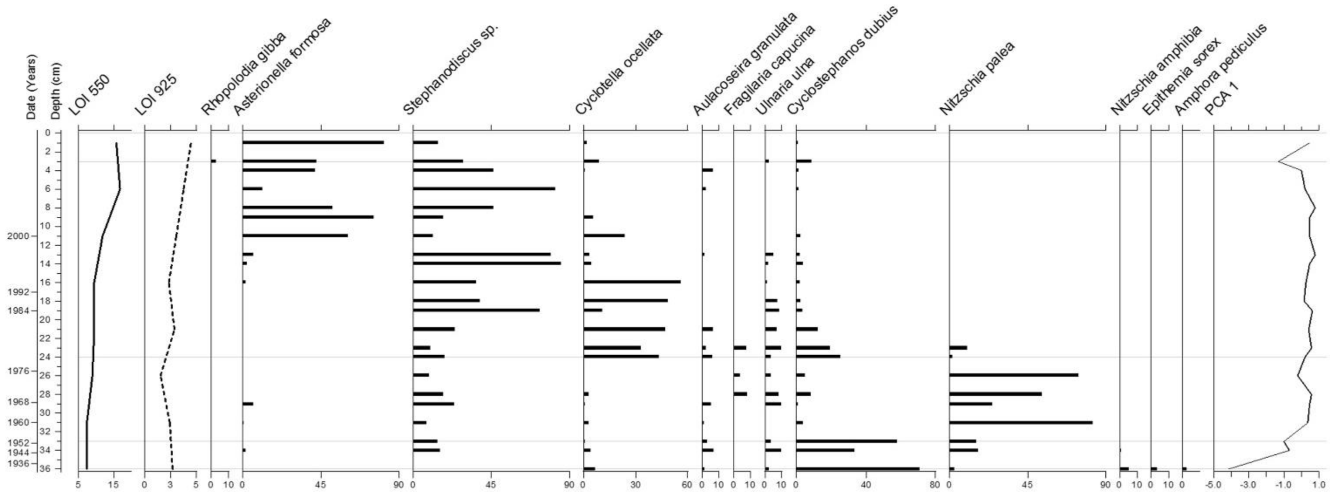

Figure 2.4 Loss on ignition (LOI 550 and LOI 925), percent abundances of diatom taxa and PCA axis 1 scores for Lake Karagöl-İzmir core. Different scales were used for individual diatom species.

Karagöl-İ core was divided into four zones according to diatom assemblage distribution (Figure 2.4). Zone 1 was corresponded between 36 and 33 cm (1930-1950s). *C. dubius*, a centric diatom, was the dominant species of the zone and reached 70 % contribution to the community while the the contribution of another centric diatom *Stephanodiscus* sp. and small pelagic species *N. palea* were around 20 %. Additionally, *C. ocellata*, *A. granulata*, *U. ulna*, *E. sorex* and *N. amphibia* were found in the sample but their contributions were lower (between 5-8 %). Zone 2 was covered between 33-25 cm (1970-1950s) and there was a significant increase in the abundance of *N. palea* (80%) (Figure 2.4). Although a few centric species with low abundance (*A. granulata*, *C. dubius, Stephanodiscus* sp.*, C. ocellata*) were found in the zone, the overall diatom composition was mostly consisted of benthic taxa (*U. ulna, F. capucina, N. amphibia*). Zone 3 covered between 23 to 3 cm (from 1970s to 2010), respectively (Figure 2.4). The occurrence of *Stephanodiscus* sp. was rather significant in that part of the sample and especially between 14 and 6 cm, the abundance was reached to almost 90 %. Another centric species, *C. ocellata* was also showed a significant increase up to 60 % contribution until 16th cm and dropped to almost 5 % contribution at the upper part of the core (2-3 cm). *A. granulata*, *C. dubius, U. ulna*, *F.capucina*, *N. amphibia* and *N. palea* were also found between 2 and 3 cm of Lake Karagöl-İ core. Above 16 cm, colony-forming pennate diatom, *Asterionella formosa* (Hassall) increased significantly (up to 85 %) and together with *Stephanodiscus* sp, dominated the top part of zone 3. Another centric species *A. granulata* and *C. dubius* were also present but their contribution were less than 10 %. Finally, zone 4 was covered the surface part of the core where the high occurrence of *A. formosa* which was found together with *Stephanodiscus* sp (Figure 2.4).

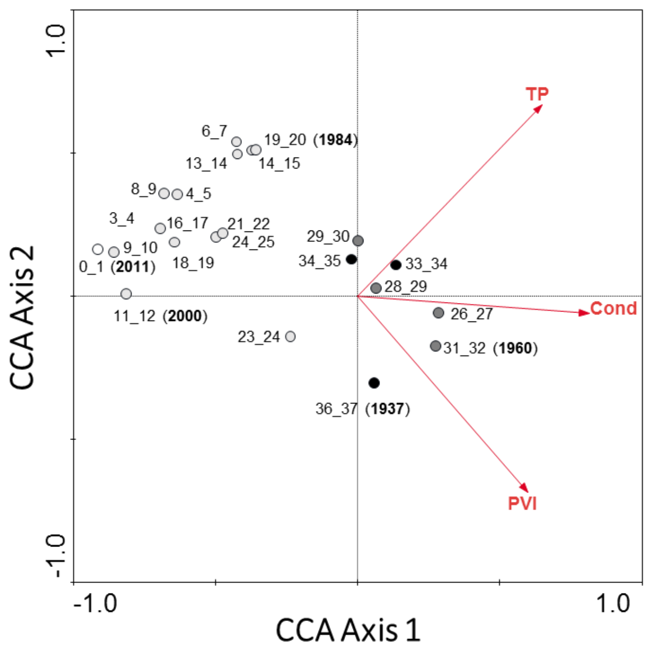

Figure 2.5 CCA plot with sedimentary diatom fossils of Lake Karagöl-İ with main environmental variables as conductivity (Cond), Total phosphorous (TP) and PVI (PVI %). Black dots represent zone 1, grey dots zone 2, light grey dots zone 3 and white dots zone 4.

For Lake Karagöl-İ, passive plotting of the core sample on RDA ordination was revealed the influence of conductivity and TP but was unable to capture the compositional changes in the diatom flora (Figure 2.5). Black dots were represented Zone 4 (1930s) and dark grey dots were covering Zone 3 (1960s) in Figure 2.5 and they were hinting higher salinity and nutrient rich conditions for the sample. While light grey dots was corresponded to Zone 2, covered between 1960s- 2009s and they were grouped opposite to conductivity gradient, white dots were represented the surface part of the core (Figure 2.5).

2.3.3.2 Lake Karagöl-Beypazarı

Loss on ignition 550 ºC (LOI) results were revealed that organic matter levels for Lake Karagöl-B dropped between 36 and 20 cm. Starting from 20 cm, a peak was spotted until 15 cm(Figure 2.6). There was not a significant change between 15 and 4cm and after that point, organic matter levels were increased slightly until the top part of the core. On the other hand, inorganic carbon levels at 925 ºC did not show a significant change throughout the core and remained between 4,5 and 3,8 % all along the sample(Figure 2.6). The unsupported ^{210}Pb activities decline irregularly with depth for Lake Karagöl-B, especially changes around 12 cm in unsupported ^{210}Pb activities was a sign of increased sedimentation rates. According to ^{137}Cs activities, the core was showed two distinct peaks at 22,5 cm and 16 – 19 cm was indicating that the core was affected both by atmospheric testing of nuclear weapons around 1963 and Chernobyl accident around 1986 and fluxes of unsupported ^{210}Pb deposited into the sediments of the coring location were fairly constant, The average sedimentation rate was 0,089417 g cm^{-2} yr^{-1},and the 30th cm of the sample dated back to 1902.

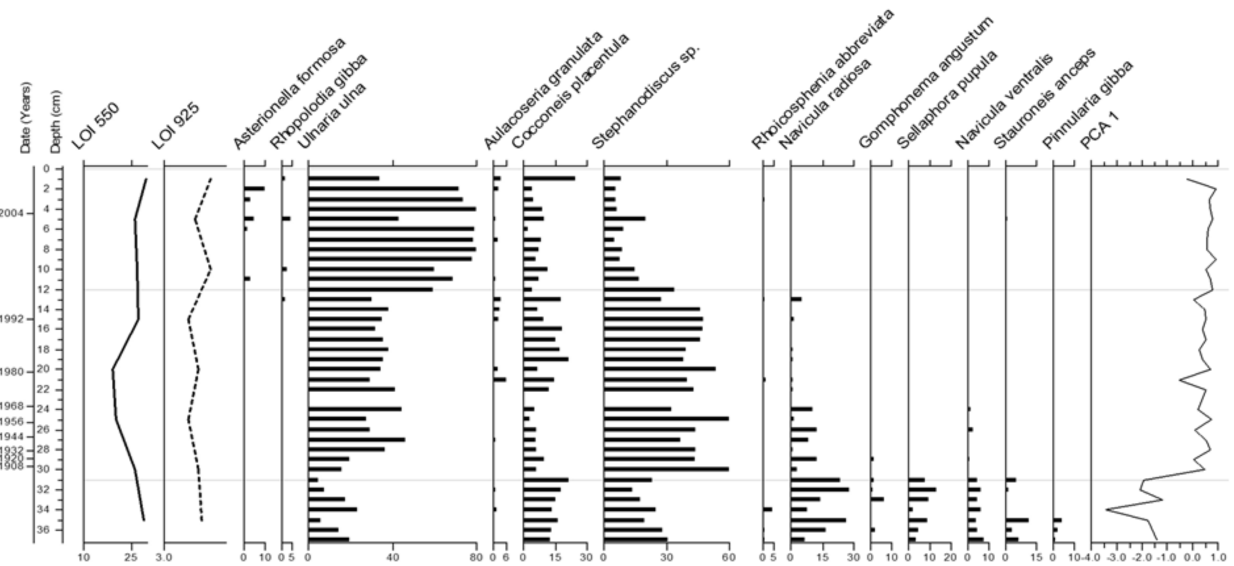

Figure 2.6 Loss on ignition (LOI 550 and LOI 925), percent abundances of diatom taxa and PCA axis 1 scores for Lake Karagöl-Beypazarı core. Different scales were used for individual diatom

Compared to Karagöl-İ core, Karagöl-Beypazarı sample was mainly dominated with two taxa overall (*Stephanodiscus sp, U. ulna*) and divided into three different zones (Figure 2.6). The first zone was between 37 and 31 cm. That part of the core was mainly dominated with benthic taxa (*Gomphonema angustum* (A.Cleve), *Sellaphora pupula* (Kützing), *Navicula ventralis* (Krasske), *Stauroneis anceps* (Ehrenberg), *Pinnularia gibba* (Ehrenberg)) and plant associated species *S. ulna* and *C. placentula*. A nutrient tolerant species *Stephanodiscus* sp.was also present in the zone with 30 % abundance. Zone 2 was started around 1900s and ended in the 1990s (30-12 cm) where *Stephanodiscus* sp.was reached the maximum contribution (58 %) and there was an increase in the contribution of *U. ulna* as well up to 40 % (Figure 2.6). The occurrence of *C. placentula* continued and increased to almost 20 % contribution between 18 and 13 cm. Another taxa found in this zone of the core were *Navicula radiosa* (Kützing), *Achnantidium minitissiumum* (Kützing), *Fragilaria capucina* (Desmazières), *Staurosirella pinnata* (Ehrenberg) and *A.granulata*. The last zone was covered the area between 11 cm and surface (Figure 2.6). In this zone, the dominant species was *U. ulna* with 80 % contribution. The contribution made by *Stephanodiscus* sp. dropped to 10 % and the same taxa were present (*C. placentula*, *A. minitissiumum*, *F. capucina*, *S. pinnata* and *A. granulata*) in the zone 3 as well compared to Zone 2. However, at the surface part, *C. placentula* increased up to 28 % and dominated the cm with *U. ulna*.

The influence of TP was revealed while plotting the short core data on RDA ordination results of Lake Karagöl-B (Figure 2.7). The distribution of the core samples for those two zones were mostly along with TP gradient. On the other hand the samples were distributed along TP and conductivity gradient in Zone 3 (Figure 2.7). (PVI %). Black, grey and white dots represent zone 1, zone 2 and zone 3, respectively.

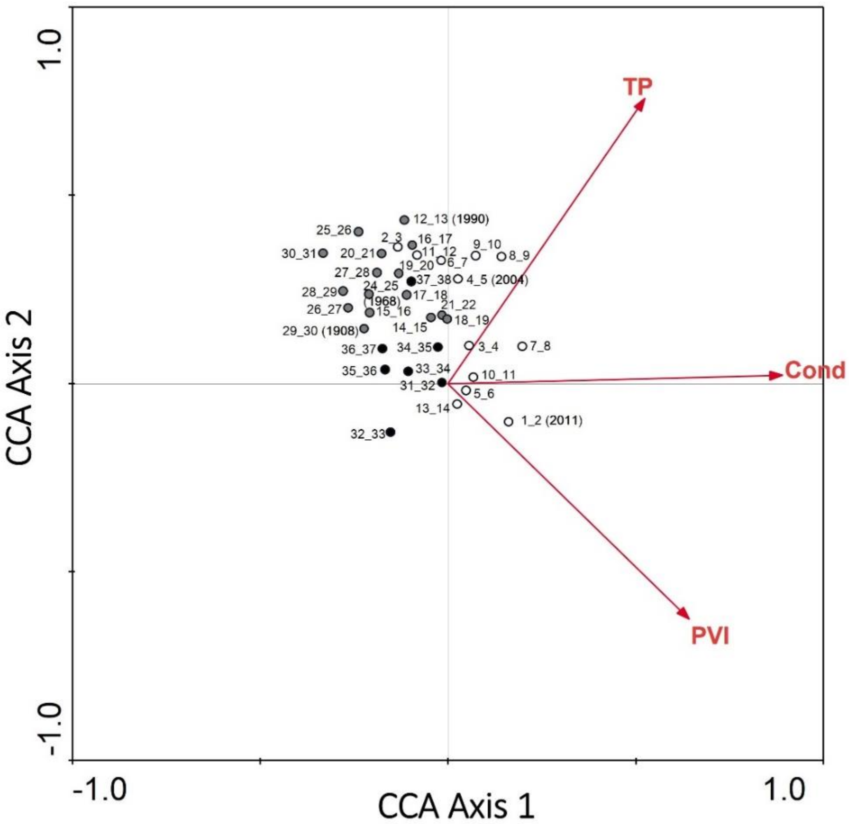

Figure 2.7 CCA plot with sedimentary diatom fossils of Lake Karagöl-B with main environmental variables as conductivity (Cond), Total phosphorous (TP) and PVI (PVI %). Black dots represent zone 1, grey dots zone 2 and white dots zone 3.

2.4. Discussion

There are several factors influence the diatom distribution in sediment (Last andSmol, 2001). In the present study; conductivity, PVI% and TP which mainly impacted the diatom assemblages of study lakes and among three environmental parameters, conductivity was found to be the most significant variable. This finding also support the results of Çakıroğlu et al., (2016) and Levi et al., (2014) that revealed the significant impact of salinity on structuring the fossil cladoceran assemblages and plant remains for Turkish Shallow lakes. Additionally, Reed et al., (2012) used a multi-proxy approach with a training set included 49 lakes from

Turkey and found out the significant impact of conductivity for shaping the species composition. Besides, the passive plotting approach allowed the qualitative projection of past changes with multiple environmental factors using the diatom fossil data. In this study, the combination of ordination analyses and ecological preferences of diatom species were used to interpret the variables that shape the diatom community structure and past status of two shallow lakes from Turkey.

The core data reflected the changes in the diatom flora for both lakes. Lake Karagöl-İ core clearly showed the shift between benthic and pelagic taxa. While the bottom part (1930s) of the core was dominated with C. *dubius*, a typical mesotrophic pelagic species, the composition of the sample was changed to dominance of benthic *N. palea,* an epiphyteknown for its tolerance and resistance to high nutrient levels (Soininen, 2002) starting from 1960s until 1970s. The shift between the benthic and pelagic flora of diatoms in lake ecosystems addressed water level changes in several studies (Leira, Filippi, and Cantonati, 2015; Wasylikowa et al., 2006). For example, a study spotted several water level change periods using pelagic-benthic diatom fauna shift in Lake Estanya (Spain) during Late glacial and Holocene (Morellón et al., 2009). Moreover, the increase in LOI at the same period was reflecting shallower conditions. The benthic diatom community of shallow lakes are directly linked with water clarity and light availability (Liboriussen and Jeppesen, 2003). Correspondingly, the compositional change of the diatom may represent water level changes or increased light availability for Lake Karagöl-İ. The passive plotting of the lake between 1930s and 1970s (Zone 1 and 2 in Figure 2.7) was associated with TP and PVI% since C. *dubius* hadhigh nutrient tolerance and *N. palea* is an epiphyte of enrich waters (Marks andPower, 2001) which might reflect eutrophic conditions with the presence of macrophytes and low turbidity which is in line with (Bradshaw et al., 2000) showed that lower lake levels might create an environment with low turbidity and higher nutrient availability

The following period covered between 1970s and 2011 and dominated planktonic taxa as *C. ocellata*, *Stephanodiscus* sp. and *A. formosa*. The pelagic dominated system in this period might point out a habitat change related with water levels (Rakowska, 2004) and might hint changes in the salinity concentrations as well.

The strong presence of *Stephanodiscus* sp. and *C. ocellata* were detected between 24 and 16 cm (1970s-1990s) for Lake Karagölİ core. *C. ocellata* is a centric diatom that favours oligotrophic and freshwater conditions (Reed et al., 2012; Round, 1973; Winder and Hunter, 2008). On the other hand, several studies from Turkish shallow lakes reported that *C. ocellata* were found in mesotrophic and eutrophic freshwater lakes (Altuner and Aykulu, 1987; Baykal, 2006; Erkaya et al., 2011; Ertan and Morkoyunlu, 1997; Şen, Yıldız, and Akbulut, 1994). Additionally, *Stephanodiscus* sp. is known for its high tolerance to nutrient increase and is a common diatom of eutrophic lakes (Anderson, 1990; Battarbee, 1986; Bennion and Simpson, 2011). The dominance of these two species suggests nutrient rich conditions for Lake Karagöl-İ between 1970s and 1990s. The passive plot of Lake Karagöl-İ, on the other hand, reflected the influence of conductivity on the distribution of diatom species rather than nutrients for the same period. The pelagic species *C. ocellata* was negatively correlated with the conductivity gradient concurs with the other studies that the species was found in freshwater lakes (Erkaya et al., 2011; Winder and Hunter, 2008). Consequently, the presence of *C. ocellata* and *Stephanodiscus* sp.suggests a freshwater environment with high levels of nutrients for Lake-Karagöl-İ. Starting from 14 cm (1996) to the surface part of the core (2011), another compositional shift was occurred in the core as *C. ocellata* was replaced with *A. formosa*. *Stephanodiscus* sp. and *A. formosa* are freshwater diatoms (Berthon, Alric, Rimet, and Perga, 2014; Hundey et al., 2014) favoured mesotrophic and eutrophic conditions (Bennion and Simpson, 2011; Reynolds, 1984; Reynolds et al., 2002) and the dominant presence of the taxa suggested eutrophic conditions for Lake Karagöl-İ concurs with the previous findings for the same lake (Sömek and Balık, 2009).

Lake Karagöl-B core was mainly dominated with benthic species which included the dominant taxa of *N. radiosa, C. placentula, U. ulna* and pelagic *Stephanodiscus* sp. during the early 1900s. The presence of *N. radiosa* and *S. pupula* might represent increased salinity and nutrient levels (Varris and Vinobaba, 2012). Additionally, *C. placentula* and *U. ulna* are categorised as epiphytic and nutrient tolerant species (Çiçek and Yamuç, 2017; Kawamura and Hirano, 1992; Lange et al., 2011; Pip and Robinson, 1984; Zaim, 2007) the presence of *U. ulna* indicated eutrophication for lake ecosystems (Kavya and Ulavi, 2014) that support our finding showing positively correlation with TP. The presence of *Stephanodiscus* sp. was another sign of nutrient richness for early 1900s, since the taxa is commonly found in eutrophic waters (Bennion et al., 2011).

The rest of the core revealed a compositional change rather than a habitat change since the abundance of two high nutrient favoured taxa, *U. ulna* and *Stephanodiscus* sp.were inversely correlated with each other in the sample. While the dominant species was *Stephanodiscus* sp. between 1900s to early 1990s (between 36 and 12 cm), *U. ulna* became abundant between 11 cm and surface where the presence of *Stephanodiscus* sp. dropped significantly. In the meantime, the continuous presence of plant attached taxa *C. placentula* together with epiphytic *U. ulna* is a hint at presence of submerged plant coverage for the lake and less turbidity. The passive plot of Lake Karagöl-B also reflected the presence of plants since the top part of the core was distributed along PVI % and TP gradient. Moreover, Bucak et al. (2012) and Özkan et al. (2010) also revealed the presence of macrophytes during less clear water conditions when there was a water drop in Turkish shallow lakes. Additionally, the high proportion of *U. ulna* at the upper parts of the core might address pollution, especially after the 2000s. The overall distribution of Lake Karagöl-B taxa were suggested nutrient rich and less turbid conditions with epiphyte and pelagic related diatom taxa. The passive plot of the lake was reflected a nutrient rich environment as well. On the other hand, the reflection of diatom-PVI % relationship was rather weak. The dominant epiphyte of the core, *U. ulna* was more influenced with nutrients in the ordination plot so it is suggested that the impact of TP was overridden the PVI % gradient for the passive plot.

The Mediterranean climate had drastic impacts on shallow lake ecosystems in Turkey as sudden water level drop and salinization due to high evaporation rates during long and warm summer periods and relatively shorter precipitation periods for the region (Beklioglu et al., 2006; Iyigun et al., 2013). Moreover, the future climate for the Mediterranean region would include higher temperatures (3–4 °C increase) with higher evaporation rates and at least 10% decrease in the precipitation rates (Önol and Unal, 2014) which will enhance the salinization process of the lakes in the area as well as will cause higher nutrient availability due to higher hydraulic residence time (Beklioğlu et al., 2017; Coppens et al., 2016). Thus, salinization might exacerbate the impact of eutrophication on lake ecosystems as well (Talling, 2001). Recently, lake ecosystems in Turkey are facing with pollution and high demand of water usage for agriculture and population growth (Bucak et al., 2017; Levi et al., 2016).

Under the pressure of several threats, Turkey holds there are more than 900 lakes and ponds with great biodiversity and endemic species and the economic value of the ecosystem is crucial (Reed et al., 2008; Yagbasan and Yazicigil, 2012) and especially for shallow lakes, the lack of having long term monitoring data makes it hard to predict the past, current and future status of these freshwater bodies. The present study revealed that the lakes have been in a eutrophic state during the past ca. 100 years respectively, despite being located in two different climatic zones. The results further suggested salinization around the 1970s as well. The short core diatom distribution and the passive plot were more correlated for Lake Karagöl-İ while the passive plot could not fully reflected the relationship between diatoms and the presence of plants. Although, the passive plot approach has advantages (more flexible, multi-variable interpretation) compared to transfer function, due to the movement in ordination space, the interpretation of the results are rather difficult (Çakıroğlu et al., 2016). Overall these findings highlighted the continuity of the problems that shallow lakes encounter and suggested the importance of taking the necessary precautions in order to prevent these long-lasting problems in the future.

2.5. Conclusion

Palaeolimnological methods and analyses were revealed that conductivity and nutrients were mainly shaped the community structure of diatoms in Turkish shallow lakes. Despite being in two different regions of Turkey, both lakes had eutrophic characteristics from the past to present and also hints of past salinization. To develop better strategist against several threats, extensive knowledge is critical.In this context palaeolimnological methods would be an important tool particularly if there is a lack of long term monitoring data. Moreover, diatom fossils were found successful proxy about tracing the changes in past environments.

CHAPTER 3

THE IMPLEMENTATION OF A MULTI-PROXY APPROACH FOR TWO SHALLOW LAKES IN TURKEY

3.1. Introduction

The importance of acquiring long-term data for predicting the current and future status of the ecosystems became crucial for environmental studies (Gillson and Marchant, 2014; Saulnier-Talbot, 2016). However, without the information about the past history of any ecosystem, making implications for management and conservation issues might be misleading (Saulnier-Talbot, 2016). In addition to their essential role in creating a habitat for a range of organisms, and ensuring water and energy supply for humans, freshwater environments create a bond between the catchment and the atmosphere which allows them to collect potential information about the whole ecosystem and its changes (Levi et al., 2016). In this sense, shallow lakes become more valuable since they constitute the majority of freshwater resources worldwide (Moss, 1998; Wetzel, 2001) and they are more fragile to overcome the impacts of human-induced stressors and climate change compared to deep lakes (Dokulil, 2014).

Palaeolimnology is a discipline, which uses biological, physical and geochemical remains deposited in sediment for gathering information about the past environmental status of lake ecosystems (Frey, 1988; Smol et al., 2005) . Sediment accumulated in lake basin offers a great archive about the past and present status of lakes and the surrounded ecosystem, as long as the preserved material have both allochthonous and autochthonous remains (Keller, 2009; Smol, 2010) . While the physical and chemical context of the sediment offers information about the changes in both lake and the catchment characteristics, biological remains (e.g. diatom, sub-fossil Cladocera, macrophytes) reveal the changes in lake trophic structure and biota (Harrison and Digerfeldt, 1993; Çakıroğlu et al., 2014; Bennion, Simpson, and Goldsmith, 2015; Levi et al., 2016). Moreover, palaeolimnological proxies offer information about the impacts of anthropogenic stressors (Bennion et al., 2011) and climate change (Laird et al., 2003; Rühland, Paterson, and Smol, 2008; Moss, 2010) on lake ecosystems in a wider point of view.

Diatoms are unicellular organisms and widely distributed in aquatic ecosystems, especially in rivers and lakes (Last and Smol, 2001). Since each diatom species have its own optima via environmental preferences and they are highly sensitive to environmental changes (Last and Smol, 2001), diatom remains are one of the most widely used biological proxies for palaeolimnological studies (Dixit, Dixit, and Smol, 1992; Ryves et al., 2001; Ginn, Cumming, and Smol 2007; Battarbee et al., 2012; Bennion, Fluin, and Simpson, 2004). Due to their critical position in lake trophy as controlling the algal biomass with their herbivorous diet and providing food for zooplanktivorous fish (Moss, 1998; Jeppesen et al., 2011), the status of Cladocerans in lake ecosystems implicate crucial information about changes in trophic structure in the community and water quality (Abrantes et al., 2006; Jeppesen et al., 2011) and cladoceran sub-fossils are frequently used in palaeolimnological reconstructions (Jeppesen et al., 1996; Amsinck, Jeppesen, and Ryves, 2003; Davidson et al., 2011; Çakıroğlu et al., 2016). The abundance of the macrophytes in shallow freshwaters have a positive influence on the whole ecosystem structure, as well as carbon and nutrients cycles individually (Scheffer et al., 1993). Moreover, changes in their composition and quality refers to structural changes in the water column, since their distribution and presence is under the impact of nutrients, water clarity and water chemistry (Van der Valk, 1987; Gafny andGasith, 1999; Thomaz et al., 2007). Therefore, both plant macrofossils and cladoceran subfossiles provide valuable information related to changes in water clarity in palaeolimnological studies (Hannon and Gaillard, 1997; Davidson et al., 2005; Birks 2007; Koff and Vandel, 2008; Levi et al., 2016).

Each biological proxy preserved in the sediment has its own advantages and disadvantages for explaining and reflecting the changes in their environment as mentioned above (Mann, 2002). Moreover, due to the complex structure of lake ecosystems, integrating several different proxies while making palaeoenvironmental interpretations might offer a wider and deeper understanding of the past status of lakes and the interactions between the organisms and the environmental variables that shapes the lake fauna and flora (Smol, 2002; Saulnier-Talbot, 2016). In that sense, the application of multi-proxy approach is a growing trend for palaeolimnological studies. For example, Hynynen et al. (2004) used diatom and chrinomid fossils to track the industrial pollution while Taylor et al. (2006) investigated the impacts eutrophication on Cladocera, diatoms and pollen distribution for Irish Lakes. For Turkish lakes, there are several examples for the application as Roberts et al. (2001) used isotopes, diatoms and pollens for assessing the changes related with climate on a deep lake, Eski Acıgöl, while Reed et al. (2012) investigated the traces of salinization using diatom and ostrocoda. However, studies investigated the past status of shallow lakes of Turkey using a multi-proxy approach are still limited.

Despite their critical importance, due to the lack of long-term monitoring and historical data, our understanding of past status of these ecosystems are unfortunately restricted. Moreover, due to the negative impacts of climate change and unregulated land use, eutrophication, salinization and water scarcity are becoming extensive challenges for Turkish lakes (Beklioglu, Altinayar, and Tan, 2006; Beklioglu and Tan, 2008; Özen et al., 2010; Bucak et al., 2012; Beklioğlu et al., 2017). In recent years, palaeolimnological studies started to fill this gap (Sari and Külköylüoğlu, 2010; Reed, Mesquita-Joanes, and Griffiths, 2012; Levi et al., 2014; Çakıroğlu et al., 2014,2016; Levi et al., 2016), yet more studies are needed to build a contemporary data set for lake management and sustainability, and making reliable predictions about the future status of Turkish shallow lake ecosystems.

The aim of the present study was to infer the past status of the selected two lakes, using three main biological proxies; diatoms, sub-fossil cladocerans and plant macrofossils. Moreover, the applicability of the multi-proxy approach was also tested for gathering reliable information. It is hypothesised that (i) biological remains extracted from the cores would provide valuable information about the past status of lakes over the last 60 years (ii) the changes in the short cores would support the idea that eutrophication and salinization are the main environmental stressors that shapes the fauna of biological proxies in Turkish Shallow Lakes.

3.2 Material and Methods

3.2.1 Study Sites

Two shallow lakes were sampled for multi-proxy analyses from Turkey (Figure 3.1). General characteristics of the lakes were given in Table 3.1. Lake Gıcı is located around the North-eastern coastal boarder of the Kızılırmak Delta (Figure 3.1). The lake itself is a part of a lagoon system in the area and connected to Lake Tatlı nearby. Due to high precipitation differences between wet and dry seasons along the Black Sea region, the lake level varies throughout the year (Maraşlıoğlu, Soylu andGönülol, 2011). The maximum depth of the lake was 1.5 m with 0.25 cm Secchi depth and the lake area was 146 ha. The surface temperature of the lake was around 25 C° and the conductivity level was as high as 1370 mS cm^{-1}. Total phosphorous and total nitrogen was 329 µg L-1 and 1445 µg L-1, respectively (Table 3.1). Lastly, the lake had a high plant coverage of 80 %.

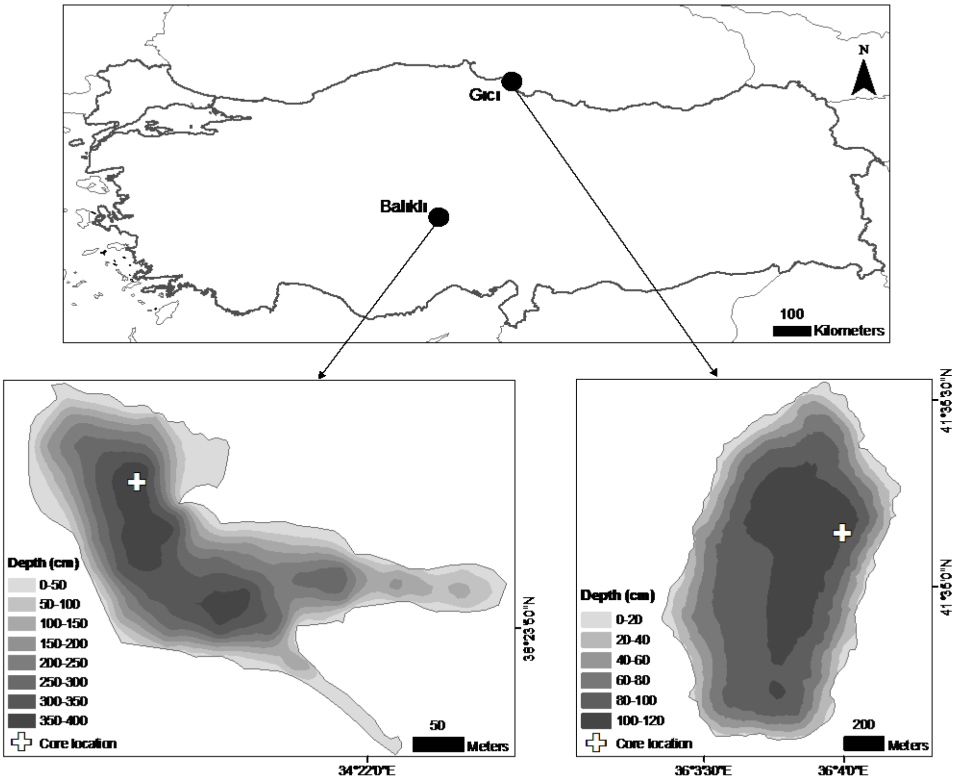

Figure 3.1 Map showing the locations of two study lakes and the bathymetric maps of Lake Balıklı (bottom left) and Lake Gıcı (bottom right). Coring locations from two lakes were represented with ‘’+’’ sign on each bathymetry map.

The second study site Lake Balıklı, lies on the border of Aksaray province, is 1173 m above sea level and categorized as eutrophic (Altinsacli, Altinsacli and Pacal, 2014) (Figure 3.1) . The main water resources of the water body are rain water, ground water and springs around the area (Altinsacli, Altinsacli, and Pacal, 2014). The area of the lake was 6 ha and the maximum depth was 4.3 m. The Secchi depth of the lake was 0.63 m with higher nutrient levels (TP: 402 and TN: 2340, respectively) compared to Lake Gıcı (Table 3.1). The conductivity of the lake was 643 mS cm^{-1} and the plant coverage was 5.4 %.

Table 3.1 General characteristics of Lake Gıcı and Lake Balıklı.

Variables	Unit	Lake Gıcı	Lake Balıklı
Altitude	**m**	0.0	1173.0
Area	**ha**	146.0	6.0
Dissolved Oxygen	**mg L^{-1}**	5.1	11.3
pH		9.2	7.0
Conductivity	**µS cm^{-1}**	1369.5	643.4
Secchi depth	**m**	0.3	0.6
Maximum depth	**m**	1.4	4.3
Total Phosphorus	**µg L^{-1}**	328.8	401.9
Total Nitrogen	**µg L^{-1}**	1445.4	2340.0
Silicate	**mg L^{-1}**	7540.0	11060.0
Alkalinity	**meq L-1**	3.2	5.2
Chlorophyll-a	**µg L^{-1}**	62.1	90.8
Percent Volume Infested	**%**	79.9	5.4
Total Fish	**CPUE**	195.0	85.5

3.2.2 Palaeolimnological Methods

During paleolimnological sampling, Kajak corer was employed to collect sediment samples from the lakes. Kajak corer is categorized as a messenger-operated type gravity corer. The working principle of the equipment is simple as a messenger-operated valve that allows the sampler to keep open until it touches to solid ground. Once it touches the sediment surface, the top part is closed by the messenger and a vacuum effect is created till the sampler reaches to the surface, collecting the sediment from the bottom (Hakala, 1971). Using this method, two short cores were collected from Lake Gıcı and Lake Balıklı. After the collection, the samples were divided into 1 cm segments and were kept in zipping top clear plastic bags until further analysis.

Short core samples were analyzed at UCL Environmental Radiometric Facility-UK for the determination of ^{210}Pb, ^{226}Ra and ^{137}Cs activities that are necessary for dating. ^{210}Pb Dating is a method that is commonly used to determine the date of the sediment samples in anthropogenic time scale (Jeter, 2000; Krishnaswamy, Lal, Martin, and Meybeck, 1971; Robbins, Edgington, and Kemp, 1978). Noble gas ^{222}Rn in the sediment transfers to the atmosphere and decays to ^{210}Pb (Jaworowski et al., 1972). Followingly, decayed ^{210}Pb (unsupported ^{210}Pb) is attached to aerosols and deposited in the sediment again. Rather than atmospheric ^{210}Pb, since sediments contain U and ^{226}Ra, ^{210}Pb can naturally be found in sediments which are referred to as ‘’Supported ^{210}Pb’’ (Krishnaswamy et al., 1971). Another technique for dating sediment is detecting ^{137}Cs particles. During the 1950s and 1960s, high amount of ^{137}Cs was released into the atmosphere and deposited in sediment due to nuclear weapon testing. Using a low background germanium gamma detector, those ^{137}Cs particles can be counted from sediment and they address the part of the sample that represents the years that nuclear testing had been conducted. The accumulation rates of the sediment samples were also calculated during that period. Additionally, loss of ignition (LOI) content of the samples were analyzed at METU, Limnology Laboratory. The LOI method is simply used for the determination of the organic material and the carbonate amount in the sediment (Dean, 1974; Santisteban et al., 2004).

3.2.2.1 Biological Proxies

Three main groups of biological proxies (diatoms, subfossil cladocerans and plant macrofossils) were analysed from the sediment. The analysis of subfossil cladocerans were applied by Ayşe İdil Çakıroğlu and the analyses of plant macrofossils were held by Eti Ester Levi at Middle East Technical University, Limnology Laboratory during their PhD studies.

Diatom

Battarbee (1986) was followed for the preparation of diatom samples for taxonomical analysis. In this method, approximately 0.01 grams of dried sediment (or 0.1 gram of wet sediment) were placed in centrifuge tubes, and 10 ml 30% H_2O_2 (hydrogen peroxide) was added. Samples then were placed in a water bath with a temperature between 70-80 ^oC. The H_2O_2 level of the samples and the water level of the water bath were checked regularly and the sample tubes were removed from the water bath when the fizzing activity stopped. Tubes then were filled with distilled water to the top and centrifuged for 4 minutes at 1200 rpm and the supernatant was removed. The process was repeated at least four times and 1-2 drops of weak ammonia (NH_3) solution was added to each sample to prevent grouping of diatoms before slide preparation. Henceforth, a few drops of the suspension material were put on the coverslip for overnight and once the coverslips were completely dried, they were fixed onto glass slides using diatom mountant (Naphrax) using a hotplate. Leica DMI4000B inverted microscope with 1000x magnificationand Leica DM750 light microscope were used for the identification and counting the diatom valves. Approximately 500 valves were counted from each sample and Krammer and Lange-Bertalot (1986), Parker (1997), Krammer and Lange-Bertalot (1988), Krammer and Lange-Bertalot (1991b), Krammer and Lange-Bertalot (1991), Hustedt (1976), Hustedt (1927-1966) were used for identification.

Sub-fossil Cladoceran

Methods by Korhola and Rautio (2001) and Jeppesen et al. (1996) were applied for sub-fossil cladoceran sample preparation. 100 mL of 10 % KOH were added onto 5 grams of wet sediment and the sample was boiled for about an hour in order to remove organic matter from the setting. A Stereo microscope (LEICA MZ 16) and an inverted light microscope (LEICA DMI 4000) were used for larger remain (<45 µm) counting and Frey (1959), Flössner (2000) and Szeroczyńska and Sarmaja-Korjonen (2007) were used during the identification process. Furthermore, specific parts from the cladoceran remains (Carapaces, head shields, post abdomens, post abdominal claws and resting eggs) were separated from the sample and added to the data set (Çakıroğlu et al., 2014).

Plant macrofossils

The preparation of plant macrofossils was conducted according to Brodersen *et al.*, (2001) and Odgaard & Rasmussen (2001). Approximately 10–300 cm^3 of sediment was washed from three different mesh sized sieves and a stereo-micro-scope (OLYMPUS SZX12) together with a stereo-microscope (LEICA MZ 16) was accompanied for counting and identification. Beijerinck (1947), Nilsson (1961), Berggren (1969), H. Birks (1980) Berggren (1981), Haas (1994), Birks (2007) and Mauquoy and Geel (2007) and reference material from Aarhus University Herbarium was used for identification issues.

In the final stage, C2 programme was applied to the biological proxy data for constructing a distribution stratigraphic diagrams for two study lakes (Juggins, 2007) .

3.3 Results

3.3.1 Core Chronology and Biological Proxies

3.3.1.1 Lake Gıcı

Loss on ignition (LOI) results revealed that the organic carbon levels at 550 ^{o}Cfor Lake Gıcı dropped until 25th cm followed by an increase afterward until the bottom (Figure 3.2). On the other hand, inorganic carbon levels at 925 ^{o}C decreased until 11th cm and remained constant below 12th cm until the bottom part of the core (Figure 3.2). It was found out that the activity of unsupported ^{210}Pb was decreased irregularly from top to the bottom of the core, which addressed the changes in sedimentation rate. Moreover, it was thought that the ^{137}Cs activity peak around 27 cm was refered to 1963 when the nuclear weapon testing was conducted (Table 3.2). Accordingly, the CRS model was applied to the results and modified in reference to^{137}Cs activity results. It had been detected that the bottom part of the core (27,5) represented the year 1963 (Table 3.3). The rest of the dating was adjusted from the model values. Additionally, the average sedimentation rate was calculated as 0,035 g cm^{-2} yr^{-1} .

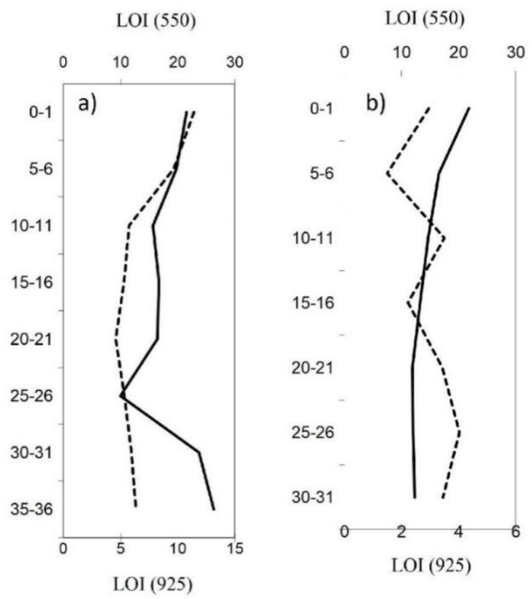

Figure 3.2 Loss on ignition (LOI) results of Lake Gıcı (a) and Lake Balıklı (b). Black lines represents LOI 550 and dashed lines represent LOI 925 values in figure.

Table 3.2 Chronology of ^{137}Cs activity along Lake Gıcı core.

Depth	**Cs-137**	
cm	Bq Kg^{-1}	±
0.5	20.33	5.46
4.5	10.69	3.01
10.5	11.36	2.42
14.5	14.16	1.7
20.5	23.33	1.66
24.5	28.54	2.02
27.5	35.38	2.39
30.5	24.69	2.93
32.5	12.89	1.93
36.5	3.61	1.61

Table 3.3 ^{210}Pb chronology and sedimentation rate of Lake Gıcı Core.

Depth	Drymass	Chronology			Sedimentation Rate		
cm	g cm^{-2}	Date (AD)	Age (yr)	±	g cm^{-2} yr^{-1}	cm yr^{-1}	± %
0	0	2011	0				
0.5	0.0906	2011	0	2	0.388	1.046	40.1
4.5	1.6668	2007	4	2	0.452	0.948	34.2
10.5	4.8602	1997	14	3	0.234	0.391	22.1
14.5	7.6525	1987	24	4	0.356	0.498	28.2
20.5	12.0024	1977	34	5	0.473	0.653	48.3
24.5	14.8965	1970	41	6	0.355	0.555	54.9
27.5	16.4797	1963	48	7	0.169	0.304	44.5

In total, 24 diatom species were identified for the whole sample. A serious dissolution problem detected for 20 cm and only several broken pieces of diatom valves were spotted during the investigation of the subsample. The overall core was dominated with benthic taxa (Figure 3.3). A planktonic species *C. meneghiniana* was only found at the top and bottom parts of the core with the presence lower than 6%. The bottom part of the core (35 cm) was dominated by plant attached species like *E. adnata*, *U. ulna* and an epipelic species *Anomoeoeis sphaerophoria* that has a high salinity tolerance. *C. placentula*, *E. sorex, A. granulata* and *P. brevistriata* was found at that part of the sample as well but in low abundances. *A. sphaerophoria* was the dominant species at 30 cm. The abundance of *E. adnata* was dropped to almost half (20 %) and an increase was spotted for the abundances of *U. ulna* and *C. placentula* as well for the same point. Followingly, *C. placentula* and *U. ulna* was dominated the core around 25 cm (the 1970s). Other than those, *E. adnata*, *P. brevistriata* and *Nitzschia palea* were also found in the sample. *C. placentula* was the abundant species again around 15 cm (the late1980s) along with *E. sorex. U. ulna* and *E. adnata* (Figure 3.3). *E. sorex,C. placentula* and *U. ulna* *dominated* the upper part (10 cm) of the core where the abundance of *N. palea* was under 10 %. *C. placentula* and *E. sorex* were the dominant species at 5 cm and small benthic, eutrophication tolerant species, *Amphora pediculus* was also found in the sample. The surface part (2010) was abundant with *U. ulna* along with the presence of other small benthic taxa as *A. pediculus*, *A. ovata* and *A. veneta*. (Figure 3.3).

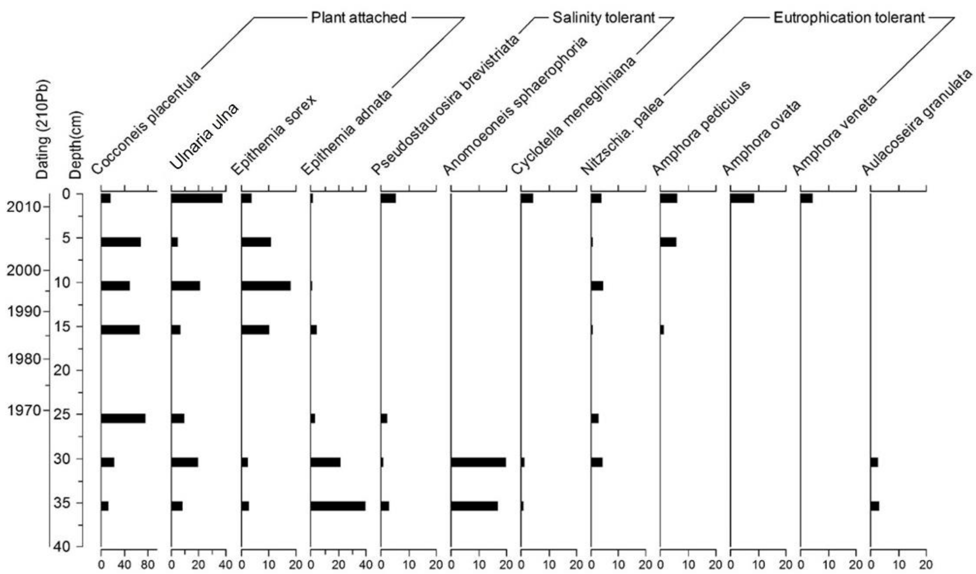

Figure 3.3 Diatom stratigraphy for Lake Gıcı core. Different scales were used for the species data as in % relative abundances.

Lake Gıcı core was mainly dominated with macrophyte and sediment- associated cladoceran taxa as *Chydorus sphaericus,A. rectangula/gutata* and a sedimentary species *Leydigia leydigi.* While the pelagic species *Daphnia* spp., *B.longirostris* and *Moina* spp started to increase from 16 cm (the 1980s) until the surface part of the core, benthic taxa *L. leydigi* and *L. acanthocercoides* showed a significant decrease for the same period. The pelagic fauna was dominated the core around 15 cm whereas the presence of plant and benthic associated species was quite low at that point (Figure 3.4).*O. tenuicaudis*, a plant associated species, only found at first the

16 cm of the core and the presence of the species was absent below that cm (Figure 4.3).

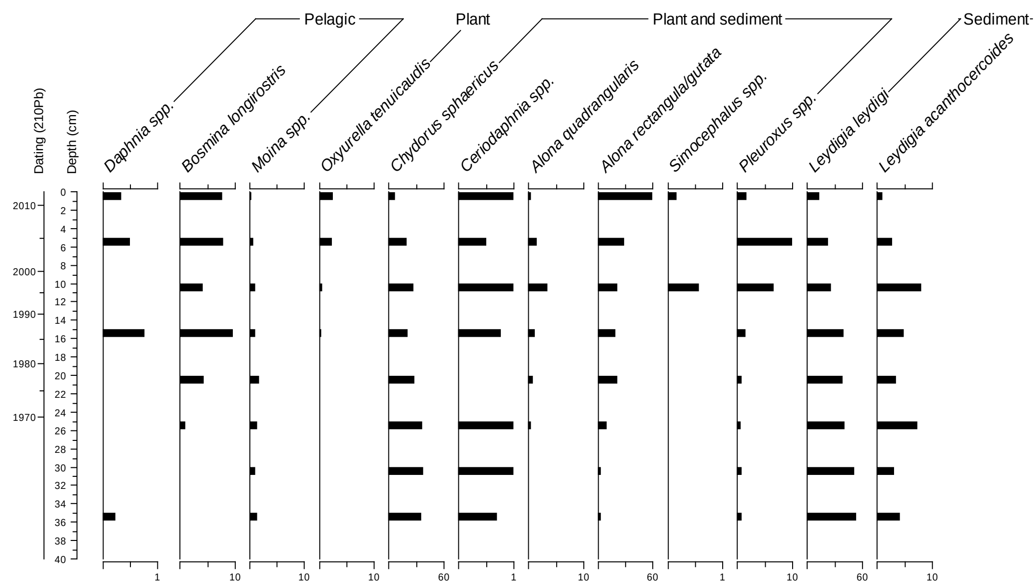

Figure 3.4 Cladoceran stratigraphy for Lake Gıcı core. Different scales were used for the species data as in % relative abundances.

The core sample was divided into three main sections for plant macrofossil analyses. The first zone was categorized between 34 and 40 cm. Zone 1 was dominated by submerged macrophytes namely *Potamogeton* sp., *N. marina* and *Ceratophyllum sp.* Besides, the floating-leaved flora was represented by *Phragmites australis* and the taxa showed a significant decrease until 34 cm. While the abundance of *Potamogeton* sp., *N. marina*, *P. australis* and *Ceratophyllum* sp. were decreased between 25 and 34 cm (the laate 1970s), the amount of *R. sect. Batrachium* (submerged) and *Typha* sp. (floating-leaved) increased throughout the Zone 2 (Figure 3.5). The abundance of *N. marina* increased between 18 cm (Around 1990s) and 14 cm. whereas, the presence of *R. sect. Batrachium* was increased for the same part of the sample (Figure 3.5). Zone 3 covered between 1990 cm and 2011 (surface part) where the abundance of *Ranunculus* remained unchanged. On the other hand, the presence of *Ceratophyllum* sp. and *Typha* spwas increased until the top part of the sample (Figure 3.5). Remains of *Chara* spp. showed an increasing trend until 5 cm but the abundance of the species was dropped until the surface part of the core.

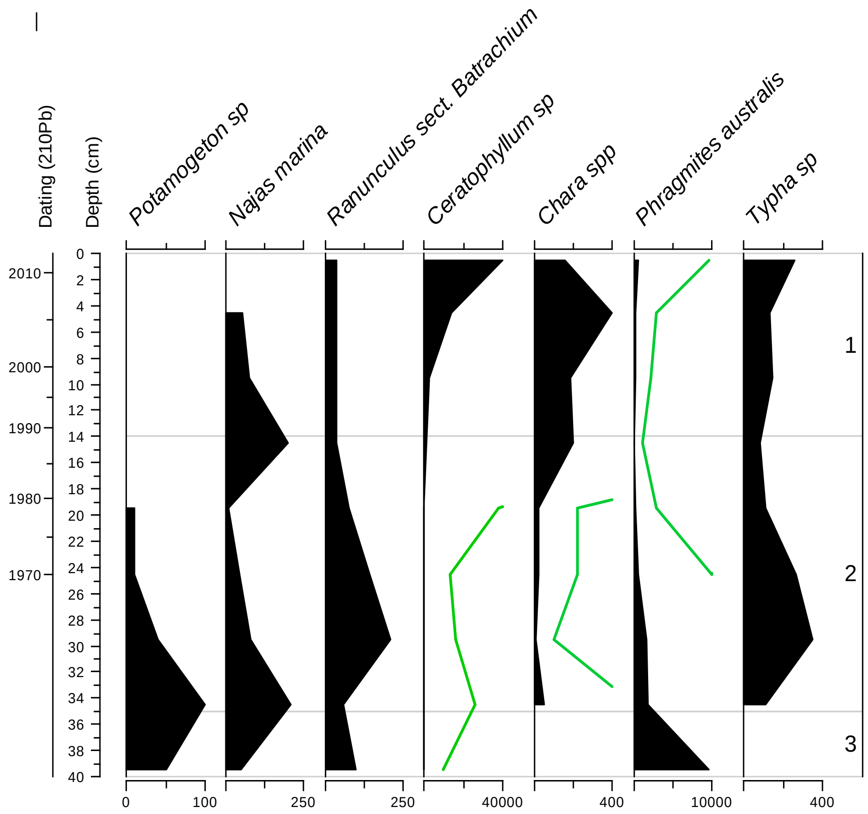

Figure 3.5 Plant macrofossil stratigraphy diagram for Lake Gıcı core.

3.3.1.2 Lake Balıklı

According to LOI results, organic carbon levels at 550 ºCshowed a linear decrease from top to the bottom parts of Lake Balıklı core.At the same time, inorganic carbon levels at ºC 925 increased throughout the core, expect for two decline peaks around 5-6 cm and 15-16 cm.During the dating of sediments samples from Lake Balıklı, it was found out that the supported ^{210}Pb was mostly unbalanced and there was only a minor decrease in unsupported ^{210}Pb activity along the core. A ^{137}Cs peak was detected around 16 cm but compared to Lake Gıcı sample, the overall ^{137}Cs activity levels were rather low. There was a small peak of ^{241}Am found around 2 cm and 12 cm but due to lack of ^{137}Cs activity at those levels, the results were not used for the dating model. According to the CRS model, the core only represented 10 years of time and the sedimentation rate tended to be as high as 0.65 g cm^{-2} year-^{1}‘(Table 3.4).

Table3.4 ^{210}Pb chronology and sedimentation rate of Lake Balıklı Core.

Depth	Drymass	Chronology			Sedimentation Rate		
cm	g cm^{-2}	Date (AD)	Age (yr)	±	g cm^{-2} yr^{-1}	cm yr^{-1}	± %
0	0	2011	0				
0.5	0.0981	2011	0	2	0.8901	3.64	67.8
4.5	1.1005	2010	1	2	0.7044	2.773	57.7
8.5	2.1301	2008	3	2	0.4992	1.887	49.2
12.5	3.2175	2006	5	3	0.741	2.81	60.9
16.5	4.2395	2005	6	3	0.8007	2.914	63.2
20.5	5.4155	2003	8	4	0.6105	2.137	58.1
22.5	5.9537	2002	9	4	0.6807	2.438	58.9
24.5	6.5323	2001	10	5	0.3489	1.209	53

The bottom part (2001) of the Lake Balıklı core was mainly dominated by small benthic diatom taxa as *N. palea* and *N. amphibia*. At that point, low nutrient favored taxa *A. minitissimum*and *A. pediculus* were also presented together with salinity tolerant taxa *Staurosirella pinnata* (Figure 3.6). The amount of *U. ulna* and *S. hantzschii* was rather low as 5 % at the bottom. Between 20 and 6 cm, *U. ulna*, a common epiphyte, was abundant throughout the sample and the abundance of *S. hantzschii* was increased as well while the number of benthic taxa decreased (*N. palea*, *N.amphibia*. *A. minitissimum* and *A.pediculus*). The top part of the sample was represented 2011 and dominated with *U. ulna*, *S. hantzschii* and epipelic *Nitzschia sigmoidea* (Figure 3.6).

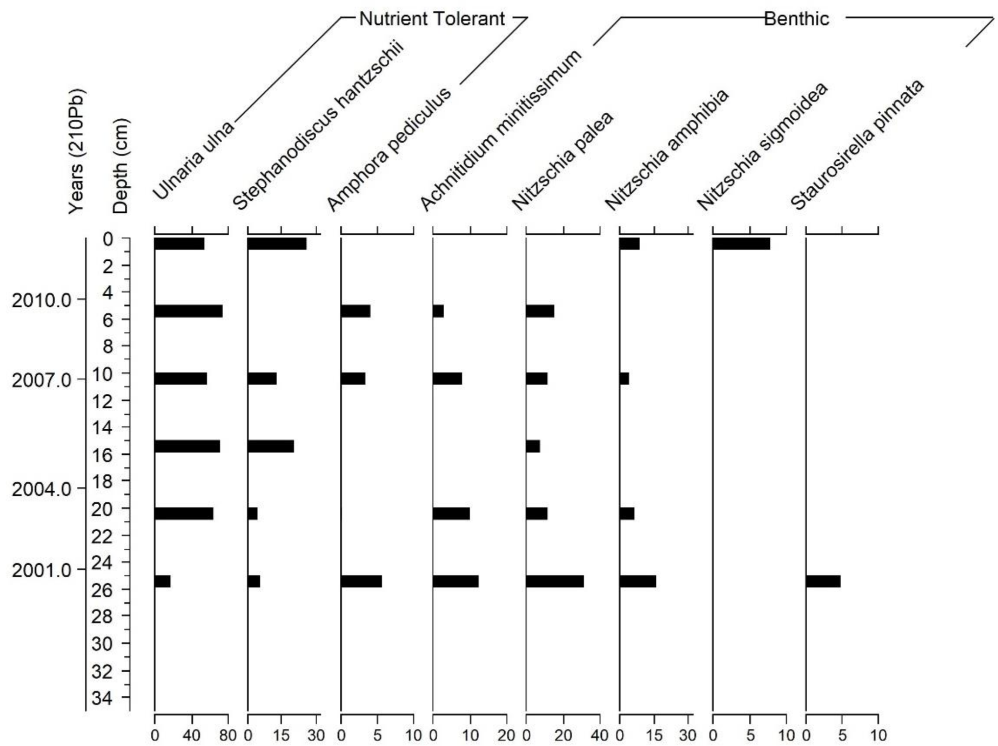

Figure 3.6 Diatom stratigraphy for Lake Balıklı. Different scales were used for the species data as in % relative abundances.

The cladoceran community of Lake Balıklı core was dominated by *C. sphaericus, A. rectangula/gutata* and *B. longirostris,* which had higher salinity tolerance compared to other taxa, along with the core (Figure 3.7). While big bodied cladoceran *Daphnia* spp. was only found at 25 cm (2001), *L. acanthocercoides* was spotted around 20 cm. Plant and sediment associated taxa were distributed all along the core. Among those, *Pleuroxus sp*, *A. excise* and *Ceriodaphnia* spp. were represented with low abundances (below 5 %). The abundance of *L. leydigi*, a benthic species, were increased up to 5% around 15 cm but decreased to 2 % at the top part of the sample (Figure 3.7).

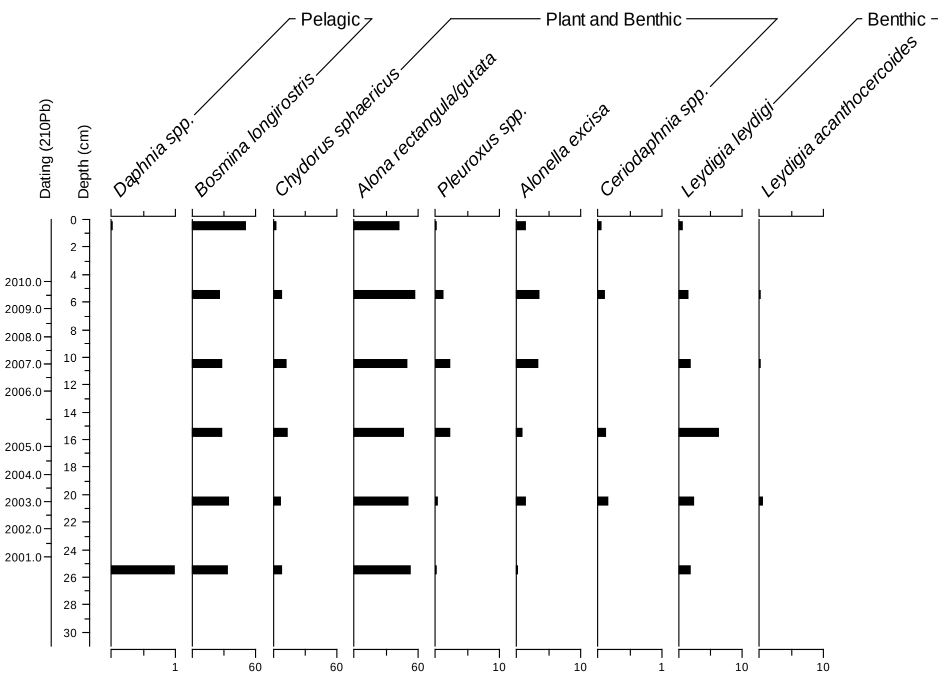

Figure 3.7 Cladoceran stratigraphy for Lake Balıklı. Different scales were used for the species data as in % relative abundances.

Three different zones were identified for plant macrofossil samples in Lake Balıklı core. *Ceratophyllum* sp. was the dominant species in Zone 1 (Between 30 cm and 24 cm). Zone 2 was engaged the years between 2002 and 2005 (Figure 3.8) and included mainly *Ceratophyllum* sp. and *Myriophyllum sp.*. A small amount of floating-leaved species *Typha* sp. was found in the sample as well for the same point. Zone 3 started around 14 cm. The abundance of *Typha* sp increased until 7 cm and decreased again until the top part of the core. While the abundance of *Ceratophyllum* sp. showed a linear decrease up to surface, the amount of *Myriophyllum sp.* was constant until 4 cm (2010) and increased between 4 cm and surface (Figure 3.8).

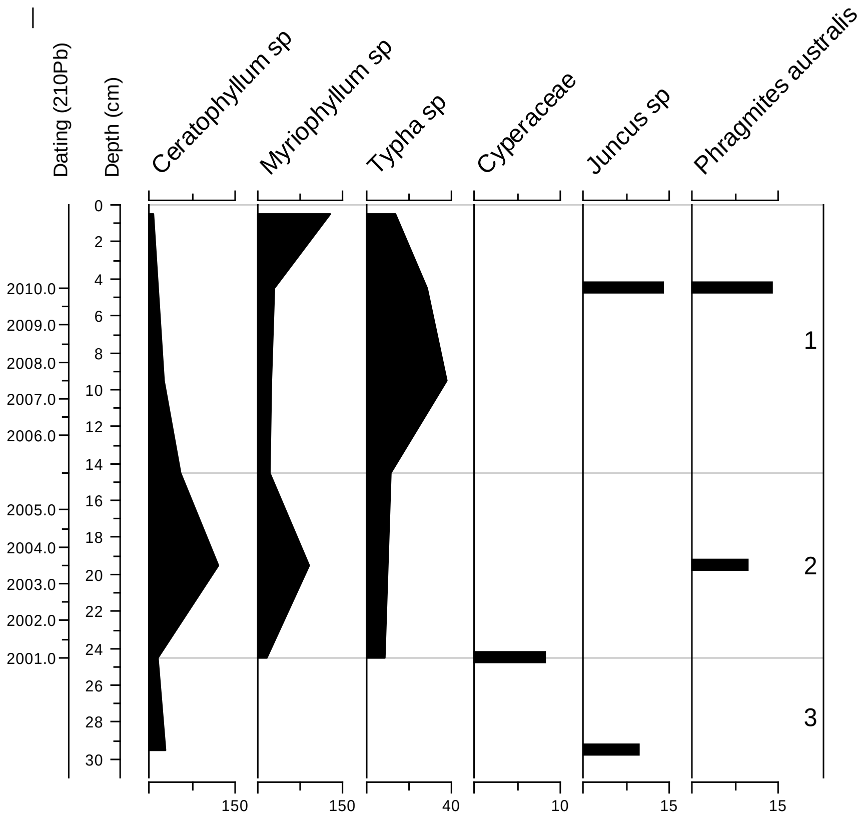

Figure 3.8 Plant macrofossil stratigraphy diagram for Lake Balıklı.

3.4 Discussion

Short core samples from two shallow lakes, Lake Gıcı and Lake Balıklı, were examined using a multi-proxy palaeolimnological approach for the study. Both lakes showed eutrophic characteristics and changes in salinity were spotted due to compositional shifts of the proxies used in the study.

The results of biological proxies were showed that Lake Gıcı was mostly dominated with benthic taxa at the bottom part of the core and mainly consisted of obligate epiphytes as *E. adnata*, *E. sorex*, *C. placentula, U. ulna* (Hobbs et al., 2014; Pip andRobinson, 1984; Zaim, 2007) and *A. sphaerophora* which is highly found in waters with high conductivity and it is a cosmopolitan species of inland saline lakes(Snoeijs and Potapova, 1995). Accordingly, the presence of *Epithemia sp.* in lake ecosystems is associated with shallow water conditions (El Hamouti, 2010). Moreover, high abundances of *E. adnata* might refer shallow fresh to oligosaline environment (El Hamouti and Gibert, 2012). In addition, sub-fossil cladoceran taxa also highlighted clear water conditions with higher salinity since the abundant species *C. sphaericus* and *L. leydigi* are reported to be tolerant to high saline conditions (Çakıroğlu et al., 2016). The high abundance of floating leaved plant fossils (*P. australis*) also suggested shallow conditions for the lake (Almquist, Dieffenbacher-Krall, Flanagan-Brown, andSanger, 2001; Hannon andGaillard, 1997) and the presence of salinity tolerant submerged plants *Potamogeton* sp. and *N. marina* was another sign for saline conditions before 1960s. Higher organic carbon content at the bottom part of the Lake Gıcı core also might point out water level changes, especially low water conditions. It is also suggested that prevaling drought conditions might have created shortage in water budget that might have tirggered salinity

Starting from the 1970s, a compositional shift from salinity tolerant diatom species *A. sphaerophora* and other dominant taxa *E. adnata* to *C. placentula* (80%) and *U. ulna* (10%) was appeared at Lake Gıcı core. *C. placentula* is an early colonizer and as a common epiphyte to lake ecosystems, it spreads worldwide (Kawamura & Hirano, 1992). On the other hand, several studies revealed that *C. placentula* had a high nutrient tolerance and the species often found in mesotrophic to eutrophic waters (Bennion et al., 2011; Marks andPower, 2001). Similarly, Tokatli, (2013) reported the dominance of the same species in highly eutrophic Porsuk Dam Lake in Turkey. For the same period, *U. ulna* was the other species spotted in the core and used as an indicator of anthropogenic pollution (Kavya and Savitha Ulavi 2014). The dominance of *C. placentula* continued until the top part of the core that hinted a nutrient increase between the 1970s until 2000s. In addition, epiphyte *E sorex* was also found in the core but in low abundances (10 %-20%). Concordantly, Bennion et al. (2011) found out that high abundance of diatom C. placentula together with a low abundance of *E.sorex* might reveal high nitrogen concentrations for the lakes. Moreover, the increase in cladoceran species *A. rectangula/gutata* and *B. longirostris* also supported the nutrient richness idea since Davidson et al.,(2007) reported the high abundance of *A. rectangula/gutata* in a eutrophic lake from Denmark and the presence of *B. longirostris* was a sign of high nutrient levels for European lakes (Nevalainen andLuoto, 2012; Nevalainen et al., 2013). Furthermore, the increased presence of *Ceratophyllum* sp., a high nutrient-tolerant macrophyte species (Beklioglu, Ince and Tuzun, 2003) and mesotrophic *Chara* sp. was also another hint for the enhanced nutrient load for Lake Gıcı. Additionally, high amount of organic carbon (Kauppila and Valpola, 2003) and the presence of short submerged plants supported shallow water phase in the time of eutrophication.

Around the 2000s, a continuous increase in organic carbon levels and changes in the macrophyte fauna from short submerged forms to tall submerged form macrophytes were hinted again the nutrient-rich environment for Lake Gıcı. Furthermore, it is thought that decreasing amount of saline tolerant macrophyte fauna during the same period might reveal the overridden impact of eutrophication over salinization (Li et al., 2011). The appearance of pelagic cladoceran species *Daphnia* spp. and *B. longirostris* might also be related with water level increase for the environment but the over all cladoceran fauna was still dominated with benthic fauna. The appearance of non-planktonic diatom flora *N. palea, A. pediculus, A. ovata* and *A. veneta* at the top part of the core together with epipthytic *C. placentula* and *U.ulna* was still pointing out nutrient-rich waters (Bennion et al., 2011). Moreover, the nutrient tolerant cladoceran taxa (*A.rectangula/gutata*) and macrophyte taxa (*Ceratophyllum* sp,) both reached the highest abundance for the same period supporting the eutrophic conditions for Lake Gıcı.

The core sample from Lake Balıklı, however, represented only the recent 10 years of the lake between 2001 and 2011 and the stability of organic carbon levels for the sample was a sign of no significant habitat change for the environment (Kauppila and Valpola, 2003). The dominant diatom species of the bottom part was *N. palea.* Kalyoncu et al.(2008) found this species in polluted environments and according to Tokatlı (2013), *N. palea* was dominant in highly eutrophic Porsuk Dam Lake. Together with *N. amphibia,* these two diatom species were reported pollution indicators for Nilüfer Stream Basin (Karacaoğlu and Dalkıran, 2017). Moreover, the presence of *U. ulna* and *S. hantzschii* for the same point also reflected nutrient rich conditions (Bennion et al., 2011; Kavya, Savitha and Ulavi, 2014). The cladoceran fauna also addressed eutrophic conditions since the dominant taxa *A. rectangula/gutata* and *B. longirostris* was reported several times as elements of polluted waters in Europe (Davidson et al; 2007; Nevalainen et al., 2013). There was not a significant change for the subfossil cladoceran taxa of the core. Two high nutrient tolerant species *A. rectangula/gutata* and *B. longirostris* were abundant thorough the sample. Other plant and benthic associated species (*C. sphaericus, Pleuroxus spp., Ceriodaphnia* sp.*)* were also found in the core together with benthic taxa (*L. acanthocercoides*) but in very low abundances (>5 %). Moreover, the presence of high nutrient tolerant plant macrofossil taxa (*Ceratophyllum* sp. *and Myriophyllum* sp.) was also found for the bottom part of the Lake Balıklı core. *Myriophyllum* sp. was the dominant species of macrophyte flora. Fan and Li (2005) reported that *Myriophyllum* sp. grows in eutrophic conditions and Hussner et al. (2009) pointed out the high abundance of the species in nutrient-rich sediment environment. A taxonomical shift was spotted at the diatom fauna replacing with *N. palea, U. ulna,* an indicator species of anthropogenic pollution, became the abundant species for the rest of the sample. Moreover, S. *hantzschii*, a common eutrophic species, was the second dominant taxa of the sample. At the same time, a diatom with low nutrient tolerance namely *A. minitissimum* started to decrease and disappeared at the top part of the sample.

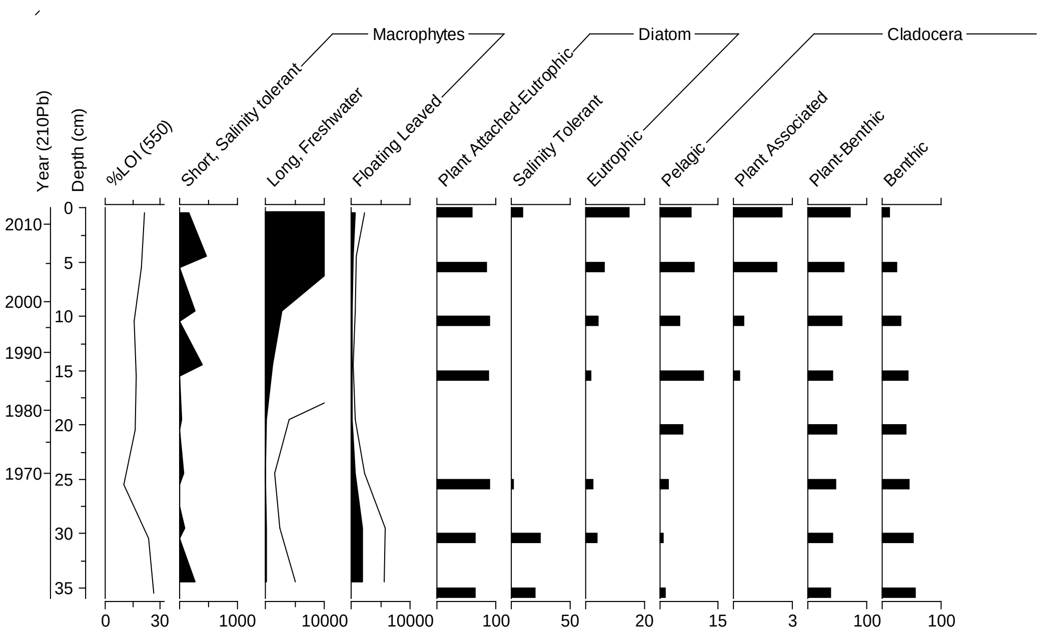

Figure 3.9 Summary diagram of Lake Gıcı core with macrophyte, diatom and cladocera data grouped by their ecological preferences.

Overall, Lake Gıcı core reflected both saline and nutrient-rich environment (Figure 3.9). The system was mainly dominated with benthic diatom and subfossil cladoceran fauna. The remaining of plant macrofossils, on the other hand, showed both showed a diverse distribution and long and short plants were spotted in the core together with floating-leaved taxa. However, it is thought that the low water level of Lake Gıcı (1,5 m) allowed the high growth of macrophytes. Although the diatom and cladoceran fauna reflected high nutrient conditions, the water clarity was not affected and allowed both benthic taxa and macrophyte growth.

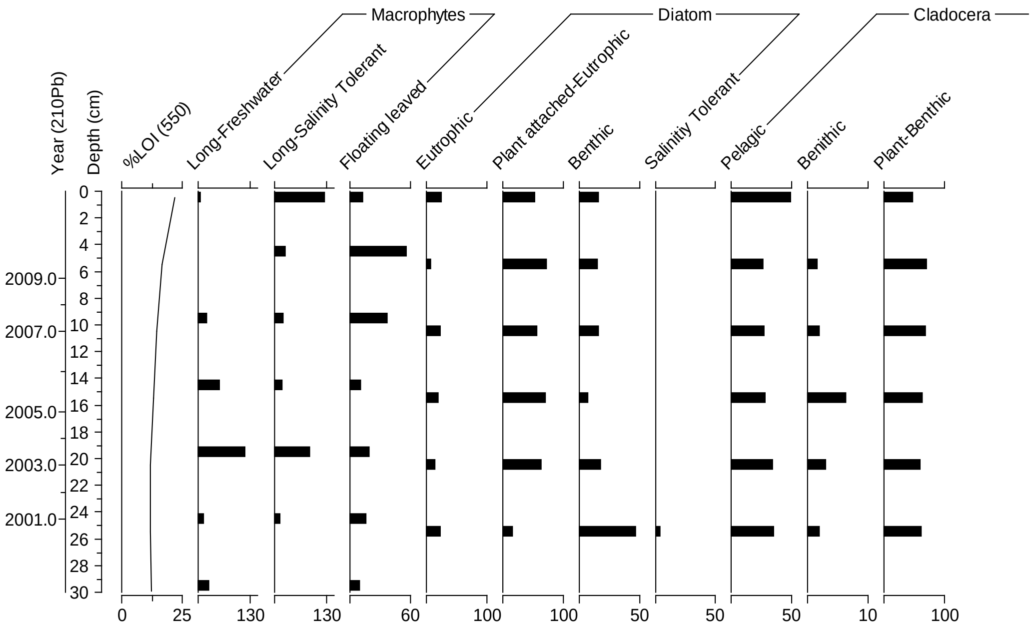

Figure 3.10 Summary diagram of Lake Balıklı core with macrophyte, diatom and cladocera data grouped by their ecological preferences.

Lake Balıklı core, on the other hand, covered only 10 years due to high sedimentation rates of the lake. All three proxies reflected nutrient rich, freshwater conditions for the environment (Figure 3.10). A small compositional shift was spotted for the diatom flora at the bottom part of the core but it was still dominated with nutrient tolerant taxa. The overall sample was dominated with benthic diatom species and cladoceran taxa as observed in Lake Gıcı. However, the presence of long, nutrient tolerant macrophytes might be the sign of deeper waters with high water clarity for the lake.

Since the absence of long term monitoring studies are limited for shallow lakes in Turkey, there is a big gap in the data sets about the past status of those environments. The negative impacts of climate change and unregulated land use, eutrophication, salinization and water scarcity are the recent challenges for Turkish lakes (Özen et al., 2010; Bucak et al., 2012; Beklioğu et al., 2017). In recent years, several studies researchers were highlighted that conductivity, nutrients and presence of macrophytes were the main environmental variables that shaped the subfossil cladoceran, plant macrofossil and diatom fauna individually for Turkish Shallow Lakes (Çakıroğlu et.al 2016; Levi et al., 2015; Bezirci et.al, in preparation-Chapter 1, Ocakoglu et al., 2015). Although the single proxy approach generated some valuable information for the region, it should be noted that each biological proxy mirrors a distinctive part of the ecosystem and the joint observation of several groups might help researchers to have a broader idea about the whole ecosystem changes and strengthen the explanation power of palaeolimnological studies (Michelutti andSmol, 2013). During the study, the compatibility of three different proxies were tested for reflecting the changes in the past status of two Turkish Shallow Lakes and it occurs that diatom fossils, subfossil Cladocerans and plant macrofossil remains established to capture the environmental changes over the past sixty years confirming the impact of conductivity, nutrients and presence of macrophytes for shaping the environment. The study also revealed that eutrophication and salinization of the shallow lakes was an ongoing process and the predicted negative impacts of climate change might enhance the impacts of these stressors for the region.

3.5 Conclusion

In this study, it appears that diatom fossils, sub-fossil Cladocera and macrophyte remains successfully traced the historical changes in Lake Gıcı and Lake Balıklı for the past 40 years. Moreover, the importance of salinity and eutrophication on shaping the fauna and flora of shallow lakes in Turkey is once again revealed. The palaeolimnological analyses showed that under the scarcity of historic or long term monitoring data, the combination of different biological proxies might help us to gain a deeper understanding of the past ecosystem changes since each proxy indicates valuable information regarding to their environment and provide new management strategies on shallow lakes.

CHAPTER 4

THE INFLUENCE OF WATER LEVEL FLUCTUATIONS ON THE DIATOM COMMUNITY OF THREE SHALLOW LAKES IN TURKEY

4.1. Introduction

Water level fluctuations which are geological, biological and climatic processes cause direct changes in habitat availability and indirectly affect chemical conditions, such as nutrient budget, turbidity and light penetration for the biological communities of the lake ecosystem (Jeppesen et al., 2015). For shallow lakes, the respond is non-linear in most cases (Coops, Beklioglu, andCrisman, 2003), since the system might change phases between clear and turbid water regardless of nutrient composition and trophic cascade (Smol and Stoermer, 2010b).In Mediterranean region, the impacts of irrigation, damming, soil erosion and groundwater drawdown have quite a significant impact on lake water levels (Coops et al., 2003) This can be specially very significant since most of the lakes are closed basin ecosystems, they tend to be more vulnerable to the changes in water level (Harrison and Digerfeldt, 1993; Shinneman et al., 2010).

Detecting long-term changes in the water table the availability of instrumental data is crucial however many shallow lakes has limited data so palaeolimnological approach might be a surrogate providing the data for longer scale investigations (Shinneman et al., 2010). The success of this method depends on the preservation quality of the biological material in the sediment. In general, diatoms that have deposited in lake sediments most commonly uses in palaeolimnology, due to their sensitivity and quick responses to the changes in their environment (Smol, 2008).

Moreover, the alteration between planktonic and non-planktonic diatom taxa may also determine the water level fluctuations (Leira et al., 2015). In low water conditions, the penetration of light across the water column tend to be higher especially to the lake bottom where the habitat availability for non-planktonic taxa, attached (benthos) or aquatic vegetation (epiphytes), increases. However, lower water level conditions might also cause a decline in diatom productivity, since especially for shallow lakes, factors such as increasing turbidity (light limitation) and nutrient limitation or enrichment is quite common during that phase (Jeppesen et al., 2000; Nõges, Nõges, and Laugaste, 2003; Vadeboncoeur et al., 2008). Besides diatom deposition, Cladocera remains, plant macrofossils and marker pigments are some other important proxies preserved in lake sediments (Davidson et al., 2011; Jiang et al., 2016). Cladocerans are used to detect changes in trophic structure, climate, eutrophication, water level fluctuations and salinity of the freshwater systems (Amsinck, Jeppesen, andLandkildehus, 2005; Jeppesen et al., 2003; Jeppesen et al., 1996; Lotteret al., 1998; Nevalainenet al., 2011) while plant macrofossils are used to infer past lake environments eutrophication, water level changes and pollution (Davidson et al., 2005; Hannon and Gaillard, 1997; Koff and Vandel, 2008)

Due to higher climate variability between summer and winter seasons along the Mediterranean region, lake water volume, depth, salinity and nutrient budget show high variations throughout the year (Beklioğlu, Altinayar, and Tan, 2006; Coppens et al., 2016; Iyigun et al., 2013; Özen, 2012). Additionally, anthropogenic factors, especially during the last century, had dramatic impacts on lake water budgets due to rapid use of ground and lake water for agriculture and irrigation (Bucak et al., 2017). For the Mediterranean region, climate change scenarios predict at least 25-30% decrease in precipitation and enhanced evaporation until 2100s (Erol and Randhir, 2012; Giorgi and Lionello, 2008). These changes have the potential to cause more dramatic seasonal water level fluctuations, and may increase the length of drought periods, which would lead to hydrological stresses for shallow lake ecosystems (Bucak et al., 2017; Molina-Navarro et al., 2014) that are more vulnerable for adscititious stress.

The present study, is as a part of a multi-proxy research within the scope of a European Union Project, REFRESH (EU FP7-ENV-2009-1) and the main aim of the current study is to investigate the responses of diatom fossils in the sediment for the recorded water level changes in three large shallow lakes from Turkey for the last 100 years.

4.2. Material and Methods

4.2.1 Study Sites

Lakes Beyşehir, Marmara and Uluabat, which are located in the mid-western part of Turkey, were sampled in October 2011 (Figure 4.1). The general characteristics of the lake were given in Table 4.1. Lakes Uluabat and Marmara are represented in the Hot-Summer Mediterranean climate zone while Lake Beyşehir is classified in the Warm-summer Mediterranean climatic zone according to Köppen-Geiger Climate classification. The lakes have been classified as important bird areas (IBA) since 2004 (Bird Life International, 2015). Moreover, Lake Beyşehir is one of the most important plant areas (IPA) in Turkey (Plant Life International, 2015) and since 1991, it has been declared as a first degree Natural Site by the Turkish Ministry of Culture. Lake Uluabat has been listed as a Ramsar site, a wetland of international importance, since 1998 (Salihoglu and Karaer, 2004).

Instrumental water-level data from these lakes were compiled by the General Directorate of State Hydraulic Works (DSI) and the General Directorate of Electrical Power Resources (EIE) of Turkey (here onwards DSI-EIE Database).

Table 4.1 General characteristics of the study sites.

Lake	Location	Surface Area (km^2)	Catchment Area (km^2)	Max. Depth (m)	Trophic status
Beyşehir*,§,¥	37°45'10.0"N; 31°30'50.0"E	730	4704	9	Oligo-mesotrophic
Marmara*	38°36'50.0"N; 27°60'55.0"E	68	508	5	Eutrophic
Uluabat*,¶	40°10'45.0"N; 28°35'30.0"E	135 - 160	10,555	4.5	Eutrophic

*** Important bird area (IBA); § Important plant area (IPA); ¥Natural site protection area; ¶ Ramsar site.**

Lake Beyşehir, one of the largest freshwater lake in the Mediterranean Basin, is located in Konya province (Figure 4.1.). The lake is fed by rivers, streams and groundwater from western and eastern catchments of Beyşehir province (Bucak et al., 2017; Nas et al., 2009). The outflow of the lake is controlled since 1908 (Altınayar, 1998), and the catchment area is surrounded mostly by agricultural lands (Bucaket al., 2017; Ciftci et al., 2010). The surface area of the lake is about 650 km^2, and the maximum depth varies between 8 and 9 meters. Most of the lake area lies within the boundaries of two National Parks (Beyşehir and Kızıldağ National Parks), and part of the lake catchment is Grade 1 Natural Site. The seasonal water level fluctuations are between 0.5 and 1 meter, and the lake was exposed to low (1928–1939, 1957–1965 and 1974–1976, 1989–2011) and high (1905–1927, 1940–1956, 1966–1973 and 1977–1988) water level periods regularly. Lake water is used for drinking and irrigation since 1914 for the Konya Basin (Oguzkurt, 2001; Bucak et al., 2016), and the lake has oligo-mesotrophic characteristics according to 2010-2012 monthly data analysis (Bucak et al., 2017).

Figure 4.1 Locations of study lakes in Turkey. Black stars represent littoral core points and white stars point the pelagic core locations. (Taken from Levi, 2016).

Lake Marmara lies in the Aegean Region-Western Turkey, was endorheic and saline until the beginning of the 20th century (Figure 4.1)(Girgin, 2000). The lake was used as a reservoir for irrigation almost 30 years (between 1932 and 1953). Several regulators and canals were built for water regulation into the area starting from 1945 until 1963 and the lake reached its maximum water storage capacity (79.2 m.a.s.l and area of 68 km^2) during 1960 (Arı and Derinöz, 2011).Currently, the lake is in hypertrophic condition due to increased nutrient inputs from anthropogenic activities (Arı and Derinöz, 2011; Gülersoy, 2013).

Located in Bursa, Turkey, Lake Uluabat had a significant lake area reduction from c. 160 km2 to c.120 km2 along with a decreasing water depth from c. 7.5 m.a.s.l. to c. 5.5 m.a.s.l. (Aksoy and Özsoy, 2002; Salihoglu and Karaer, 2004). The main water resources of the lake are Mustafa Kemal Paşa River and groundwater, and it is connected to Susurluk River by an outflow. Water level regulation has been applied since 1990s (Kazancı et al., 2010), and the lake had been categorised as eutrophic since 19th century (Reed et al., 2008). It is also surrounded by agricultural areas, having a significant impact on the nutrient levels (Salihoglu and Karaer, 2004).

4.2.2 Field Methods

Sediment cores from the three lakes were collected using a Livingstone piston corer from both littoral and pelagic points during October, 2011. The length of the littoral cores were 0.6 m, 0.9 m and 0.9 m, while pelagic cores were measured as 0.7 m, 1.6 m and 1.1 m for Lakes Beyşehir, Marmara and Uluabat, respectively. Lake Uluabat pelagic core was retrieved from the nearest deepest area to the pelagic point due to bad weather conditions. The core collection locations are presented in Table 3.2. After core collection, each sample was divided into 0.5 cm intervals for the first 20 cm except for Lake Beyşehir (first 30 cm).

Table 4.2 Summary of core locations collected in October 2011**.**

Lakes	**Location**	**N**	**E**
Beyşehir	Littoral	37°45'18.0"	31°37'34.5"
	Pelagic	37°45'01.4"	31°30'48.9"
Marmara	Littoral	38°35'59.5"	28°00'09.6"
	Pelagic	38°36'31.8"	27°58'53.0"
Uluabat	Littoral	40°12'53.1"	28°29'06.2"
	Pelagic	40°12'04.0"	28°29'08.5"

4.2.3 Laboratory Methods

Both pelagic and littoral cores were analysed for biological and geochemical proxies as well as all of them were dated with ^{210}Pb dating method. The preliminary examination of the cores revealed that the diatom preservation in pelagic cores were significantly low, therefore they were removed from the study and only littoral cores were selected for further detailed analyses to determine the change of the sedimentary proxies along with the water level fluctuations and to compare the results of other proxies with diatom distribution. ^{210}Pb, ^{226}Ra, ^{137}Cs and ^{241}Am analyses were carried out at Environmental Radiometric Facility at University College London by ORTEC HP Ge GWL series well-type coaxial low-background intrinsic germanium. Following, constant rate of supply (CRS) dating model was applied for the calculation of lake ^{210}Pb chronologies. Heiri et al. (2001) was used for measuring Loss on Ignition (LOI) at 105°C, 550°C and 950°C for measuring the water, organic matter and carbonate contents of the sediment. Geochemical and trace element analyses from the sediment samples were done at University College London, using a Spectro XLAB2000 X-ray fluorescence (XRF) spectrometer.

During diatom sample and slide preparation, methods from Battarbee (1986) were applied. Leica DMI4000B inverted microscope with 1000x magnification at Middle East Technical University was used for counting and identification (Cleve-Euler A., 1922; Hustedt, 1966, 1976, Krammer andLange-Bertalot, 1986, 1988, 1991a, 1991b) of diatom valves. At least 500 valves were counted from each slide and the diatom taxa were grouped into planktonic and non-planktonic for further use. C2 programme was applied to the data for constructing diatom stratigraphic diagrams for three shallow lakes (Juggins, 2007).

4.3 Results

The results of the Lake Beyşehir, Lake Marmara and Lake Uluabat were mainly focused on the diatom distribution of littoral cores for the lakes. The diatom valve preservation was low/absent in the pelagic cores of all three lakes so only littoral cores were used in the study to achieve a data set of diatoms.

4.3.1 Lake Beyşehir

Eighty three years of water level data revealed two different phases for Lake Beyşehir as low (1928-1939, 1957-1965, 1974-1976 and 1989-2011) and high (1905-1927, 1940-1956, 1966-1973 and 1977-1988) water levels and the mean water level was calculated as 1123.3 m.a.s.l (Figure 4.3).

The unsupported ^{210}Pb activity was relatively higher in top parts of the Lake Beyşehir littoral core (between 5 cm and 0 cm) and reached the maximum value around 5 cm. which was a sign of sediment accumulation increase in the sample (Table 4.3). Additionally, the increase in sedimentation rates from 1990s to present were supported the findings (Figure 4.2). On the other hand, high ^{137}Cs activities in the surface part might be related with soil wash from the catchment and resuspension of the sediment in recent years. The peak detected between 17 and 10 cm for ^{137}Cs activity was corresponded to 1963 when the ^{137}Cs activity was reached its maximum due to the increase of atmospheric testing of nuclear weapons. Concordantly, the constant rate of supply (CRS) model suggested that year 1963 was placed between 11.8 and 13.3 cm and the sample was dated back to 1900s for 17 cm. Below that point, dating was not accomplished due to low unsupported ^{210}Pb activities.

Table 4.3 Radiometric analysis (^{210}Pb and ^{137}Cs) of Lake Beyşehir littoral core

Depth	Dry Mass	^{210}Pb Supported		^{210}Pb Unsupported		^{137}Cs		Chronology		
								Date	Age	
(cm)	(g cm^{-2})	(Bq Kg^{-1})	±	(Bq Kg^{-1})	±	(Bq Kg^{-1})	±	(AD)	(yr)	±
								2011	0	
0.0-0.5	0.13	51.31	4.06	58.03	15.13	43.92	2.99	2011	0	2
3.0-3.5	1.82	28.84	2.56	65.89	10.95	36.71	2.29	2004	7	2
5.0-5.5	3.00	30.12	2.10	80.06	9.78	36.40	1.82	1997	14	2
8.0-8.5	4.90	35.40	2.27	44.80	9.28	29.08	1.75	1985	26	3
10.0-10.5	6.22	31.00	1.97	47.85	8.49	34.79	1.65	1976	35	4
11.0-11.5	7.26	32.13	2.26	57.24	10.09	35.13	1.90	1965	46	5
13.0-13.5	8.35	33.00	2.22	29.43	8.91	36.56	1.84	1951	60	7
15.0-15.5	9.95	30.61	1.18	20.22	4.68	31.39	0.95	1932	79	12
16.5-17.0	11.26	27.86	1.26	23.15	4.83	33.06	1.05	1903	108	26
18.0-18.5	12.59	28.63	1.44	3.12	5.39	30.00	1.12			
20.0-20.5	14.36	30.96	1.61	3.07	5.94	11.82	0.99			
23.0-23.5	17.10	27.18	1.22	-4.74	4.27	0.87	0.52			

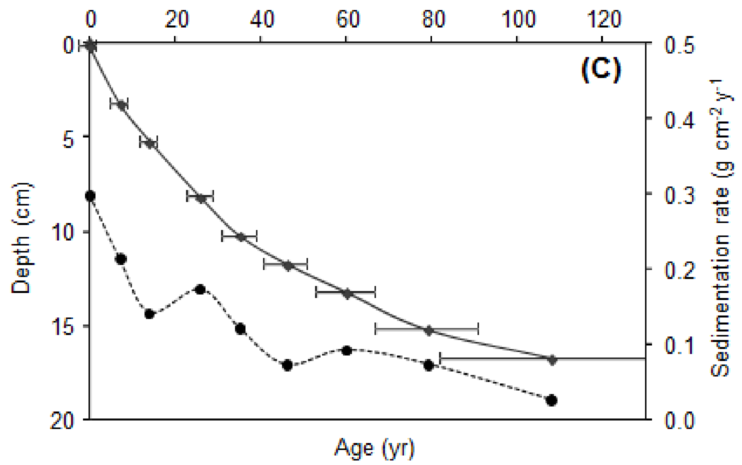

Figure 4.2 Chronology of Lake Beyşehir core with CRS model ^{210}Pb dates and sedimentation rates. Dashed lines represents sedimentation rates while solid lines shows age throughout depth.

LOI950, DW and LOI550 followed the same pattern throughout the sample as they reduced between surface and 15 cm (1932 ±12) which corresponds to the drought period (Figure 4.2). Moreover, sediment indicators implied an enhancement of terrigenous/mineral indicators (i.e. Ti, Zr) . The sedimentation rate were increased from bottom to the top part of the sample and the maximum value was 0.28 g cm^{-2} yr^{-1} (Figure 4.2).

Thirty nine diatom species were identified during the investigation of Lake Beyşehir down core and species that presented lower than 4% were eliminated from the visual presentation. Before 1990s, the lake was completely dominated with benthic taxa. The abundant species (~ 50 %) were *Navicula subrotundata* (Hustedt) and *A. pediculus* (Figure 4.3). The presence of *C. disculus* (Schumann), *C. placentula and P. brevistriata* were also found between 19 and 11 cm (Between 1900s and 1960s, respectively) of the sample. A planktonic species *A. granulata*was also appeared around 11 cm but the abundance was below 5% (Figure 4.3). Plant attached species *E. sorex* was first found at 11 cm of the core. At the same point, the abundance of another plant attach species *C. placentula* was reached to its maximum (20 %) and the amount of *N. subrotundata* was decreased to 20 % (Figure 4.3). *A. pediculus* was still the dominant species of the core at 11 cm together with low amounts of *C. disculus, C. placentula, P. brevistriata, U. ulna, C. dubius* and *C. ocellata.* Until 1980s, there was not a significant change spotted for the sample but starting from 7 cm, planktonic species *C. ocellata* and *A. granulata* were increased all the way to the top part of the sample (Figure 4.3) but the sample was still dominated with benthic species. Thetop 0-1cm (2011) however, was dominated with both planktonic (*C. ocellata* and *A. granulata*) and non-planktonic taxa (*N. subrotundata* and *A. pediculus*), where the lake was in a low water level phase (Figure 4.3). Moreover, planktonic taxa *C. ocellata* reached the maximum abundance around 30% at the surface part of the core while the abundance of *N. subrotundata* and *A. pediculus* were dropped around 15 % (Figure 4.3).

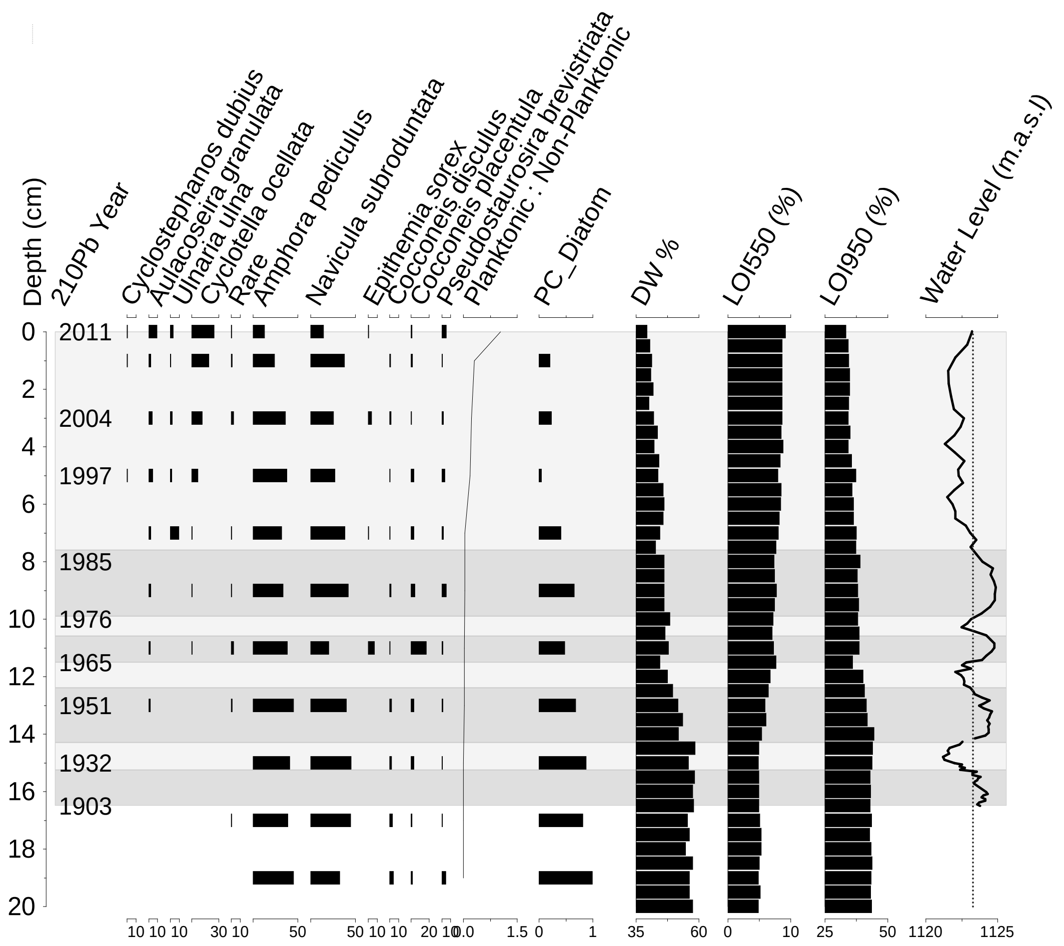

Figure 4.3 Summary diagram of Lake Beyşehir diatom distribution, LOI (550 and 950), DW and water level data (mean water level). While the light grey areas representing low water years, dark grey areas showing the high water periods.

4.3.2 Lake Marmara

The instrumental water-level data from Lake Marmara covered a rather shorter period of 40 years (1970 to 2011) (Figure 4.7). The mean water level was 75.7 m.a.s.l; the highest water level was recorded around 2011 (79 m.a.s.l.), and the lowest water level (73 m.a.s.l.) was spotted around 1992's when the lake was facing a serious dry out. During the year of 2008, a dam was constructed that improved the water level, and resulted in the peak level of 2011 (Figure 4.5).

The changes in the unsupported 210Pb activities were irregular in Lake Marmara littoral core and the maximum activity was recorded around 8 cm (Table 4.4). Moreover, the variation in the sedimentation rates and low amount of unsupported 210Pb activities caused difficulties for the interpretation of the age-depth relationship for the sample. Anyhow, CRS model was suggested that 14.3 cm corresponded to 1963 and the core was dated back to 1940s. The irregular pattern of the sedimentation rates also observed for the sample but there was an increase detected for the upper part of the sample (Figure 4.5).

Table 4.4 Radiometric analysis (^{210}Pb and ^{137}Cs) of Lake Marmara littoral core

Depth (cm)	Dry Mass (g cm^{-2})	^{210}Pb Supported (Bq Kg^{-1})	±	^{210}Pb Unsupported (Bq Kg^{-1})	±	^{137}Cs (Bq Kg^{-1})	±	Chronology Date (AD)	Chronology Age (yr)	±
								2011	0	
0.0-0.5	0.12	56.26	6.14	3.97	26.02	0.00	0.00	2011	0	2
1.0-1.5	0.60	39.80	2.50	18.42	12.37	2.41	1.40	2010	1	2
2.0-2.5	1.08	39.46	3.03	21.83	14.88	0.00	0.00	2009	2	2
3.0-3.5	1.57	41.64	2.53	7.04	11.32	4.65	1.28	2008	3	3
4.0-4.5	2.07	41.56	1.64	24.92	7.74	3.44	0.85	2007	4	3
5.0-5.5	2.57	35.58	1.95	19.81	9.20	3.83	1.08	2005	6	3
6.0-6.5	3.08	38.75	1.52	21.02	7.19	0.00	0.00	2003	8	4
7.0-7.5	3.70	34.62	1.25	27.12	6.07	1.85	0.67	2000	11	4
8.0-8.5	4.32	32.69	2.10	43.57	9.87	3.44	1.21	1995	16	5
9.0-9.5	5.09	35.98	1.41	33.06	7.24	2.88	0.74	1988	23	7
10.0-10.5	5.86	37.17	1.14	15.98	4.92	2.63	0.58	1982	29	9
11.0-11.5	6.60	34.89	1.14	11.78	5.35	5.21	0.65	1978	33	11
12.0-12.5	7.35	33.80	1.02	14.15	4.98	2.47	0.56	1973	38	13
14.0-14.5	8.73	38.87	2.24	16.04	10.45	6.76	1.28	1962	49	18
16.0-16.5	10.12	39.93	2.17	15.16	10.07	4.98	1.25	1943	68	23
18.0-18.5	11.60	34.82	2.34	12.39	10.64	5.89	1.34			
20.0-21.0	13.57	38.11	1.40	-3.14	6.25	6.10	0.83			
22.0-23.0	15.80	34.03	1.53	-0.84	6.55	3.42	0.80			
26.0-27.0	20.78	33.59	1.01	1.77	4.58	0.00	0.00			
28.0-29.0	23.49	34.01	1.65	5.76	7.84	1.53	0.80			
30.0-31.0	26.28	27.45	1.05	0.19	4.71	0.69	0.48			
34.0-35.0	31.70	31.42	1.25	-3.96	5.69	3.04	0.63			

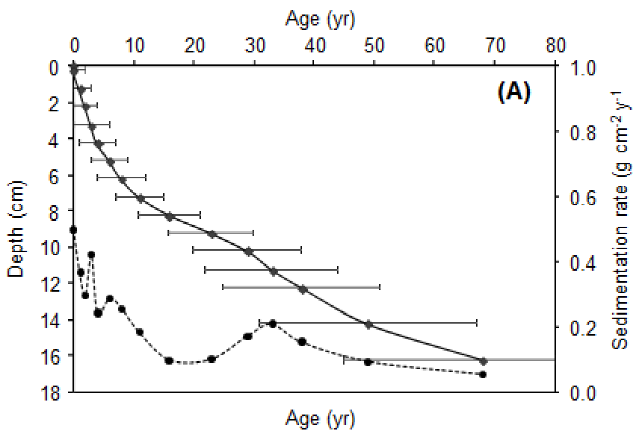

Figure 4.4 Chronology of Lake Marmara core with CRS model ^{210}Pb dates and sedimentation rates. Dashed lines represents sedimentation rates while solid lines shows age throughout depth.

While LOI950 and LOI550 followed a lower profile (10% and 5%) between 23 and 40 cm, which corresponds to a low water period, DW values were higher (80 %) during that time and a significant decrease of DW appeared above 24 cm which clearly pointed out the construction period (Figure 4.5). At that point, the amount of sediment minerals (Rb:Sr, Ti, Zr) reduced as well. Moreover, during 1980s, another decrease in sediment elements pointed out a change in sediment composition. There was a decrease at LOI550 values (3.5 %) around 8-9 cm (c. 1990-1995) and they remained constant through the surface of the sample afterwards (Figure 4.5). The sedimentation rate, on the other hand had three peaks. The first peak was appeared around 1980s. The second peak was spotted at 2008 while the third peak was at the top part of the core (Figure 4.4).

There were 32 species found in total in the Lake Marmara down core sample. The core was dated until the 16th cm and the counting was applied until the 18cm. Below that point, the counting could not be applicable due to low preservation of diatom valves. Lake Marmara core was dominated with non-planktonic and eutrophication tolerant species. The bottom part of the Lake Marmara core was dominated with *F. capucina* (Desmazières) (65 %) and *C. placentula* and R. *abbreviata* (C.Agardh) were the other abundant (15 % and 10 %) species of the sample. Moreover, the presence of other benthic taxa namely *B. paradoxa*, *N. hungarica* (Grunow), *A. libyca*, *A. pediculus*, *E. sorex* and *Rhopalodia gibba* (Ehrenberg) were found at the bottom part of the core together with *U. ulna* and *A granulata* with low abundances. *F. capucina* was dominated the core until 1982s (10 cm) but afterwards the abundance of *F. capucina* were dropped to almost 10 % while *C. placentula* became the abundant species and planktonic turbid water indicator species *A. granulata* reached the maximum abundance of 30 % (Figure 4.5). Between 8 and 2 cm, the abundance of *C. placentula* was decreased to almost 10 % while *F. capucina* was increased to 50 %. *R. abbreviata* had a peak (up to 10 %) around 6 cm but the presence of the species were remained low until the top part of the core (Figure 4.5). Epiphytic *E. sorex* was also reached to maximum abundance (30%) around 2009s. The dominant species of the top part of the core (2011) was *F. capucina, C. placentula* and *A. minutissimum*. Moreoover, C. *dubius, C. affinis, E. sorex, A. pediculus, U. ulna, A. granulata, A. libyca* and *B. paradoxa* was found in the sample.

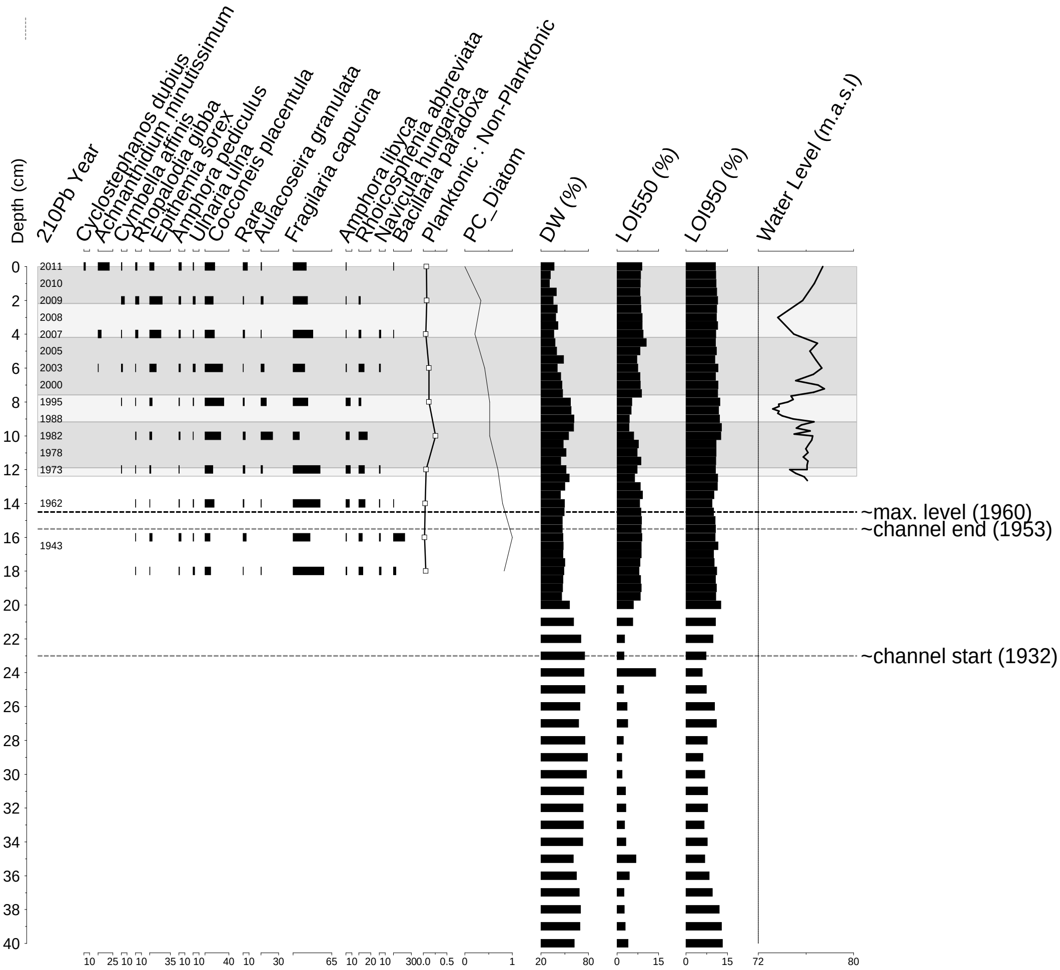

Figure 4.5 Summary diagram of Lake Marmara diatom distribution, LOI (550 and 950), DW and water level data (mean water level). While the light grey areas representing low water years, dark grey areas showing the high water periods.

4.3.3 Lake Uluabat

Lake Uluabat had a yearly water level changes around 1 m since the 1960s. Between the years of 1960 and 1984, there was a long-term high water level phase followed by a long term low water level period starting in 1985 until today (Figure 4.7). The lowest water level was recorded in 2001 with 3.96 m.a.s.l. while the highest water level was recorded around 1980s.

Due to low amount of ^{210}Pb activities and high counting errors, dating of Lake Uluabat core was rather difficult. The irregular decrease in the unsupported ^{210}Pb activities indicated changes in sedimentation rate (Table 4.5). While the ^{137}Cs activity peaked between 26 and 35 cm which was related with the 1963 fallout maximum from the atmospheric testing of nuclear weapons, according to ^{210}Pb CRS model 1963 was corresponded with 18.5 cm. Henceforth, the corrected chronologies together with CRS model calculated sedimentation rates placed 1963 at 34.5 cm. Apart from two peaks detected around 1960s and the 1970s, sedimentation rates of the sample were constant and the mean was 0.28 g cm^{-2} yr^{-1} (Figure 4.6).

Table 4.5 Radiometric analysis (^{210}Pb and ^{137}Cs) of Lake Uluabat littoral core

Depth	Dry Mass	^{210}Pb Supported		^{210}Pb Unsupported		^{137}Cs		Chronology		
								Date	Age	
(cm)	(g cm^{-2})	(Bq Kg^{-1})	±	(Bq Kg^{-1})	±	(Bq Kg^{-1})	±	(AD)	(yr)	±
								2011	0	
0.0-0.5	0.05	62.86	8.36	28.57	30.73	0.00	0.00	2011	0	2
2.5-3.0	1.06	51.94	4.33	47.50	15.62	13.08	2.50	2008	3	2
7.0-7.5	3.26	50.58	3.27	42.31	13.32	8.65	1.76	2000	11	3
12.0-12.5	5.77	48.74	2.55	28.71	10.14	11.15	1.36	1991	20	4
14.5-15.0	7.14	50.50	1.33	19.67	5.28	12.82	0.75	1987	24	4
16.5-17.0	8.31	44.96	2.06	21.90	8.26	10.96	1.16	1984	27	5
18.0-18.5	9.22	44.05	2.53	37.98	10.27	16.94	1.60	1979	32	5
19.5-20.0	10.15	47.44	2.23	11.03	8.11	13.31	1.19	1975	36	6
22.0-23.0	11.92	48.27	2.13	2.76	7.00	17.02	1.31	1974	37	6
24.0-25.0	13.20	50.09	1.40	0.60	5.39	19.13	0.93	1973	38	6
26.0-25.0	14.49	48.57	2.27	11.49	8.79	21.13	1.57	1972	39	5
28.0-29.0	15.79	47.93	2.12	25.64	8.90	20.72	1.39	1966	45	6
30.0-31.0	17.14	54.41	2.00	-6.06	7.12	19.59	1.24	1965	46	6
32.0-33.0	18.49	50.17	1.53	-2.74	5.35	21.66	1.00	1964	47	5
34.0-35.0	19.92	48.86	1.41	5.09	5.56	21.74	0.95	1963	48	5
38.0-39.0	22.74	48.94	2.22	13.83	8.84	17.92	1.48	1956	55	8
40.0-41.0	24.14	48.68	1.75	2.63	6.69	10.69	1.01			
44.0-45.0	26.84	46.22	1.41	-1.56	5.15	3.22	0.61			
56.0-57.0	35.21	45.80	1.25	1.95	4.81	0.00	0.00			

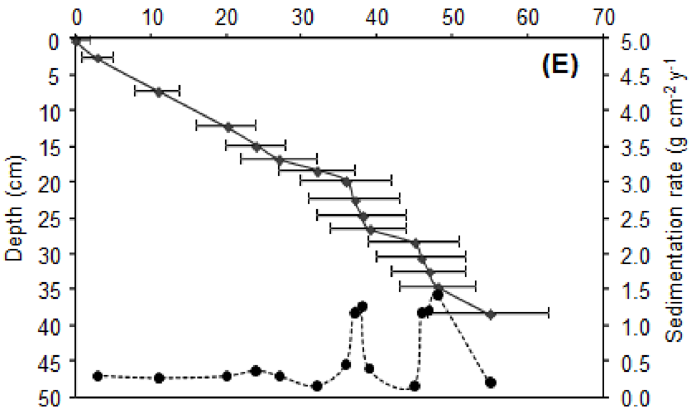

Figure 4.6 Chronology of Lake Uluabat core with CRS model ^{210}Pb dates and sedimentation rates. Dashed lines represents sedimentation rates while solid lines shows age throughout depth.

The homogenous structure of the core resulted in a linear increase for LOI550 values starting from 20 cm until 0 cm of the core except for a little variation around

7 cm (7%) and 1 cm (11%), and lower LOI550 values addressed a quick degradation and recycle of the organic material (Figure 4.7). Increased Br was another evidence for higher organic content for the first 20 cm. Moreover, an increase was detected for LOI950, Ca and Sr, but mineral indicators were decreased for the same period.

The three common species of Lake Uluabat core was *A. granulata*, *C. placentula* and *E. sorex* (Figure 4.7). Between bottom and 13 cm (around 1990s), the abundance of plaktonic *A. granulata* was reached to almost 45 % while benthic, non-planktonic plant attached taxa *C. placentula* and *E. sorex* were represented with lower abundances (20% and 18 %, respectively). Moreover, the benthic habitat of the sample were included *A. pediculus, E. adnata, B. paradoxa* and *F. capucina*. Planktonic taxa *C. dubius* and *C. meneghiniana* were also spotted for the period but in very low abundance (> 10 %). Between 12 cm and 1 cm (mid 1990s until 2010s), the dominance of plant attached species of *E. sorex* and *C. placentula* were increased while turbid water indicator *A. granulata* was also represented with 15% average abundance (Figure 4.7). Additionally, the abundance of *F. capucina* and *A. pediculus* were increased up to 10 % for the period. However, eutrophication indicator pelagic species *C. dubius* were increased to 25 % and dominated the core with benthic *E. sorex* for surface part of Lake Uluabat sample (Figure 4.10). The other species were spotted in the core was *C. meneghiniana, B. paradoxa, E. adnata, A. pediculus, C. placentula* and *A. granulata* (Figure 4.7).

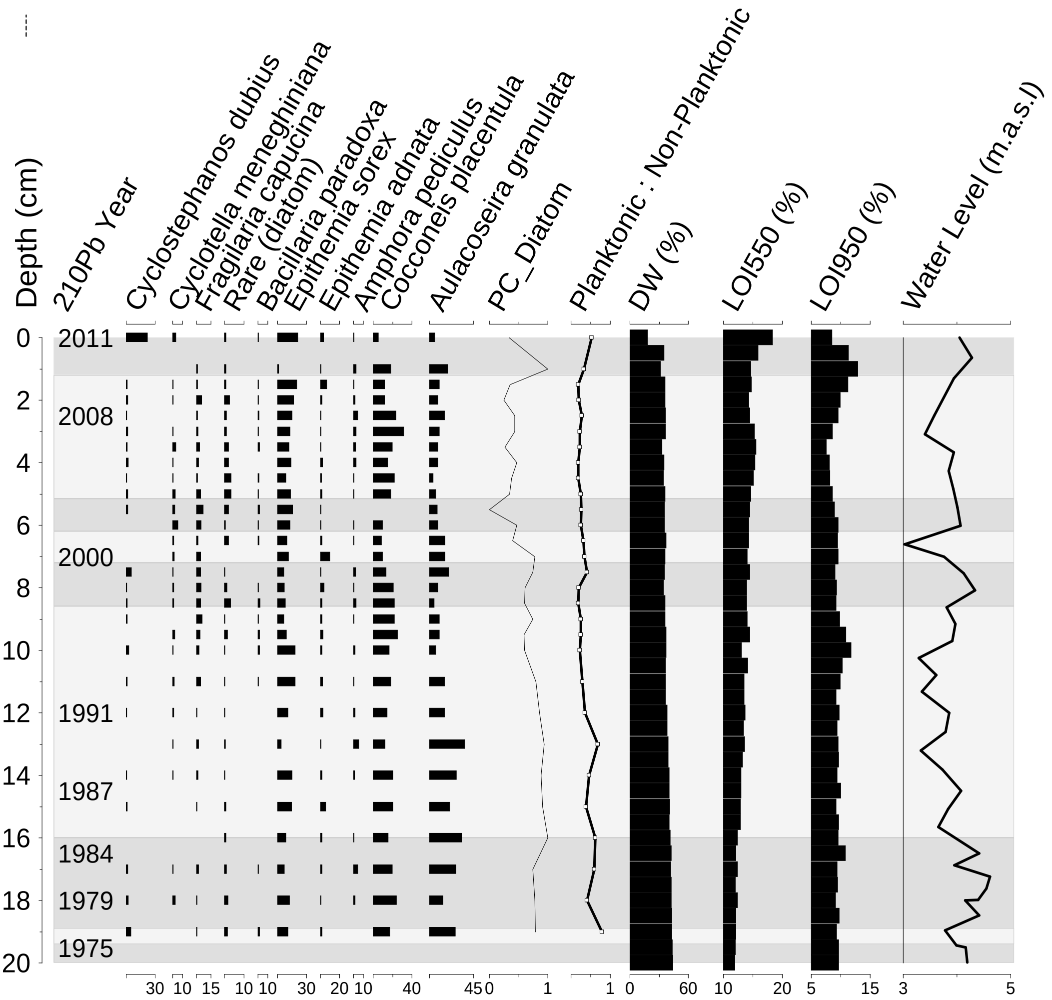

Figure 4.7 Summary diagram of Lake Uluabat diatom distribution, LOI (550 and 950), DW and water level data (mean water level). While the light grey areas representing low water years, dark grey areas showing the high water periods.

4.4. Discussion

In this study, three large shallow lakes from Turkey (Lakes Beyşehir, Marmara and Uluabat) were investigated using palaeolimnological techniques and the existing instrumental water level data was accomplished to enlighten the impacts of water level changes on the diatom community for the last 100 years.

Lake Beyşehir core was dated back to 1905s, and water level records revealed both low and high water level phases in eight different periods, but unfortunately the response of the diatom species for capturing water level changes were relatively weaker than expected from the littoral core. Except from the surface part, the sample were mainly dominated with two small benthic taxa *N. subrotundata* is a small raphid taxa and *A. pediculus*, a fast growing pioneer species which easily adapts to and dominates the existing environments (B-Béres et al., 2014). Several studies revealed that the presence of benthic diatom taxa indicates lower water levels for the lake ecosystems (Riis and Hawes, 2003; Sarmaja-Korjonen and Alhonen, 1999; Moos, Laird, and Cumming, 2005). However, in Lake Beyşehir, the diatom composition was not responsive to water level changes significantly and the benthic flora dominance were accomplished in both high and low water levels. Moreover, during the high water level period (1900s to mid-1980), benthic associated Cladocerans and short-growing plants were also found in high abundances together with benthic diatom flora in Lake Beyşehir littoral core. According to Bucaket al., (2017), the lake has oligo-mesotrophic characteristics and it had been suggested that the growth of benthic habitat during high water years might be correlated with high light penetration through the water column in a low turbid environment.

The existence of planktonic species *C.ocellata* and *A. granulata* were captured starting from 1940s, but they remained in lower abundances until the top part of the core. *C.ocellata* is a cosmopolite diatom found in littoral zone of the shallow lakes and epilimnetic habitat in the surface waters, which have a rather broad tolerance of nutrients (Houk, Klee, and Tanaka, 2010) from ultra–oligotrophic (Cvetkoska, Reed, and Levkov, 2012) to eutrophic (Schlegel and Scheffler, 1999) conditions. On the other hand *A. granulata* is a eutrophic water species (Gomez, Riera, and Sabater, 1995; Hötzel and Croome, 1996)that favours well- mixed environments (Agbetiet al., 1997). Due to the large open surface area, Lake Beyşehir is affected by wind turbulence significantly (Fethi et al., 2014; Kazancı et al., 2010) and wind induced sediment resuspension was detected especially for the littoral parts of the lake according to the unsupported ^{210}Pb activity results. Correspondingly, the Secchi depths were lower in the area compared to the pelagic zone (Dalkiranet al., 2006) which might explain the higher abundance of planktonic species *C.ocellata* and *A. granulata,* since those factors created suitable conditions for the species. Moreover, the reduced abundance of benthic diatom species in the surface part might also correlated with reduced light conditions and high turbidity since Beklioğlu *et al.* (2014) also reported low Secchi depth for the Lake.

Within the period investigated (1940-2011), the sedimentary diatom flora in Lake Marmara is indicative of moderately eutrophic conditions rather than showing clear impacts of water level fluctuations. Changes in the planktonic:benthic diatom ratio in lake sediments have been offering information about water level fluctuations in the system while the abundance of open-water taxa is correlated with high water levels, benthic taxa is a sign of lower water conditions (Leira et al., 2015; Wolin and Stone, 2010). However, in Lake Marmara core, the system was mainly dominated with benthic taxa expect for the presence planktonic species *A. granulata* which corresponds with higher nutrient content and turbidity in the water column (Gomez et al., 1995) between 1980s and 1990s during the high water level. The overall dominance of epiphytic *F. capucina*, a mesothropic species, might be explained by its high dispersion capacity in the littoral area since, according to Holland (1993) *F. capucina* is a common member of near shore diatom communities. On the other hand, during the high water years until 1970s, *F. capucina* might showed tychoplanktonic features and switch between benthic to planktonic (Buczkó, Korponai, and Padisák, 2010). Correspondingly, Schmidt et al.(2004) found the positive relationship between *F. capucina* complex and increasing depths in Austrian lakes.

Lake Marmara was surrounded with agricultural sites and the settlement around the catchment area increased quite significantly especially after 1970s (Gülersoy, 2013). Moreover, the shift from an endorheic to an open and deeper lake system and the high demand of lake water for irrigation purposes throughout the time affected the trophic status of the environment which categorised as hypertrophic (Gülersoy, 2013). Therefore, after 1970s, the diatom composition of the lake was reflecting nutrient rich conditions. Higher abundance of *C. placentula* and *A. minitissimum* might have a strong correlation with nitrogen availability in the water column due significant impact of anthropogenic stressors (intensive agricultural activities and settlement) on the lake since they are categorised as nitrogen autotrophic taxa and found in environments with moderate to high nutrient levels (meso-eutraphentic/eutraphentic taxa) and moderate levels of pollution (b-mesosaprobous) (Weilhoefer and Pan, 2007). High presence *A. minutissimum* at the surface part along with high water level period also correlated with the wide tolerance of the species to light depletion. Thus, Lange et al. (2012) showed that the abundance *A. minutissimum* increased with lower light conditions.

Lake Uluabat core involved high abundance of planktonic *A. granulata* in contrast to Lakes Beyşehir and Marmara and the benthic community was mostly expressed by epipehytic *E. sorex* and *C. placentula*. Even though distribution of diatoms were used for detecting water level fluctuations (Brugam, Mckeever, and Kolesa, 1998; Laird and Cumming, 2008) in many cases, due to complex interactions of various factors like temperature, light availability and nutrients on diatom species composition, predicting the response of the species for a single factor might be misleading (Laird et al., 2011). In Lake Ulubat case, the occurrence of the simplistic impact of water level changes on the diatom fauna were not clear. It had been reported that Lake Uluabat was under a serious pressure of anthropogenic impacts coming from the surrounding catchment area that caused eutrophication with a higher input of nutrients to the ecosystem (Elmaci, 2008; Yenilmez and Aksoy, 2013). Hence the water quality of the lake was degraded to second-fourth/worst class from first-second class in 2000s (Salihoglu and Karaer, 2004). Moreover, increased Br was another evidence for higher organic content for Lake Uluabat. The diatom composition of the core were also reflected eutrophic conditions. The presence of *A. granulata* and *C. meneghiniana* are linked to nutrient richness in lake ecosystems (Dong et al. 2008). Moreover, *A. granulata* has a higher tolerance to turbulence, deforestation and erosion (Dong et al. 2008; Fontana et al., 2013). Since Lake Uluabat was under the effect of wind induced resuspension that caused continuous sediment mixing into the lake (Kazancı et al., 2010; Reed et al., 2008), that might have created favourable conditions for continuous *A. granulata* growth together with high nutrients input from the catchment. Additionally, presence of *C. dubius* at the surface sediment indicated a serious eutrophication state for Lake Uluabat as well, since this diatom is a common planktonic species found in nutrient-rich lakes in Europe (Bradshaw and Anderson, 2003; Hickel and Hakansson, 1987). Moreover, the existence of epiphytic species *E. sorex* and *C. placentula* might be correlated with the macrophyte coverage since Levi et al. (2016) revealed the high abundances of macrophytes between 1975 and 2011. The high occurrences of *C. placentula* is another sign of nutrient enrichment in Lake Uluabat. Concordantly, Bennion, (1994) reported that *C. placentula* is a common epiphyte of eutrophic waters in Europe with high tolerance to eutrophicaton.

Due to their sensitive response capacity to the environmental changes, diatoms are frequently used for detecting the past status of lake ecosystems (Dixit et al., 2001; Gell et al., 2007; Reid and Ogden, 2009). Moreover, correlation between diatom distribution and lake levels were issued to reconstruct past water level fluctuations for many cases (Barker, Fontes, and Gasse, 1994; Brugam et al., 1998; Laird and Cumming, 2008; Laird et al., 2011). Hence, the factors that influence diatom distribution (aquatic macrophytes, water clarity, etc.) are in such complicated relationship (Laird et al., 2011) that the diatom community might not reveal the real ecosystem response in that sense. In this study, the response of diatom flora for the fluctuations in the water level was not clear and there are several possible factors that might explain the poor relationship. The instrumental water-level and the core data was covering the latest 100 years and the length of the water level intervals might be relatively short to accomplish a significant change in the composition of the diatom flora. A study by Punning and Puusepp (2007) were also revealed that the relationship between water depth and planktonic/periphytic diatom ratio was rather weak in shallow waters (>3-4m). Moreover, especially after 1970s, the eutrophication signal was more influencing for shaping the diatom composition of the lakes in the study than water level changes. A study, from Estonia reported the same pattern for large-shallow Lake Võrtsjärv (Heinsalu et al., 2008). Especially for the Lake Marmara, the changes related with water levels were human-induced and the site was heavily under the stress of pollution during the study period so the reaction of the diatom flora was more correlated with anthropogenic impacts compared to water level changes.

4.5. Conclusion

Due to distinct differences between the responses of lake ecosystems to the water level changes, making predictions about past status of such environments are difficult. This study investigated the response of fossil diatom data to water level fluctuations obtained from instrumental water-level data however, the diversity of diatom flora was in a poor agreement with the water level changes. Though, the negative impact of eutrophication was captured especially for Lake Uluabat and Lake Marmara after 1970s. It is suggested that diatom fossils are still good indicators for capturing the environmental changes and can be used in the absence of historical data. However, to achieve clear signs of the composition alteration, a long-term perspective is needed and the complexity of multiple drivers for shaping the community structure of the diatom flora is highlighted.

CHAPTER 5

CONCLUSIONS

The scope of the current study was:

i). to determine the significant environmental variables that structured the diatom assemblages of Turkish Shallow Lakes and gatherinformation about ecological preferences.

ii). Reconstructing the historical changes of two shallow lakes to determine the changes in the short cores using a multi proxy approach and define the past environmental status these lakes. Iii). to investigate the responses of diatom fossils in the sediment for the recorded water level changes in three large shallow lakes from Turkey for the last 100 years.

The results suggested that conductivity was the main variable shaping diatom fossils in surface sediment samples. While the down core investigations revealed the significant impact of eutrophication on the past diatom flora and also hints of past salinization.The passive plotting of the core data were pointed the influence of PVI % on the historical changes in study lakes.

The application of the multi-proxy approach for capturing the environmental changes for two shallow lakes were confirmed the influence of salinity and eutrophication on shaping the past fauna and flora of shallow lakes in Turkey.

No significant overlap revealed on the comparisons between instrumental water-level data and the fossil diatom data. On the other hand, the negative impact of eutrophication was captured by the interpretation of the compositional changes in the diatom fossils especially for Lake Uluabat and Lake Marmara after 1970s.

Diatom fossils are widely used in palaeolimnological research and they provide crucial information about the past status of lakes. The study confirmed the accuracy of diatom fossils for tracking the past environmental changes for shallow lakes in Turkey and provided information about the recent historical changes for the selected lake ecosystems. The results were emphasized the consideration of complex relationship between the biological proxies and environmental variables and the distinct responses of the proxies for different lakes while interpreting the past ecosystem changes in shallow lakes. Under the scarcity of historic or long term monitoring data, palaeolimnological approaches might help researchers to gain a deeper understanding of the past ecosystem changes and allow policy makers to develop new management strategies.

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APPENDIX A

DIATOM DISTRIBUTION OF STUDY LAKES

This appendix is prepared for CHAPTER 2 and included the diatom fauna of 32 shallow lakes used in the study.

Table A. Diatom species in 32 shallow lakes. Lake numbers were corresponded to 1: Hamam, 2: Golcuk_B, 3: Gokgol, 4: Sakli, 5: Derin, 6: Karagol-İ, 7: Saka, 8: Ince, 9: Golcuk_Ö, 10: Azap, 11: Emre, 12: Pedina, 13: Karagol_D, 14: Golcuk_S, 15: Abant, 16: Cubuk, 17: K.Akgol, 18: Eymir, 19: Golhisar, 20: Gerede, 21: Kaya, 22: Keci, 23: Balikli, 24: Koca, 25: Karagol-B, 26: Uyuz, 27: Sarp, 28: Kaz, 29: Taskısığı, 30: Eğri, 31: Seyfe, 32: Gici.

Diatom	Lake	Species abbreviations
Achnanthidium exiguum (Grunow) Czarnecki	13,14	AExigu
Achnanthidium microcephalum Kützing	25	AMicro
Achnanthidium minutissimum (Kützing) Czarnecki	1,2,4,5,6,7,8,11,13,14,15,16,21,22,23,25,27,28,31	AMinut
Psammothidium sacculus (J.R.Carter)	13	PSaccu
Amphipleura rutilans Cleve	13	ARutil
Amphora coffeiformis (C.Agardh) Kützing	2,3	ACoffe
Amphipleura pellucida (Kützing) Kützing	7	APellu
Amphora libyca Ehrenberg	4,5,7,8,16,19,31	ALibyc
Amphora commutata Grunow	27	ACommu
Amphora ovalis (Kützing) Kützing	7,14,19,20,24,25,26,27,30	AOval
Amphora pediculus (Kützing) Grunow ex A.Schmidt	13,15,17,24,27,28,29,31,32	APedic
Amphora veneta Kützing	3,10,13,32	AVenet
Anomoeoneis costata (Kützing) Hustedt	26	ACosta
Anomoeoneis sphaerophora Pfitzer	3,13,26	ASphae
Aulacoseira ambigua (Grunow) Simonsen	1,9,17	A ambigua
Aulacoseira granulata (Ehrenberg) Simonsen	6,7,10,11,16,17,19,20,22,24,25,27,28,29,31	AGranu
Aulacoseira muzzanensis (F.Meister) Krammer	9	AMuzza
Asterionella formosa Hassall	2,5,6,11,14,15,20,22,25	AFormo

Table A continued

Bacillaria paradoxa J.F.Gmelin in Linnaeus	3,5,7	BParad
Caloneis amphisbaena (Bory) Cleve	25	CAmph
Cocconeis neodiminuta Krammer	4	CNeodi
Cocconeis placentula Ehrenberg	1,3,4,7,8,15,16,17,19,21,22,23,24,25,28,29,30,31,32	CPlace
Cocconeis placentula var lineata (Ehrenberg) Van Heurck	2,5,18,27	CPlace
Cocconeis placentula var placentula Ehrenberg	5,14	CPlace
Cocconeis scutellum Ehrenberg	24	CScute
Craticula cuspidata (Kützing) Kützing	2	CCuspi
Cyclostephanos dubius (Hustedt) Round in Theriot et al.	3,6,9,17,19,24,25	CDubiu
Cyclostephanos tholiformis Stoermer, Håkansson & Theriot	6,14	CTholi
Cyclotella sp.	3,15,17,20	Cycl
Cyclotella atomus Hustedt	3,10	CAtom
Cyclotella comensis Grunow in van Heurck	15	CComen
Cyclotella distinguenda Hustedt in Gams	19	CDisti
Cyclotella kuetzingiana var planetophora Fricke in A.Schmidt et al.	15	CKuetz
Cyclotella meneghiniana Kützing	3,7,8,9,10,12,13,14,15,16,17,18,19,20,22,23,24,25,27,28,30,31	CMeneg
Cyclotella ocellata Pantocsek	1,2,6,10,11,15,16,19,20,22,24,29	COcell
Cyclotella pseudostelligera Hustedt	1,7,9,12	CPseud
Cyclotella radiosa (Grunow) Lemmermann	7,8,11,15	CRadio
Cyclotella woltereckii Hustedt	13,15	CWolte
Cymatopleura solea (Brébisson) W.Smith	14,24	CSolea
Cymbella sp.	7,14,16	Cymbe
Cymbella afinis Kützing	14,19,21,24,28	CAfinis
Cymbella affiniformis Krammer	11	CAffin
Cymbella alpina Grunow	3	CAlpin
Cymbella cistula (Ehrenberg) O.Kirchner	12,14,16,28	CCistu
Cymbella microcephala Grunow in Van Heurck	31	CMicro
Cymbella neocistula Krammer	22,25,26	CNeoci
Cymbella silesiaca Bleisch in Rabenhorst	3	CSiles
Cymbella subcistula Krammer	13	CSubci
Diadesmis contenta (Grunow) D.G.Mann in Round, Crawford & D.G.Mann	4,5	DConte
Diatoma ehrenbergii Kützing	3,30	DEhren
Diatoma tenius C.Agardh	20,21	DTeniu
Diploneis elliptica (Kützing) Cleve	2	DEllip
Diploneis ovalis (Hilse) Cleve	4,5,8	DOvali
Diploneis smithii var dilata (Grunow) Hustedt	7,15	DSmith

Table A continued

Encyonopsis microcephala (Grunow) Krammer	1,11,12,13,14,15,16	EMicro
Eolimna minima (Grunow) H.Lange-Bertalot in G.Moser et al.	4	EMinim
Eunotia arcus Ehrenberg, C.G.	24	EArcus
Eunotia bilunaris var bilunaris (Ehrenberg) Schaarschmidt in Kanitz	1	EBilun
Eunotia bilunaris var linearis (Okuno) Lange-Bertalot & Nörpel in Krammer & Lange-Bertalot	1	EBilun
Eunotia frickei var. Elongata Hustedt	1,10	EFrick
Eunotia pectinalis (Kützing) Rabenhorst	25	EPecti
Epithemia adnata (Kützing) Brébisson	2,4,7,13,14,15,19,27,28,31	EAdnat
Epithemia sorex Kützing	1,3,4,7,11,14,16,17,19,22,24,25,26,27,28	ESorex
Epithemia turgida (Ehrenberg) Kützing	3,24,25,26	ETurgi
Fallacia pygmaea (Kützing) Stickle & D.G.Mann in Round, R.M.Crawford & D.G.Mann	3,4,5	FPygma
Fallacia tenera (Hustedt) D.G.Mann in Round, R.M.Crawford & D.G.Mann	8	FTener
Fragilaria sp	3,14	Fragil
Fragilaria acus (Kützing) Lange-Bertalot	25	FAcus
Fragilaria biceps Ehrenberg	13	FBicep
Pseudostaurosira brevistriata (Grunow) D.M.Williams & Round	2,5,7,16,17,18,19,21,22,24,27,29,30,31	PBrevi
Fragilaria capucina Desmazières	1,5,9,10,14,16,19,20,24,25,31	FCapuc
Fragilaria construens (Ehrenberg) Grunow	7,16,17,19,30	FConst
Fragilaria construens var. Venter Ehrenberg	9	FConst
Fragilaria delicatissima (W.Smith) Lange-Bertalot	11	FDelic
Fragilaria dilatata (Brébisson) Lange-Bertalot	13,24,25	FDilat
Fragilaria elliptica Van Heurck	12,13	FEllip
Fragilaria intermedia (Grunow) A.Cleve	25	FInter
Pseudostaurosira parasitica (W.Smith) Morales	2,3,4,5,6,7,12,15,17,19,28,29	PParas
Fragilaria pulchella (Ralfs ex Kützing) Lange-Bertalot	25,2	FPulch
Fragilaria rumpens (Kützing) G.W.F.Carlson	20	FRumpe
Pseudostaurosira parasitica var. subconstricta (Grunow) E.Morales in E.Morales & Edlund	4	PParas
Staurosirella pinnata (Ehrenberg) D.M.Williams & Round	1,2,7,9,11,12,13,14,16,17,19,24,25,27,31,32	SPinna
Fragilaria tenera (W.Smith) Lange-Bertalot	11,12,13,20,28,30	FTener
Frustulia rhomboides (Ehrenberg) De Toni	13,14	FRhomb
Gomphonema sp.	2,5	Gomph.
Gomphonema olivaceum (Hornemann) Brébisson	20	GOliva
Gomphonema acuminatum Ehrenberg	4,8,10,11,12,24,25	Gacumi
Gomphonema affine Kützing	4,5	GAffin
Gomphonema angustatum A.Cleve	4,25,26,27,30	GAngus
Gomphonema augur Ehrenberg	25	GAugur

Table A continued

Gomphonema constrictum Hustedt	25,2	GConst
Gomphonema gracile Ehrenberg	22,2	GGraci
Gomphonema intricatum Kützing	10,25	GIntri
Gomphonema parvulum (Kützing) Kützing	1,3,7,11,12,14,16,18,19,24,28,31	GParvu
Gomphonema pumilum (Grunow) E.Reichardt & Lange-Bertalot	15	GPumil
Gomphonema subclavatum (Grunow) Grunow	18	GSubcl
Gomphonema truncatum Ehrenberg	27,28	GTrunc
Gyrosigma acuminatum (Kützing) Rabenhorst	4,11,16,22,24,26,28	GAcumi
Gyrosigma attentuatum (Kützing) Rabenhorst	19	GAtten
Gyrosigma kützingii (Grunow) Cleve and G. peisonis (Grunow) Hustedt	2	GKützi
Gyrosigma nodiferum (Grunow) Reimer	1,7,8	GNodif
Hantzchia amphioxis (Ehrenberg) Grunow in Cleve & Grunow	25	HAmphi
Melosira varians C.Agardh	4	MVaria
Meridon circulare (Greville) C.Agardh	4,5	MCircu
Navicula sp.	2,3	Navic
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski	4,7,10,11,14,17,18,27,30,31	HCapit
Navicula atomus (Kützing) Grunow	31	NAtom
Navicula capitatoradiata H.Germain	11,30	NCapit
Hippodonta costulata (Grunow) Lange-Bertalot, Metzeltin & Witkowski	15	HCostu
Navicula crucicula Frenguelli	1	NCruci
Navicula cuspidata (Kutzing) Kutzing	27	NCuspi
Navicula cryptocephala Kützing	12,27	NCrypt
Navicula cryptotenella Lange-Bertalot	30	NCrypt
Navicula exigua (Gregory) Grunow in van Heurck	1	NExigu
Navicula menisculus Schumann	4,5,7,8	NMenis
Navicula pseudolanceolata Lange-Bertalot	1,7	NPseud
Navicula radiosa Kützing	3,4,5,7,12,13,15,22,24,25,27,28,31,32	NRadio
Navicula rhynchocephala Kützing	22	NRhync
Navicula salinarum Grunow	3,31	NSalin
Navicula seminuloides Hustedt	3,12	NSemin
Navicula subrhynchopcephala Hustedt	11	NSubrh
Navicula tripunctata (O.F.Müller) Bory	5,6	NTripu
Navicula trivialis Lange-Bertalot	22,26	NTrivi
Navicula veneta Kützing	15	NVenet
Navicula ventralis (Krasske)	28	Nventr
Nitzschia acicularis (Kützing) W.Smith	20,21,22,23,24,30,31	NAcicu

Table A continued

Nitzschia amphibia Grunow	21,22,28	NAmphi
Nitzschia angustata (W.Smith) Grunow	18	NAngus
Nitzschia constricta (Gregory) Grunow	18	NConstr
Nitzschia dissipata (Kützing) Rabenhorst	7,10,11	NDissi
Nitzschia elegantulata Grunow	3	NElega
Nitzschia gracilis Hantzsch	12	NGraci
Nitzschia hungarica Grunow	2,19	NHunga
Nitzschia intermedia Hantzsch	25	NInter
Nitzschia lanceolata W.Smith	6	NLance
Nitzschia linearis W.Smith	13,27	NLinea
Nitzschia palea (Kützing) W.Smith	1,3,12,18,21,22,25,26,27,30,32	NPalea
Nitzschia sigma (Kützing) W.Smith	21,22,23,24,25	NSigma
Nitzschia supralitorea Lange-Bertalot	12	NSupra
Nitzschia tryblionella Hantzsch in Rabenhorst	3,24	NTrybl
Nitzschia wuellerstorffii Lange-Bertalot in Lange-Bertalot & Krammer	8	NWuell
Pinnularia brebissonii (Kützing) Rabenhorst	25	PBrebi
Pinnularia borealis Ehrenberg	10	PBorea
Pinnularia gibba Ehrenberg	1,2,7,25	PGibba
Pinnularia intermedia (Lagerstedt) Cleve	14	PInter
Pinnularia major (Kützing) Rabenhorst	13	PMajor
Planothidium lanceolatum (Brébisson ex Kützing) Lange-Bertalot	1,7,8,13,24	PLance
Psammothidium helveticum (Hustedt) Bukhtiyarova & Round	1	PHelvet
Planothidium frequentissimum (Lange-Bertalot) Lange-Bertalot	4,5,6	PFrequ
Rhoicosphenia abbreviata (C.Agardh) Lange-Bertalot	4	RAbbre
Rhopalodia gibba (Ehrenberg) Otto Müller	2,3,7,19,22,24,25,26,27,28	RGibba
Rhopalodia gibberula (Ehrenberg) Otto Müller	26	RGibbe

Table A continued

Sellaphora pupula (Kützing) Mereschkovsky	5,5,7,8,11,12,13,14,16,21,25,27,28,31	SPupul
Staurosira construens Ehrenberg	25	SConst
Stauroneis nobilis Schumann	7	SNobil
Stauroneis phoenicenteron (Nitzsch) Ehrenberg	4,25	SPhoen
Stauroneis sp.	4	Staur
Stephanodiscus hantzschii Grunow in Cleve & Grunow	3,20,28	SHantz
Stephanodiscus medius Håkansson	2	SMediu
Stephanodiscus parvus Stoermer & Håkansson	2,5,6,7,9,13,15,16,17,20,21,22,23,25,28,31	SParvu
Surirella brebissoni Krammer & Lange-Bertalot	5	SBrebi
Surirella brebissonii var kuetzingii Krammer & Lange-Bertalot	1,4	SBrebi
Surirella brightwellii W.Smith	10	SBrigh
Surirella linearis W.Smith	8,19	SLinea
Surirella ovalis Brébisson	8,25	SOvali
Surirella ovata Kützing	25	SOvata
Fragilaria acus (Kützing) Lange-Bertalot	9	FAcus
Fragilaria capucina ssp. rumpens (Kützing) Lange-Bertalot	7,11	FCapuc
Synedra rumpens var. familaris (Kützing) Grunow in van Heurck	2	SRumpe
Ulnaria ulna (Nitzsch) Ehrenberg	3,4,5,15,16,18,19,20,21,22,23,28,31	UUlna
Ulnaria ulna var. Biceps (Kützing) Schönfeldt	23,24,25	UUlna
Tabellaria fenestrata (Lyngbye) Kützing	1,7,10,12	TFenes
Tabularia fasciculata (C.Agardh) D.M.Williams & Round	21	TFasci
Tryblionella hungarica (Grunow) Frenguelli	1,3,4,5,7,13	THunga

VITA

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DEGREES

2011 - 2017	Ph.D., Department of Biological Sciences, METU
2006 - 2009	M.Sc., Department of Biological Sciences, METU
2001 - 2005	B.Sc., Department of Biological Sciences, Ankara University

WORK EXPERIENCES

Year	**Place**	**Enrollment**
2016-2017	*Project:* TÜBİTAK,115Y468	Scholar-Research Assist.
2014	*Project:*FISHTIO2, FP7 CIG 618006	Scholar-Research Assist.
2010-2013	*Project:* TÜBİTAK ÇAYDAG 110Y125.	Scholar-Research Assist.
2009-2014	*Project:*REFRESH, FP7 244121	Scholar-Research Assist.
2007-2009	*Project:*TÜBİTAK ÇAYDAG 104Y308	Scholar-Research Assist.

THESES:

- **M.Sc. Thesis:**Impacts of multistressors on the survival and life history traits of *Daphnia pulex*
- **Ph.D. Thesis:**Tracking the historical changesin Turkish shallow lakes based on a palaeolimnological approach using diatoms

PUBLICATIONS:

- Beklioglu, M., Banu Akkas, S., Elif Ozcan, H., Bezirci, G., & Togan, I. (2010). Effects of 4-nonylphenol, fish predation and food availability on survival and life history traits of Daphnia magna straus. Ecotoxicology, 19(5), 901–910. http://doi.org/10.1007/s10646-010-0470-7

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- Boll, T., Levi, E. E., Bezirci, G., Özuluğ, M., Tavşanoğlu, Ü. N., Çakıroğlu, A. İ., Beklioğlu, M. (2016). Fish assemblage and diversity in lakes of western and central Turkey: role of geo-climatic and other environmental variables. Hydrobiologia, 771(1), 31–44. http://doi.org/10.1007/s10750-015-2608-3

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- Jovanović, B., Bezirci, G., Çağan, A. S., Coppens, J., Levi, E. E., Oluz, Z., Beklioğlu, M. (2016). Food web effects of titanium dioxide nanoparticles in an outdoor freshwater mesocosm experiment. Nanotoxicology, 10(7), 902–912. http://doi.org/10.3109/17435390.2016.1140242

- Levi, E. E., Bezirci, G., Çakıroğlu, A. İ., Turner, S., Bennion, H., Kernan, M., Beklioğlu, M. (2016). Multi-proxy palaeoecological responses to water-level fluctuations in three shallow Turkish lakes. Palaeogeography, Palaeoclimatology, Palaeoecology, 449, 553–566. http://doi.org/10.1016/j.palaeo.2016.02.052

- Tavşanoğlu, Ü. N., Brucet, S., Levi, E. E., Bucak, T., Bezirci, G., Özen, A., Beklioğlu, M. (2015). Size-based diel migration of zooplankton in Mediterranean shallow lakes assessed from in situ experiments with artificial plants. Hydrobiologia, 753(1), 47–59. http://doi.org/10.1007/s10750-015-2192-6

SCHOLARSHIPS

- **2006-2008**The Scientific and Technological Research Council of Turkey (Tübitak) scholarship (**ÇAYDAG 104Y308**)
- **2009-2010** Erasmus scholarship, Deparment of Geography, University College London, London-UK
- **2010-2013**The Scientific and Technological Research Council of Turkey (Tübitak) scholarship (**ÇAYDAG 110Y125**)
- **2015** The Scientific and Technological Research Council of Turkey (Tübitak) scholarship (**2214-A**)

LANGUAGES

- English (Advanced)
- German (Basic)

COMPUTER SKILL

- Microsoft Office Programs
- SPSS Statistics
- Canoco 4.5

HOBBIES

Photography, Scuba Diving, Running, Felt Artist