ROLE OF HYDROLOGY, NUTRIENTS AND FISH PREDATION IN DETERMINING THE ECOLOGY OF A SYSTEM OF SHALLOW LAKES

A THESIS SUBMITTED TO
THE GRADUATE SCHOOL OF NATURAL AND APPLIED SCIENCES
OF
MIDDLE EAST TECHNICAL UNIVERSITY

BY

ARDA ÖZEN

IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR
THE DEGREE OF MASTER OF SCIENCE
IN
THE DEPARTMENT OF BIOLOGY

AUGUST 2006

Approval of the Graduate School of Natural and Applied Sciences

Prof.Dr. Canan Özgen
Director

I certify that this thesis satisfies all the requirements as a thesis for the degree of Master of Science.

Prof. Dr. Semra Kocabıyık
Head of the department

This is to certify that we have read this thesis and that in our opinion it is fully adequate, in scope and quality, as a thesis for the degree of Master of Science.

Assoc. Prof. Dr. Meryem Beklioğlu Yerli
Supervisor

Examining Committee Members

Prof. Dr. Temel Oğuz (METU, IMS)

Assoc.Prof. Dr. Meryem Beklioğlu Yerli (METU, BIO)

Assoc.Prof. Dr. Sertaç Önde (METU, BIO)

Assist.Prof.Dr. Ayşegül Çetin Gözen (METU, BIO)

Assist.Prof.Dr. Yeşim Büyükateş (COMU, FOF)

I hereby declare that all information in this document has been obtained and presented in accordance with academic rules and ethical conduct. I also declare that, as required by these rules and conduct, I have fully cited and referenced all material and results that are not original to this work.

Name, Last name : Arda Özen

Signature :

ABSTRACT

ROLE OF HYDROLOGY, NUTRIENTS AND FISH PREDATION IN DETERMINING THE ECOLOGY OF A SYSTEM OF SHALLOW LAKES

Özen, Arda

M.S., Department of Biology

Supervisor: Assoc. Prof. Dr. Meryem Beklioğlu Yerli

August 2006, 119 pages

In this study, the hydrology and physical, chemical and biological variables of a shallow lake system including the Lakes Mogan and Eymir between 1997-2005 were evaluated.

In Lake Eymir, a biomanipulation study was conducted between August, 1998 – December, 1999. Upon biomanipulation, Lake Eymir shifted to clearwater state with submerged vegetation domination during 2000-2003. However, in 2004, the lake shifted back to algae-dominated turbid state since the buffer mechanisms provided by submerged plants were absent. In the summer of 2005, fish kills were observed due to algal bloom. However, due to increasing hydraulic residence time in the lake, internal processes became more important for nutrients.

Lake Mogan faces seasonal and interannual water level fluctuations. During the low water levels experienced in 2001 and 2005, which coincided with the high hydraulic residence times, the in-lake phosphorus amount was controlled by internal processes rather than external loading. Moreover, results revealed that hydrology and submerged plants were important in the ecology of Lake Mogan.

Furthermore, the relationship between the phytoplankton, zooplankton and the environment in Lakes Eymir and Mogan, which was predicted via Canonical Corresponding Analysis, revealed that nutrients and water transparency were both important for plankton communities. Both the top-down and bottom up effects were valid in Lake Eymir, while only the bottom-up effect and submerged plants were important for Lake Mogan.

Finally, the present study provided a good example for the submerged plant dominated clearwater state triggered by biomanipulation, and the impact of hydrology on the ecology of shallow lakes.

Keywords: Biomanipulation,alternative stable states, zooplankton, phytoplankton,canonical correspondence analysis (CCA).

ÖZ

AYNI HAVZADAKİ SIĞ GÖLLERİN EKOLOJİK YAPISININ BELİRLENMESİNDE SU SEVİYESİ DEĞİŞİMİ, BESİN TUZLARININ YOĞUNLUĞU, VE BALIK STOKUNUN DEĞİŞEN ROLLERİ

Özen, Arda
Yüksek Lisans, Biyoloji Bölümü
Tez Yöneticisi: Doç. Dr. Meryem Beklioğlu Yerli

Ağustos 2006, 119 sayfa

Bu çalışmada Eymir ve Mogan göllerine ait 1997-2005 yılları arası hidrolojik, fiziksel, kimyasal ve biyolojik veriler değerlendirildi.

Eymir gölünde biyomanipulasyon çalışması Ağustos 1998 - Aralık 1999 tarihleri arasında gerçekleştirildi. Eymir gölü biyomanipulasyondan sonra 2000 – 2003 yılları arasında suiçi bitkilerin baskın olduğu berrak su durumuna geçti. Suiçi bitkilerin sağladığı tampon mekanizmaların kaybolmasına bağlı olarak 2004 yılında göl tekrar alglerin baskın olduğu bulanık su durumuna geçti. 2005 yılında alg patlamalarına bağlı olarak balık ölümleri gerçekleşti. Gölde artan hidrolik kalma

zamanına bağlı olarak içsel mekanizmalar besin tuzları için daha önemli hale gelmiştir.

Mogan gölü mevsimsel ve yıllar arası su seviyesi dalgalanmalarına mahruz kaldı. Su seviyesinin düşük olduğu ve hidrolik kalma zamanının yüksek olduğu 2001 ve 2005 yıllarında göl içi fosfor seviyesi dışsal yüklemeden çok içsel mekanizmalarla kontrol edildi. Ayrıca Mogan gölünün ekolojik yapısının belirlenmesinde hidrolojinin ve suiçi bitkilerin de önemli olduğu anlaşıldı.

Eymir ve Mogan göllerindeki fitoplankton, zooplankton ve çevresel faktörler arasındaki ilişki kurallı karşılama çözümlemesi kullanılarak belirlendi ve besin tuzlarının ve suiçi ışık geçirgenliğinin plankton komüniteleri için önemli olduğunu gösterdi. Eymir gölünde yukardan - aşağı ve aşağıdan - yukarı etkiler geçerliyken Mogan gölünde sadece aşağıdan yukarı mekanizmaların ve suiçi bitkilerin önemli olduğu belirlendi.

Son olarak mevcut çalışma suiçi bitkilerin önemini ortaya koyması, biyomanipulasyondan sonra bulanık su durumuna geçiş mekanizmalarını açıklaması ve hidrolojinin sığ göller üzerindeki etkisini ortaya koyması bakımından iyi bir örnek teşkil etmektedir.

Anahtar kelimeler: biyomanipulasyon, alternatif kararlı durumlar, zooplankton, fitoplankton, kurallı karşılama çözümlemesi.

To My Family

ACKNOWLEDGEMENTS

I would like to express my sincere gratitude to Assoc. Prof. Dr. Meryem Beklioğlu Yerli for her great support, supervision and understanding throughout in this study.

I would also express my appreciation to Özge Karabulut Doğan for her help during the preperation of plant maps of lakes with GIS and İsmail Küçük for providing the hydrological data of the lakes. I would like to thank to the fishermen and Lake's Warden Team for their precious help in the field. I am very grateful to Metehan Demirelli for his help during the fish stock estimations. Special thanks to Burcu Karapınar, Özge Karabulut, Ceran Şekeryapan, Gizem Bezirci, Korhan Özkan and Onur Kerimoğlu for their help in the field and in the laboratory.

Finally, I would like to thank to my family and my housemate Önder Metin for their patience and support to complete this thesis.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

CHAPTER 1

ROLE OF NUTRIENTS, HYDROLOGY, AQUATIC VEGATATION AND FISH PREDATION IN DETERMINING THE ECOLOGY OF LAKE EYMIR AND LAKE MOGAN

1. INTRODUCTION

For most of its history, limnology has focused on deep lakes that stratify in summer. This stratification largely isolates the upper water layers (epilimnion) from the colder, deep water (hypolimnion) and from interaction with the sediment during the summer. The impact of macrophytes on the community is relatively small in such lakes, as plant growth is restricted to a relatively narrow marginal zone. In contrast, shallow lakes can be largely colonized by macrophytes and do not stratify for long periods in summer. Obviously, the intense sediment-water interaction and the potentially large impact of aquatic vegetation makes the functioning of shallow lakes different from that of their deep counterparts in many aspects.

In several parts of the world shallow lakes are more abundant than deep ones. Shallow lakes constitute the bulk of the world's freshwater area (Wetzel, 1990; Moss, 1998); they have great importance in providing productive fisheries in Asia, Africa and South America (Dugan, 1994); they are of great conservation value, often for the migratory bird communities they support in the temperate world. In densely populated areas even small lakes can be very important from a recreational point of view. Fishing, swimming, boating and bird watching attract a large public. During the couple of past decades, eutrophication (the over enrichment of aquatic ecosystems with nutrients leading to algal blooms and anoxic events) is a persistent condition of surface waters and become a widespread environmental problem because of increases in human population, demand for food, land conversion, fertilizer use, and nitrogen deposition (Carpenter et al., 2005).

The pristine state of the most shallow lakes is probably one of clear-water and a rich aquatic vegetation. Any disturbance in the pristine state of an ecosystem might cause serious disruptions in ecosystem structure and functioning. However, shallow lakes do not respond to any disturbance, which may make the system depart from pristine state, in a linear and immediate fashion. A possible explanation for this delay in response was first suggested by Scheffer (1989), as alternative stable states theory.

1.1 Alternative Stable States

Ecologists are gathering increasing empirical support for the idea, first proposed in the 1960s (Lewontin 1969), that communities can be found in one of several possible alternative stable states (ASS) (Holling 1973; May 1977; Knowlton 1992; Scheffer et al. 1993; Scheffer et al. 2001; Dent et al. 2002). It has evoked a lot of attention by both theoreticians and empiricists and stimulated numerous research activities. In a nutshell, ASS predicts that ecological systems may exist potentially indefinitely in contrasting states under the same external environmental conditions. The concept of ASS and its important implications for basic and applied ecology are nowadays widely recognised among ecologists.

Shallow lakes are one of the most challenging lake types to manage for water quality. Most shallow can exist in either of two conditions; the turbid water state or the clear water state. When these lakes exist in the turbid water state they are characterized by very turbid water, absence of submerged plant, limited emergent vegetation, a sparse fishery dominated by planktivorous and benthic fishes, algal blooms, high turbidity. When these lakes exist in clearwater state they are characterized by clear water, submerged vegetation, and strong communities of fish and invertebrates (Scheffer et al., 1993).

The existence of stabilizing mechanisms that tend to keep the system in either a vegetation- or phytoplankton-dominated state suggests the potential for alternative stable states. However, in mathematical models alternative equilibria usually occur for limited ranges of parameter settings only, and likewise, real systems will normally have these properties only for a limited set of conditions. Indeed, the hypothesized stabilization of the vegetated state seems unlikely in deep lakes where the narrow littoral zone that can be vegetated has a less dramatic impact on turbidity than in shallow lakes that can be entirely vegetated. Also in shallow lakes the existence of alternative stable states will be limited to an intermediate range of nutrient levels, as oligotrophic lakes are rarely turbid and very high nutrient loading usually excludes vegetation dominance. Therefore, the demonstration of stabilizing mechanisms per se is not sufficient to conclude that a lake has alternative stable states. (Scheffer, 1993).

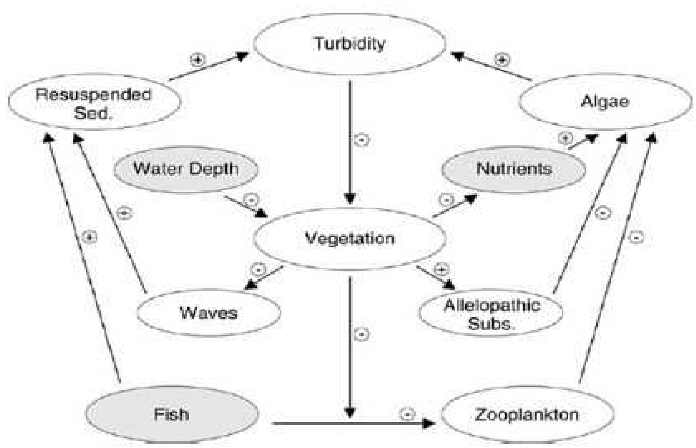

Figure 1.1. Feedbacks that may cause a vegetation dominated state and a turbid state to be alternative equilibria. The qualitative effect of each route in the diagram can be computed by multiplying the signs along the way. This shows that both the vegetated and the turbid state are self-reinforcing. (Taken from Scheffer, 1993)

A simple graphical model suffices to clarify roughly how nutrient loading and water level may affect the stability properties of the ecosystem. The model is based on three assumptions:

1. Turbidity increases with increasing nutrient levels.
2. Vegetation reduces turbidity.
3. Vegetation disappears entirely when a critical turbidity is exceeded.

In view of the first two assumptions, equilibrium turbidity can be drawn as two different functions of the nutrient level (Figure.1.1): one for a macrophyte dominated, and one for an unvegetated situation. Above a critical turbidity, macrophytes will be absent, in which case the upper equilibrium line is the relevant one; below this turbidity the lower equilibrium curve applies. Over a range of intermediate nutrient levels two alternative equilibria exist: one with macrophytes, and a more turbid one without vegetation. At lower nutrient levels, only the macrophyte-dominated equilibrium exists, whereas at the highest nutrient levels, there is only a vegetationless equilibrium (Figure.1.2). The course of the eutrophication process can be derived from this picture. Gradual enrichment, starting from low nutrient levels will cause the system to proceed along the lower equilibrium curve until the critical turbidity is reached at which macrophytes disappear. A jump to a more turbid equilibrium at the upper part of the curve occurs. In order to restore the macrophytedominated state by means of nutrient management, the nutrient level must be lowered to a value where algal growth is limited enough by nutrients alone to reach the critical turbidity for macrophytes again. At the extremes of the range of nutrient levels over which alternative stable states exist, either of the equilibrium lines approaches the critical turbidity that represents the breakpoint of the system. This corresponds to a decrease of stability. Near the edges, a small perturbation is enough to bring the system over the critical line and to cause a switch to the other equilibrium. Efforts to reduce nutrient level will change the stability landscape again. Despite the reappearence of alternative clear equilibrium, tje locally turbid state tends to be sustained. Only a drastic reduction of nutrient level will be sufficient to switch to the clear state (Figure 1.3).

Water level in the lake is another important control variable with respect to aquatic macrophyte dominance. Since vegetation can resist a higher turbidity if the lake is shallower, the horizontal breakpoint-line in the diagram will be at a higher critical turbidity in shallower lakes. It can be seen from the graphical model that a small shift in critical turbidity resulting from a change in water level can bring about a switch from one state to the other state that is independent of nutrient enrichment and top-down effects (Wallsten & Forsgren, 1989; Blindow, 1992; Beklioglu et al., 2004, 2006; Tan & Beklioglu, 2006).

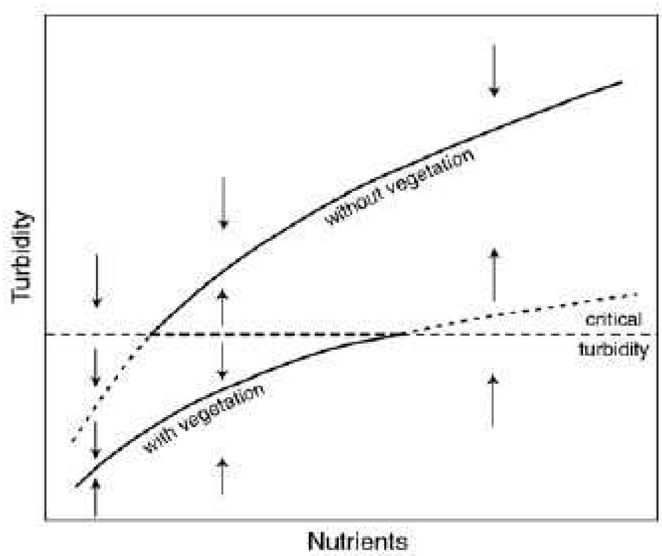

Figure 1.2. Alternative equilibrium turbidities caused by disappearance of submerged vegetation when a critical turbidity is exceeded. The arrows indicate the direction of change when the system is not in one of the two alternative stable states. (Taken from Scheffer, 1998)

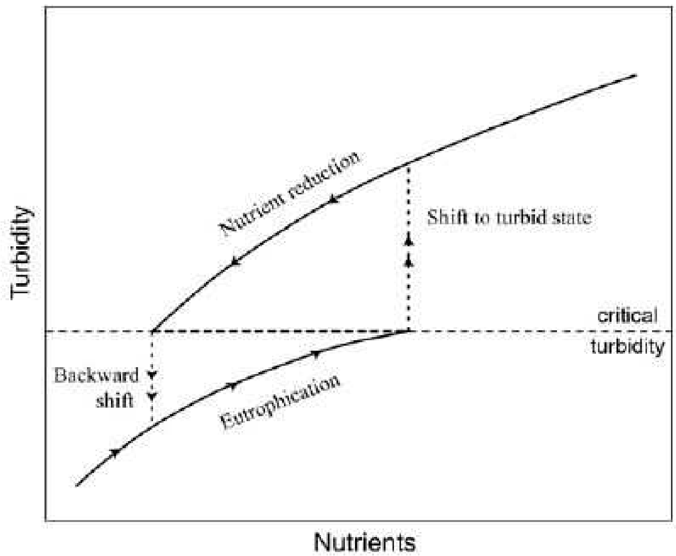

Figure 1.3. As a consequence of the alternative stable states the lake shows hysteresis in response to changes in nutrient loading.Once the lake has shifted to the turbid state, a reduction of nutrients leads to a relatively moderate decrease in turbidity along the upper branch of the equilibrium graph until nutrient levels have dropped enough to go through a backward switch. (Taken from Scheffer, 1998)

There are different mechanisms for maintaining clear and turbid water state. Figure 1.4 summarizes the main mechanisms involved. A simple way of evaluating the overall effect of the depicted interactions is to multiply the signs along the way of a path through the scheme. This exercise shows that through all depicted routes turbidity enhances turbidity, and vegetation enhances vegetation.

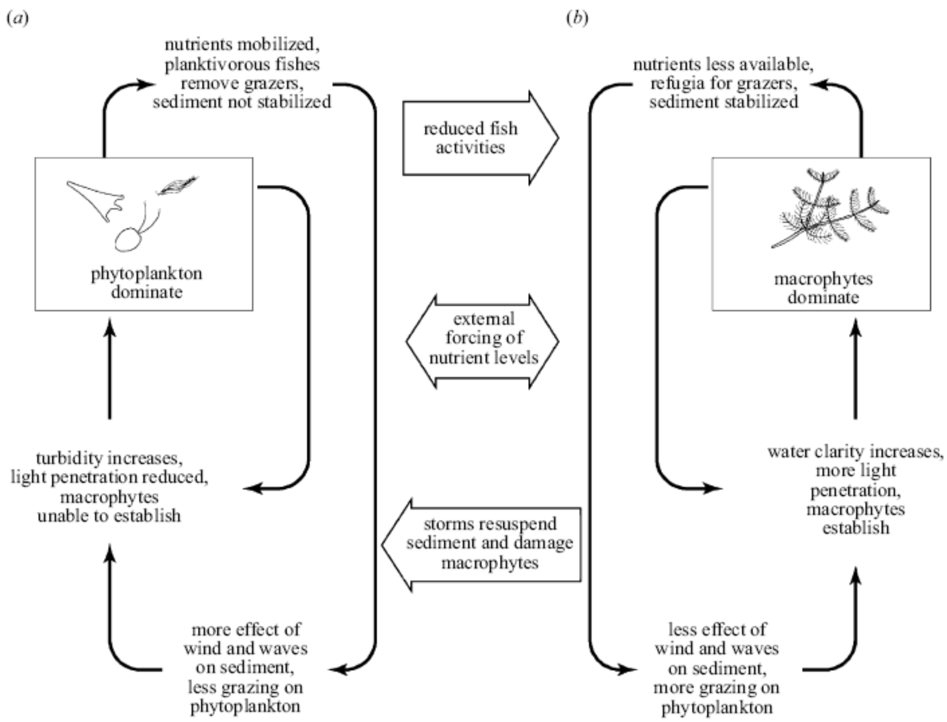

Figure 1.4. Conceptual model for multiple states of water clarity in shallow lakes. **(*a*)** Feedbacks associated with the turbid state, and **(*b*)** feedbacks associated with the clear-water state. Processes that can move the system from one state to the other are shown between **(*a*)** and **(*b*).** (Taken from Dent et al., 2002).

1.2. Eutrophication

During the couple of past decades, increased urbanization with sewage disposal and intensive farming practices have increased the nutrient loading to many lakes worldwide (Gulati & Van Donk, 2002; Jeppesen et al., 1999, Jeppesen et al., 2003a; Sagrario et al., 2005). This has lead to water quality deterioration, namely eutrophication of lakes or cultural eutrophication (Moss, 1998). Because of intensive exchange of nutrients between water column and sediments, shallow lakes are sensitive to eutrophication.

Eutrophic shallow lakes are characterized by algal blooms, high turbidity, absence of submerged plant, decreased ratio of piscivorous to planktivorous fish stock (Scheffer et al., 1993; Beklioglu et al., 2000; Jeppesen et al., 2003b).

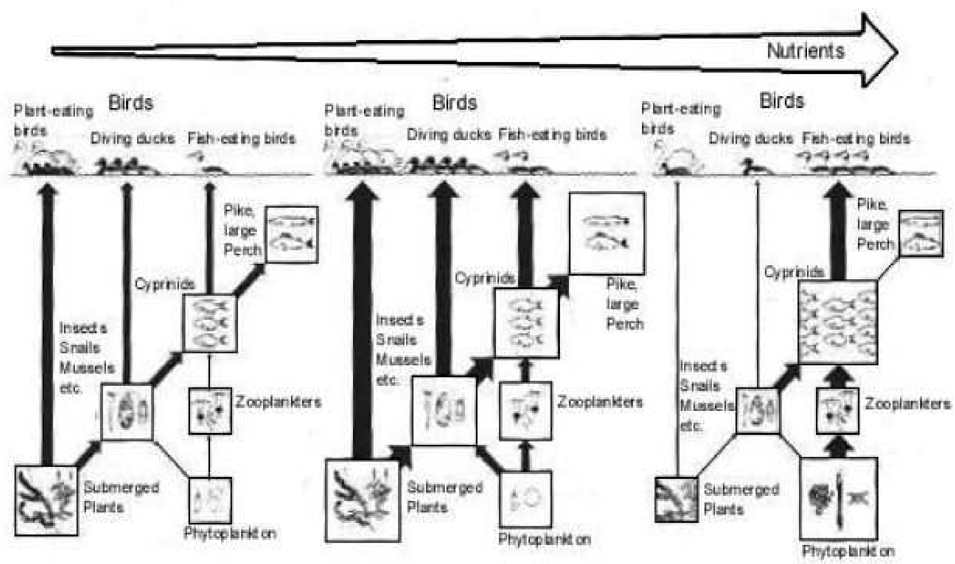

Figure 1.5. The change in biological structure and processes with nutrient enrichment. The width of the arrows indicate the strength of the relation between compartments. (Taken from Jeppesen, 1998)

1.3 Restoration of eutrophicated shallow lakes

1.3.1 Restoration techniques

Much of effort has been made to reduce the external loading of phosphorous and to restore lake water quality. Some lakes respond rapidly to such loading reductions with improvement of water clarity with decrease in the abundance of planktivorous fish and increase in the share of piscivores, and decrease in concentrations of TP and chlorophyll-a (Jeppesen et al., 2003a), while others are highly resistant due to internal loading or biological homeostasis (Søndergaard et al., 1999; Jeppesen et al., 2002).

Reduction of external nutrient loading may have little effect, as during the period of eutrophication a large amount of phosphorus has often been adsorbed by the sediment. When the loading is reduced and its concentration in the water drops, phosphorus release from the sediment becomes an important nutrient source for phytoplankton. Thus a reduction of the external loading is often compensated by 'internal loading', delaying the response of the lake water concentration to the reduction of external loading.

However, nitrogen may play a far more important role than previously appreciated in the loss of submerged macrophytes at increased nutrient loading and would cause a delay in the reestablishment of the nutrient loading reduction as it may vary with fish density and climatic conditions (Sagrario et al., 2005). Nutrient reduction alone may have little impact on water clarity (Van der Molen & Portielje, 1999), but an ecosystem disturbance like fish manipulation namely biomanipulation can bring the lake back to a clearwater state (Meijer et al., 1994; Perrow et al., 1997; Meijer et al., 1999; Mehner et al., 2002).

The average success rate of food-web manipulations is about 60% (Hansson et al., 1998; Drenner & Hambright, 1999). General criteria for a successful biomanipulation mainly would include decrease in algal turbidity, concentrations total phosphorous and chlorophyll-a and plankti-benthivorous fish abundance, increase in Secchi depth transparency and macrophyte cover. Biomanipulation may produce positive results early in the study and can be categorized as successfull, given to at least 4 to 5 year time span after the intervention (Hannson et al., 1998; Meijer et al., 1999). In many cases, lakes returned to turbid state with a few years after biomanipulation. There are several factors, which may act as a forward switches to trigger an turbid water state in lakes: nutrient increases, mechanical damage, vertebrate grazing (birds, fish), piscivore kills, rising salinity, use of herbicides and pesticides and heavy metal toxicity to Daphnia (Gulati &Van Donk, 2002) and reestablishment of dense populations of planktivorous or benthivorous fish (Kasprzak et al., 2002).

1.3.2. Lake restoration differences between Temperate, Tropical/Subtropical and Mediterranean Lakes

Experience with lake restoration in subtropical/tropical lakes and Mediterranean lakes are far less than with temperate lakes (Sacsso et al., 2001; Beklioğlu et al., 2003). Nutrient loading reduction may lead to improvement of the ecological state with declining algal biomass and increased lake water transparency (Lowe et al., 2001). As northern temperate lakes, Mediterranean and subtropical lakes have been observed to respond to decreased phosphorus loading, nutrient availability declined, but only a limited impact on the overall food web was recorded (Coveney et al., 2005; Romo et al., 2005).

Recently Jeppesen et al., 2005 suggested that fish stock manipulation would not have the same positive effect on the environmental state in tropical and subtropical lakes as in temperate lakes because of several factors. First, the richness in fishness species is often higher in tropical and subtropical lakes. (Sunaga and Verani, 1997). Second, the fish stock in tropical and subtropical lakes is often dominated by omnivorous fish and independent of trophic state (Branco et al., 1997). The fish stock have the potential of feeding on zooplankton and increase the the potential control of the zooplankton. According to Blanco et al., (2003), the omnivorous structure of fish communities in Mediterranean lakes resembles that described for tropical lakes. Few piscivorous species are present and omnivorous fish generally dominate independently of trophic state. Thirdly, compared the North American and European freshwater fish communities , only few large piscivores and small sized carnivores occur and sit and wait predators are often more common (Quiros, 1998). Therefore, top-down control by piscivores is most likely weaker in tropical and subtropical lakes than in temperate lakes. Fourthly, recent studies show that fish density is substantially higher in tropical and subtropical lakes in South America than in North temperate lakes (Meerhorff et al., 2003).

Fish reproduction occurs throughout the year in many tropical and subtropical lakes although fish reproduction took place once a year in temperate lakes.

High abundance of small fishes leads to higher predation pressure on zooplankton than in temperate lakes, where the effect of juvenile fish is typically strong mainly in mid-late summer (Jeppesen et al., 1997).

Since there is high predation by fish, the zooplankton communities in tropical and subtropical lakes are often dominated by small cladocerans and rotifers and by juveniles and small copepodites (Garcia et al., 2002). The classic control of phytoplankton by large zooplankton in temperate lakes is therefore not usually found in tropical lakes. In conclusion, biomanipulation may be used as a supplementary tool in order to speed up the recovery process in tropical and subtropical lakes (Lazorro, 1997).

In several studies, King et al., 1997 and Angeler et al., 2002, the role of cyprinids (the dominant and often non-planktivorous species in Mediterranean ecosystems) can importantly affect interactions between sediments and water column, including food web dynamics. Benthic feeding of most adult cyprinids truncates the pelagic trophic cascade. In Mediterranean lakes, zooplankton community seems mostly dominated by small-sized species (Beklioglu et al., 2003; Romo et al., 2004; Fernández-Aláez et al., 2004; Romo et al., 2005). Dominance of omnivorous and benthivorus fish species, whose diversity is usually high, together with frequent spawning and absence of piscivores seem responsible for the lack of large-bodied zooplankton grazers (Blanco et al., 2003; Fernández-Aláez et al., 2004; Romo et al., 2005).

As shown in Romo et al., 2005 large zooplankton (mainly Cladocera) can only temporarily influence water clarity and phytoplankton climatic zone, restoration of Lake Eymir by biomanipulation is a first case study and community structure and biomass after nutrient reduction. Also from a dry showed that biomanipulation was a very effective measure in arid climates as it observed temperate lakes (Beklioglu et al., 2003).

1.4. Role of Hydrology Water Level Fluctuations in shallow lakes varying along the geographic gradient

Hydrology is a crucial factor for nutrients, growth rates, timing of reproduction, balance of photosynthesis and respiration, rates of mineralization especially for shallow lakes. Previous studies mostly focused on external nutrient loading that depends on water input. The relationship between in-lake nutrient concentration and hydraulic residence time has shown already by Vollenweider, (1975). However, the significance of hydrology on internal nutrient dynamics of shallow lakes has been realized recently. Drought-induced decrease in water level and increase in water residence time may provide longer contact with sediment that may enhance the internal release of nutrients, such as phosphorous (Romo et al., 2005; Karapınar & Beklioglu, 2006).

Water level fluctuations (WLFs) emerge as the decisive element of hydrology especially in shallow lakes, which are particularly sensitive to any rapid change in water level and input. Therefore, WLFs may have an overriding effect on the ecology, functioning and management of shallow lakes. Water levels in shallow lakes naturally fluctuate intra- and interannually depending largely on regional climatic conditions. Through global climate change, water level fluctuations may become as significant as nutrients on functioning of shallow lakes (Coops et al., 2003).

Water level fluctuations may be a catastrophic disturbance that may cause shifts between the turbid and the clear, macrophyte-dominated state that is independent of nutrient enrichment and top-down effects (Wallsten & Forsgren, 1989; Blindow, 1992; Beklioglu et al., 2004; 2006). Water level seems to be a major factor influencing summer thermal stratification, nutrient dynamics and submerged plant development. Some studies suggested that WLFs may be a catastrophic disturbance for submerged plant communities since excessively high water level in growing season may suppress plant development and, in turn, such a lake may shift to a state associated with low vegetation development (Blindow 1992 and Blindow et al. 1997). Conversely, too low water level in winter and in summer may damage the expansion of plants in the littoral zone through ice and wave action in winter and desiccation in summer (Blindow, 1992; Blindow et al., 1997; Beklioglu et al., 2001, 2006). Decrease in summer water level results in lack of thermal stratification. This, in turn, enhances phytoplankton growth by continuous supply of nutrients through increased internal loading (Naselli-Flores, 2003).

Moreover, shallow lakes are particularly sensitive to water stock any rapid changes in water level and the water input flux related to local climatic patterns. In arid tropical shallow lakes, sensitivity to hydrological conditions also has strong consequences on in-lake concentrations of nutrients and major ions, especially in determining the salinity (Talling, 2001).

Furthermore, when nutrient control along with food web management is not achievable or not feasible to improve submerged plant development, other methods such as hydrological adjustments like water level manipulation in shallow lakes may offer a large potential to induce recovery.

Tropical lakes are particularly sensitive to hydrological changes, which strongly affect their structure and functioning. In low water, season higher densities and richness of zooplankton and fish were found in Bolivian lakes (Rejas et al., 2000) and macrophyte dominance in Lake Alalay in water level increase.

Water level fluctuations in shallow lakes of arid or semi-arid regions have strong impact on vegetated state (Gafny & Gasith, 1999, Beklioglu et al., 2001, Zalidis et al., 2002; Beklioglu et al., 2004; 2006, Havens et al., 2004; Van Geest et al., 2005, Tan & Beklioglu, 2006). These lakes are subject to cold winters, leading to a mixed fish community and with low species richness compared to subtropical and tropical lakes, and dominance of northern species (Beklioglu et al., 2003) and state between temperate and tropical/subtropical lakes.

In Mediterranean areas, groundwater hydrology is as important as surface hydrology for the persistence and functioning of aquatic systems. Water availability depends a great deal on groundwater discharge, which is either lagged from rainfall seasonality or lack any seasonality because it depends in turn upon the hydrogeolagic features of subterranean aquifers. In addition to water availability, groundwater can be a source for nutrients (Alvarez et al., 2004).

1.5. Role of aquatic vegetation in shallow lakes varying along the geographic gradient

Submerged macrophytes play a crucial role for maintaining clearwater state in shallow lakes since they provide useful stabilizing mechanisms. They cause reduction of sediment resuspension (Boström et al., 1982), nutrient limitation of algae through nitrogen uptake and enhancement of denitrification (Ozimek et al., 1990; van Donk et al., 1993), release of allelopathic substances by plants that inhibit the growth of algae (Wium-Andersen, 1987), refuge for zooplankton against fish predation (Timms & Moss, 1984) and spawning grounds and refuge against cannibalism of piscivorous fish (Grimm, 1989). These mechanisms may contribute to the effect of aquatic macrophytes on water clarity, although the relative importance of each of these factors varies considerably from case to case.

In tropical and subtropical lakes, the effects of macrophytes on trophic interactions are more complex, as all life forms (emergent, submerged and floating-leaved species) can extremely prominent. Many fish species exhibit spatial distribution patterns often connected with zooplankton and predator densities, and also with different macrophyte life forms (Meerhoff et al., 2003). One might therefore expect that the vegetation is poor refuge for large bodied zooplankton in warm lakes.

While in temperate lakes, the introduction and development of aquatic plants are considered as key step in a restoration process (Moss et al., 1996) many aquatic plants in tropical and subtropical lakes are often considered as a nuisance and subject to severe eradication measures. In temperate lakes, submerged plants acts as a daytime refuge for zooplankton against fish predation (Timms & Moss 1984, Burks et al., 2002). At night large zooplankton come to the open water to feed and therefore they contribute to clearwater state in lakes with high macrophyte coverage (Jeppesen et al., 1997). In contrast, macrophytes are extremely important to fish in the tropics and subtropics (Meerhoff et al., 2003).

Many Mediterranean lakes are shallow, thus increasing the importance of emergent and submerged plants and the pelagic and benthic interactions for overall limnological functioning (Moss et al., 2004). For shallow Mediterranean lakes, submerged macrophytes may be even more effective in maintaining water clarity, since they potentially persist during all year round and they have advantages in the competition for nutrients and light between plants and algae. Processes related to vegetation and the pelagic benthic couplings regulate ecosystem processes in Mediterranean lakes and these phenomena are only beginning to be explored in Mediterranean aquatic ecology (Alvarez et al., 2005).

1.6 Scope of the study

One of the scopes of this study was to investigate the effects of water level fluctuations on the physical, chemical and biological variables, especially on submerged plant development in shallow Lakes Eymir and Mogan which are located in arid Anatolia.

In Lake Eymir, sewage effluent diversion was undertaken in 1995 and biomanipulation took place between August 1998 and December 1999. From a dry climatic zone, restoration of Lake Eymir by biomanipulation is a first case study and it proved that clear water condition can be initiated by top-down mechanisms. This study aims to describe the processes responsible for the shift back to turbid water state that took place in 2004 in Lake Eymir and to develop tools that managers can use to shift and maintain shallow lakes in a clearwater state. The processes, which were responsible for the shift, were evaluated in relation with similarities and differences from temperate zone biomanipulation cases.

Finally, the possible effects of climate change on Turkish shallow lakes will be discussed by using long period lake data set for Lake Eymir and Lake Mogan.

2. STUDY SITES

Lake Eymir and Lake Mogan are located 20 km south of Ankara, within the Gölbasi Municipality. Lakes Eymir and Mogan are alluvial dam lakes formed by damming of Imrahor River. The basin was formed by tectonic depression and has been named as Gölbasi formation (Görgün, 1994).

The region has the Central Anatolian climatic conditions (semi-arid), with most of the rain falling during late winter and spring. The summers are hot with an average temperature of 22.0±0.7ºC and relatively dry with an average precipitation of 58.7±41.9 mm. Both lakes are important recreational sites of Ankara. Besides Gölbasi Municipality, there are TEAS settlement, ten villages and Police Academy in the catchment area of both lakes, and they are affected by agricultural practices, recreation, as well as small-scale industries in the catchment.

2.1 Lake Mogan

Lake Mogan (39°47’ N 32°47’ E) is a shallow lake with of a mean depth 2.1 m and a maximum depth of 3.5 m. It is a large lake with a surface area of 5.4-6 km^2 and a total of 925 km^2 drainage area situated 20 km south of Ankara. The lake is mainly fed by four main inflows, the Sukesen brook in the north, the Gölcük and Yavrucak brooks in the west and the Çölovasi brook in the east. These brooks first run through agricultural lands and then through reed beds before reaching the lake. The outflow of the lake empties into downstream Lake Eymir through a canal and a wetland in the north.

Gölbasi town located north of the lake has a population of 62.602 people (census data of 2000, State Institute of Statistics). Sukesen brook, which runs through the town before reaching the lake, used to receive sewage effluent discharge until 1999. Since then, the effluent has been connected to a collector.

However, the west catchment of the lake was recently opened to settlement and the sewage effluent of the houses was discharged into Gölcük and Yavrucak brooks, which were connected to the collector later.

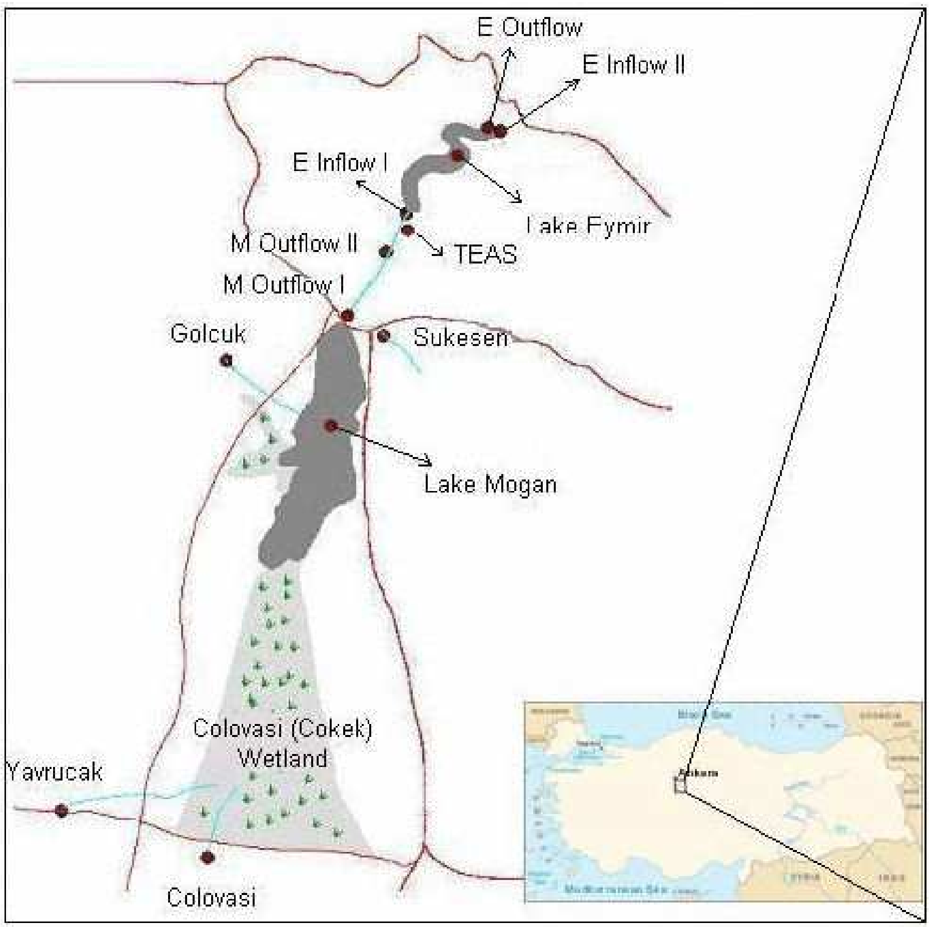

Figure 2.1 Lake Eymir and Lake Mogan

The earliest limnological survey, which was carried out during 1971 1972, recorded a turbid water state (Secchi disc transparency: 39 ± 7 cm) with limited distribution of submerged plants confined to the very shallow areas, as a belt of 8-10 m width on the shores (Tanyolaç & Karabatak, 1974). The subsequent survey, which was carried out after the implementation of new water regime using the sluice gate, recorded an improvement in the water clarity (Secchi depth of 90 ± 6 cm), and a slight increase in the vegetation coverage (Obali, 1978). Furthermore, a submerged plant dominated clearwater state was recorded, with a very high vegetation cover in 1991 (80%) and a high Secchi disc transparency (168 ± 56 cm) (DSI, 1993).

Burnak & Beklioglu (2000) reported that the lake was in macrophyte dominated clearwater state at moderate TP (63 ± 5 µg l^{-1}), low DIN (166 ± 58 µg l^{-1}) and chlorophyll-a (8.8 ± 2 µg l^{-1}), very high Secchi depth transparency and 70-80% macrophyte coverage. The lake experienced regular intra- and interannual water level fluctuations. Tan (2002) found that an increase in the spring water level compared to the previous years resulted in disappearance of macrophytes in 2000, regardless of the low nutrient concentrations. Moreover, in the consecutive year a decrease in the spring water level led to re-colonization of the submerged macrophytes to 90% of the lake surface area (Tan, 2002). In 2002, the annual and spring water levels increased again and the submerged macrophyte coverage decreased to 19% (Kisambira, 2003). During these state shifts in the lake, deteriorations in the chlorophyll-a concentrations and Secchi depth transparency were observed. However, it was concluded that these deteriotions were a consequence of change in the submerged plant coverage induced by water level rise. This state shift took place regardless of a significant change in the concentrations of nutrients, suspended solids and chlorophyll-a and appeared to have been dependent on the water level. Morever, it was concluded by field observations that the chlorophyll-a concentrations in Lake Mogan was more important in maintaining water clarity (Tan & Beklioğlu, 2005).

A study carried out on Lake Mogan to investigate role of waterfowl and fish on growth of submerged macrophytes revealed their effects were low in Lake Mogan with abundant submerged vegetation (Sandsten et al., 2005).

Lake Mogan and the surrounding wetlands have been famous for the high density of waterfowl and mostly host >20,000 waterfowl in winter (Kiraç et al., 1995; Özesmi, 1999).In 1990 Lake Mogan was given the status of 'Specially Protected Area' by the Ministry of Environment. Furthermore the lake also obtained the status of the 'International Bird Area' and internationally important wetland due to the rich and diverse community of waterfowl.

2.2 Lake Eymir

Lake Eymir is located 20 km south of Ankara (area: 1.25 km2, Zmax: 5.5 m, Zmean: 3.1 m, catchment: 971 km^2 and shoreline: 13 km).The upstream Lake Mogan empties into Lake Eymir at the southwest corner, forming the main inflow of Lake Eymir, named Inflow I. Inflow II is located at the northern end, which usually dries up in summer.

The lake was originally in a clearwater state with summer Secchi disc transparency being >4 m and a submerged plant community mainly dominated by Charophytes whose outer depth of colonization was 6-7 m (Geldiay, 1949). The lake was eutrophicated onward late 1970s as a consequence of discharge of untreated sewage effluent from nearly 25.000 inhabitants of a nearby town into the main inflow of the lake, Inflow I. The prediversion concentrations of total phosphorous (TP), dissolved inorganic nitrogen (DIN), chlorophyll-a, and suspended solids were very high and Secchi depth transparency was low (727 ± 43 µg l^{-1}, 1.49 ± 0.82 mg l^{-1}, 27 ± 7 µg l-1, 38 ± 18 mg l^{-1} and 56 ± 6 cm, respectively), (Altinbilek et al., 1995).

A succesful effluent diversion was undertaken in 1995, which led to 88% and 95% reduction in the areal loadings of TP and DIN, respectively. Even though the diversion reduced the in-lake TP level, the poor water clarity and low submerged plant cover persisted (Beklioglu et al., 2003). Dominance of the fish stock by planktivorous tench (*Tinca tinca L.*) and the benthivorous common carp (*Cyprinus carpio L.*) appeared to perpetuate the poor water condition. This, in turn, led to a decline in submerged plants.

Restoration of Lake Eymir by fish manipulation, which was implemented following major external nutrient loading reduction, was performed by removal of half of benthi-planktivorous fish between 1998 and 1999. This enhanced several variables, including the underwater light climate, and led to a significant reduction of the concentrations of especially suspended solids and chlorophyll-a (Beklioglu et al., 2003). Re-colonisation of submerged plants with 40 to 90% of surface coverage occurred upon the improvement of the underwater light climate from onward 2000 to 2003 (Beklioglu & Tan, accepted). Following re-colonization of submerged plants, the concentrations of all the nutrients especially TP significantly decreased. In 2001, the water level dropped nearly a meter and a great increase in the submerged plant cover to 90% of the total lake surface area with high nutrient concentrations was observed. Although deterioration in water quality was observed in 2003 especially in Secchi disk transparency resulted from increased concentrations of chlorophyll-a and suspended solids, the lake maintained its vegetated state. Despite a significant recovery in 1999, the submerged plants did not re-establish and their coverage remained very low, 6% of the lake surface area. Colonization of submerged plants began in 2000 and reached 50% coverage of the lake area. This study will mainly focus on the data that was collected during 2004-2005 along with the previous data (1997-2003) to maintain the integrity of the whole data set.

3. MATERIALS & METHODS

Water samples from both Lakes Eymir & Mogan were collected from March 2004 to November 2005. Data for the period of 1997-2003 were obtained from Burnak (1998), Tan (2002), Beklioglu et al., (2003); Karapınar & Beklioglu, 2006). Samples for chemical analyses, phytoplankton and zooplankton were taken biweekly intervals during the spring and summer, and monthly intervals during the winter. Prior to our study, the lake was sampled with a limited number of occasions between 1993-1995 for water chemistry (Altinbilek et al., 1995).

In Lake Mogan, depth-integrated water samples were taken from a fixed buoy at a mid-lake station using a 1.6 m long plastic tube sampler, which sampled a water column of about 1.5 m. Water samples were also taken from Sukesen brook just before reaching the Lake Mogan, Gölcük, Yavrucak and Çölovasi brooks, which had about 1, 0.9 and 3 km distances to the lake Mogan, respectively, as well as from the outflow just after leaving the lake. In Lake Eymir, water samples were taken from the epilimnion using a weighted length of polyethylene hose, and from Inflows of Lake Eymir (I and II), just before reaching to the lake Eymir and the outflow of the lake.

Water for chemical analyses was stored in acid-washed 1-l Pyrex bottles. TP and nitrite & nitrate (NO_2-N+NO_3-N) were analysed using the methods described by Mackereth et al., (1978) to precisions of ± 8%. Ammonium-nitrogen (NH_4-N) was determined according to Chaney & Morbach (1962) to precisions of ± 4%. Temperature and dissolved oxygen concentrations were measured at 0.5 m intervals using a WTW oxygen meter to a precision of ± 1 %. Secchi disk transparency was recorded using a 20 cm in diameter Secchi disk.

pH of the water samples were measured using an Orion 410A+ model pH meter. Conductivity and salinity were measured using an Orion model 115 conductivity meter to a precision of ± 1%.

Soluble reactive phosphate (SRP), total phosphorus (TP) and total oxidised nitrogen (nitrite and nitrate: NO_2-N+NO_3-N) were analysed using the methods described by Mackereth et al., (1978) to precisions of ± 3%, ± 8% and ± 8%, respectively. Ammonium-nitrogen (NH_4-N) was determined according to Chaney & Morbach (1962) to precisions of ± 4%. Silicate was measured following Golterman et al. (1978) and chlorophyll-a concentration was measured after ethanol extraction (Jespersen & Christoffersen, 1987).

The hydraulic residence time was estimated from dividing the lake volume (V_{lake}) by the volume of water flowing into the lake (I) per unit of time (Vollenweider, 1975).

Phytoplankton samples were collected with long tube sampler and 25 ml of the sample was fixed with a 1% Lugol solution to determine the phytoplankton species composition. At least 100 individuals were identified using a Leica DMI 4000B model inverted microscope (Presscott, 1982, Barber & Haworth, 1981, John & Brook 2005). Biovolumes were determined from the measurements of the linear dimensions of ten preserved cells of each taxon, using formulae for appropriate geometric shapes (Wetzel & Likens, 1991, Hillebrand et al., 1999 and Sun J. et al., 2003). Biovolume density of each species (μm^3 ml^{-1}) was determined by the multiplication of average cell volume by cell population density. Community biovolume density was calculated as the sum of the values for all species.

Zooplankton samples were collected with plankton net with a 45-μm meshed size (50 cm length and 12.5 cm mouth opening) and was preserved with a Lugol solution. Zooplankton was identified and counted using a Leica MZ16 model stereo microscope. If sample were dense, subsamples were taken and at least 100 individuals of dominant species were counted (Scourfield & Harding , 1966, Bottrell et al., 1976, Smith, 2001). Copepods were categorised as adults, copepodites and nauplii, and Cladocera and Rotifera were identified possible to the nearest taxonomic level.

The body length of zooplankton and phytoplankton was measured using Leica DFC 280 camera and using Leica Qwin and Leica IM50 computer programmes.

Submerged plants were surveyed in September 2004 and August 2005. Lake Eymir was divided into eighteen and twentytwo transects with 200 to 250 m between each in 2004 and 2005, respectively. On average three observations were conducted (1 to 3 points) on each transects. Lake Mogan was divided into fourteen and fifteen transects with 200 to 250 m between each in 2004 and 2005, respectively. On average three observations were conducted on each transects. In both lakes, the observations on transects were made from a boat using a glass and 1x1 m wooden quadrat to estimate Plant Volume Infested (PVI; Canfield et al., 1984). This was used to express macrophyte density by measuring percentage coverage, plant height and water depth. In each zone, the sampling stations were recorded electronically using a Global Positioning System (GPS) to develop a digital vegetation map. Then data from all the stations were interpolated to estimate coverage for the whole lake on a digitised map using ArcGIS Ver. 9.1 software (Rockware Inc, 2004). 1998-2003 maps were taken from Tan, 2002. 2004 and 2005 maps were prepared using ArcGIS Ver. 9.1 (Doğan Karabulut O, unpublished data). The plant samples were taken using an Ekman grab and anchor. Aquatic plants were identified using Haslam et al., (1982) and Altinayar (1988). The samples were dried at 105 ^{o}C for 24 h and weighed.

The fish density estimation was conducted once in a year when the fish were expected to be most evenly distributed and young-of the-year (YOY) large enough to be caught in the nets. Fish abundance was obtained from overnight catches in multiple mesh-sized gill nets and expressed as catch per unit effort (CPUE). In 2004 and 2005, the gill nets included ten different mesh sizes (7, 9, 12, 16, 22, 25, 36, 42, 50, 55, 65 and 90 mm). The length and depth of each net were 100 m and 3.5 m, respectively. The nets were placed along the shoreline in the littoral zone and perpendicular to the shoreline from the littoral zone to the open water. Each fishing effort lasted a week to cover the whole lake. The total length of the fish was measured from the mouth to the end of the tail fin. The specimens were also weighed. No fish stock determinations were carried out in Lake Mogan.

Hydrological data were obtained from the General directorate of Electrical Power, Resource Survey & Development Administration (EİE) (EİE, 2004, 2005). The lake level, given in metres above sea level (m a.s.l.), was recorded daily from a fixed gauge positioned at the south shore of the lake.

3.1. Statistical methods

The XLSTAT (Addin Software Design, U.S.A, 2006) was used for performing statistical analysis. Selected variables measured across 1997-2005 in Lakes Eymir and Mogan were used for analyses. Significance of differences was tested using Kruskal-Wallis test and post hoc comporisons were made by Dunn's procedure with the Bonferroni type adjustments. Water levels were given in meters above sea level. Spring lake level and spring Secchi depth included the measurements of the variable from March to June. The annual areal TP and DIN loadings were estimated as the sum of the loads of the inflows per unit of lake area given in g m^{-2} yr^{-1}. Algal volumes were given in million of μm^3 mI^{-1} and zooplankton in numbers per litre ($ind.l^{-1}$). Zooplankton densities were given as summer means including measurements from May to September. Remaining variables were given as annual mean±SD.

Z-scores for water levels were calculated in order to determine the trends in long-term data. Z-scores were calculated by first subtracting the calculated mean of all water level data for the study period from an individual observation and then dividing this difference by the standard deviation of all the observations (Gerten & Adrian, 2000). Z-score transformation normalised the variables for easier visual comparison. Significance of differences between the years for each pair of means of water levels was tested using Kruskal-Wallis test and post hoc comporisons were made by Dunn's procedure with the Bonferroni type adjustments.

4. RESULTS

4.1. Lake Eymir

4.1.1 Hydrology

Lake Eymir had a thick and stable ice cover between January to March 2004 and January to February 2005. Therefore, the lake was not sampled.

The hydraulic residence time differed between the years. The longest hydraulic residence time was recorded in 2005 (13.15 yr^{-1}) which differed significantly from the previous years. The shortest hydraulic residence time 0.3 yr-1 was recorded in 1998 (Table 4.1).

In Lake Eymir, fluctuations of the lake level varied with the season. A typical annual fluctuation was observed every year with the increasing water level during the rainy periods in late winter and spring and with the decreasing water level during the dry period in summer, and in some years fluctuations also occurred in autumn and winter. The amplitudes (maximum-minimum water levels) of the annual water level fluctuations recorded from 1997 to 2005 were 0.9, 0.7, 0.66, 1.25, 0.73, 1.13, 1.18, 0.99 and 0.45 m.a.s.l, respectively (mean amplitude of 0.88±0.27 m a.s.l).

The lake level fluctuated both annually and interannually throughout the study period (Figure 4.1). Z-scores were calculated to standardize the water level data. Firstly, Kruskal-Wallis ANOVA test was applied to z-scores of annual water levels. The results showed that years were significantly different in terms of water levels ($p< 0.0001$). Multiple pairwise comparisons using the Dunn’s procedure with the Bonferroni type adjustments was used to determine the differences between years. As a result, the water levels in 2001, 2004 and 2005 were significantly lower than years 1998, 1999 and 2000 (Bonferroni corrected significance level: 0.0014). Using the z-scores values of the water levels for the study period, multiple pairwise comparisons using the Dunn's procedure constructed three hydrologically different periods, which were grouped in homogeneous subsets as low, regular and high water levels.

Years 2001, 2004 and 2005 were referred to low water level (LWL) years; whereas years 1998, 1999 and 2000 were referred to high water level (HWL) years and the remaining study period is regarded as regular water level (RWL) years (Figure 4.1).

The spring water level fluctuation showed similarity to the annual water levels because the lowest spring lake levels were recorded in 2001 and 2005 (Table 4.1). Kruskal-Wallis ANOVA test was applied to spring water levels. The results showed that years were significantly different in terms of the spring water levels (p: 0.002). As a result of multiple pairwise comparisons, the spring water levels in 2001 and 2005 were significantly lower than other years (Bonferroni corrected significance level: 0.0014). Year 2000 was referred to HWL. Consequently, 2001 and 2005 were regarded as the driest year throughout the study period due to the lowest lake level.

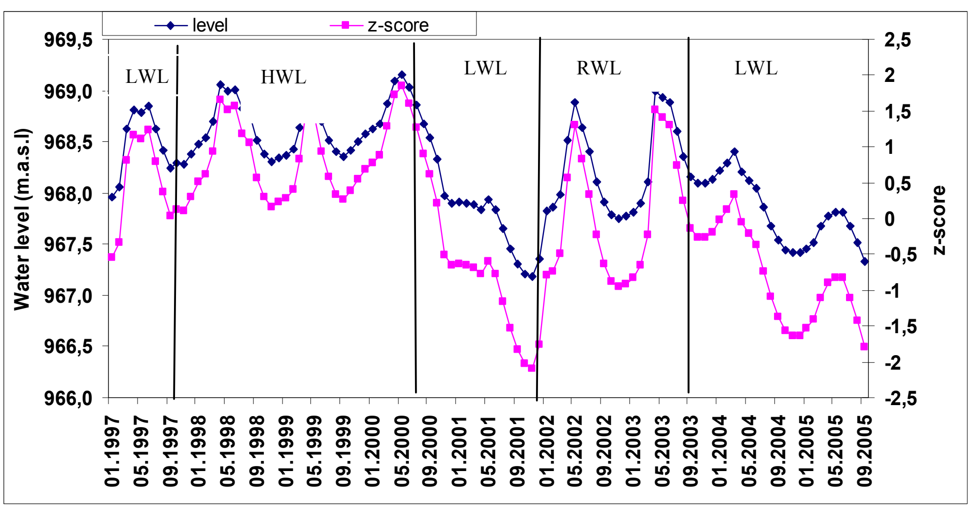

Figure 4.1. Water level fluctuations recorded from 1997 to 2004 in Lake Eymir. (RWL: regular water level, HWL: high water level, LWL: low water level).

4.1.2 Conductivity and Salinity

The salinity and conductivity recorded in 2001 and 2005 were significantly higher than that of the other years ($p<0.0001$ for both salinity and conductivity) (Table 4.1). Both the salinity and conductivity inversely correlated with the lake level (r:-0.37, p: 0.000 and r:-0.12, p: 0.000, respectively). In addition, the salinity and conductivity in 2004 was higher than that of other years (1.12 ‰ and 2.25 mS cm^{-1}, respectively). Salinity and conductivity changes between 1997 and 2005 were given in the Table 4.1 and Figure 4.2.

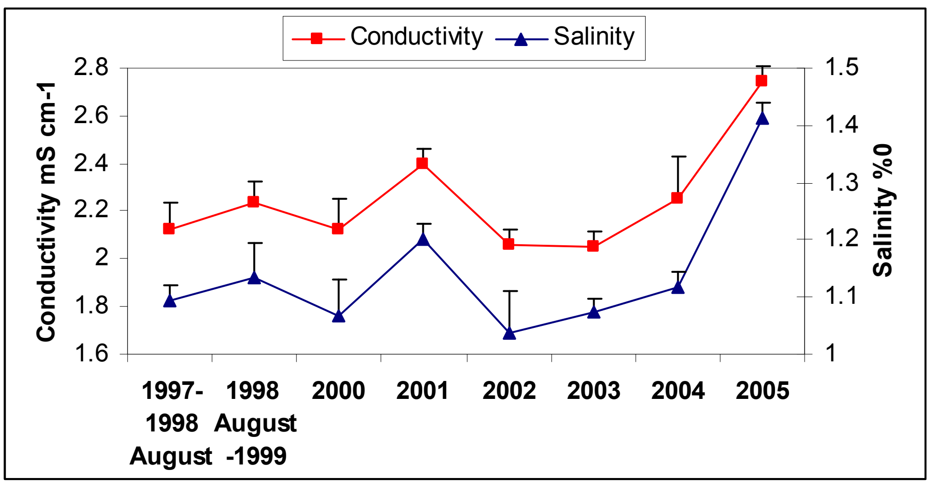

Figure 4.2. Changes in the conductivity and salinity in Lake Eymir from 1997 to 2005

4.1.3 pH

The pH changes between 1997 and 2005 were given in the Table 4.1 and Figure 4.3. The pH value was also significantly higher in 2002 and 2003 than that of other years ($p<0.001$, Table 4.1, Figure 4.3). Moreover, the pH was inversely correlated with the water level (r:-0.413, p:0.000).

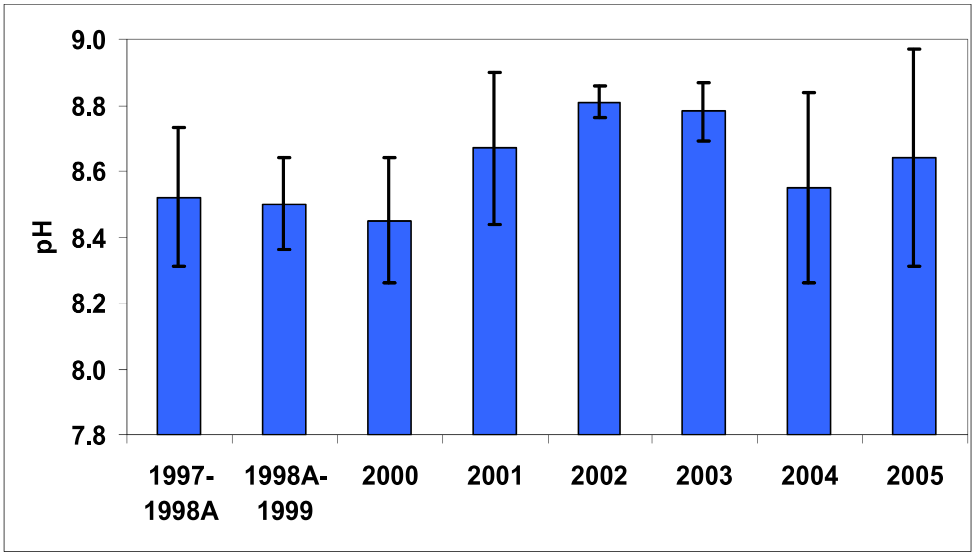

Figure 4.3. Changes in the mean ± SD of pH in Lake Eymir from 1997 to 2005**.**

4.1.4 Temperature and Oxygen

During the period of 1997-2005, there were no significant changes in the water temperature and dissolved oxygen concentration (p: 0.388 and p: 0.088 respectively). The lake water stratified and the thermocline set at the depth of 3.5 m early in stratification period and then deepened to 4.5 m later in summer, leaving the hypolimnion with depths of 2 to 1.5 m, respectively. The summer thermal stratification lead to a steep temperature gradient between the hypolimnion and epilimnion, (summer mean: 16.9±3.7 and 22.2±2.4^{o}C, respectively) in this period. The hypolimnetic dissolved oxygen concentration remained much lower than that of the epilimnetic dissolved oxygen concentration (0.62±0.9 and 4.9±1.9, respectively. However, in 2003, the hypolimnetic dissolved oxygen concentration was close to the epilimnetic dissolved oxygen concentration (3.68±0.72 and 5.9±1.5, respectively). In 2001 and 2004, thermal stratification did not occur and the water mixed throughout the summer due to the significant decline in the water level. Moreover, the water temperature was warmer in 2001 (24.2±2^{o}C) and 2004 (23.4±0.5^{o}C). In addition, the dissolved oxygen concentration was higher in 2001 (6±2.5 mg l-1) even though the concentration was low in 2004 (2.34±0.26 mg l^{-1}).

Summer temperature and summer oxygen values were significantly different in 2001 and 2005 than other years (p: 0.02 and p: 0.06, respectively) (Table 4.1). During the study period summer oxygen values were lower than annual values due to strafication in the lake (Figure 4.4).

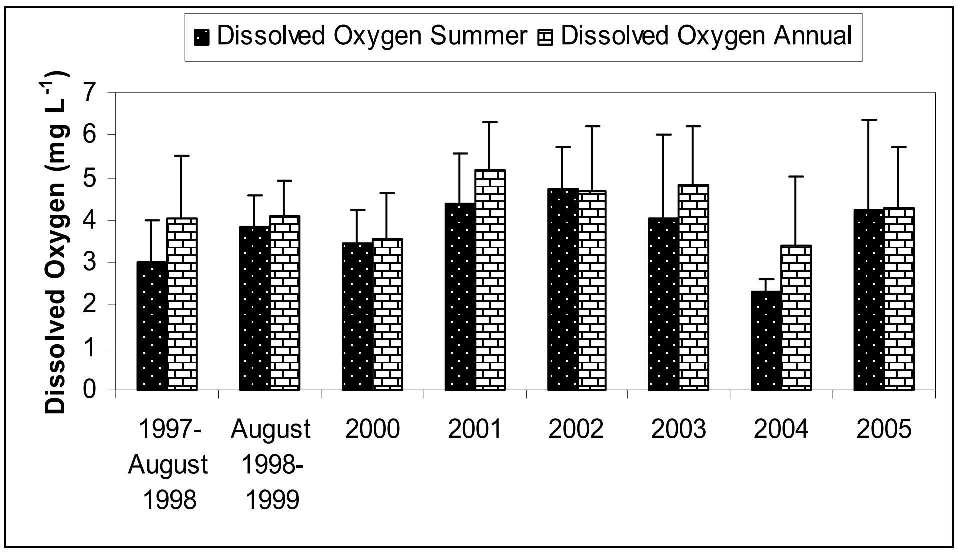

Figure 4.4. Annual and summer mean±SD of dissolved oxygen concentration from 1997 to 2005 in Lake Eymir**.**

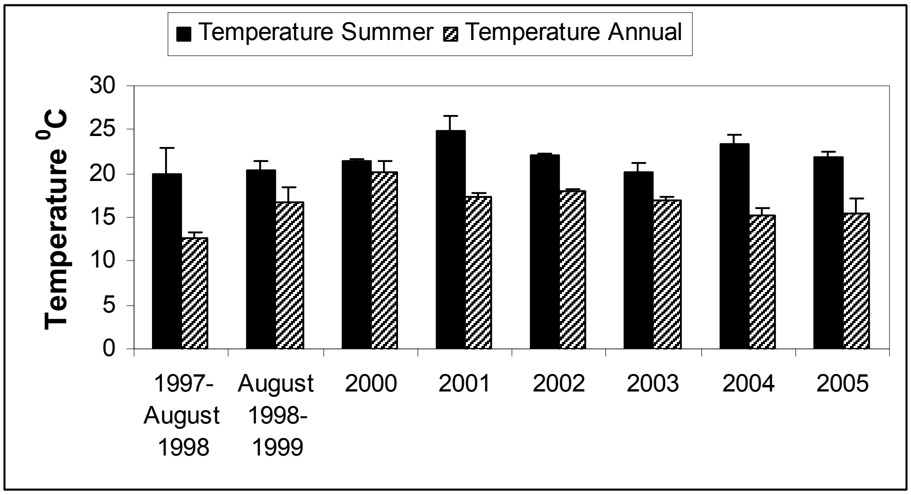

Figure 4.5. Annual and summer mean±SD of temperature from 1997 to 2005 in Lake Eymir**.**

4.1.5 Secchi disk transparency, concentration of suspended solids and chlorophyll-a

Spring Secchi disk transparency recorded in 2004 was very low compared to the clearwater state (75 cm and 275 cm, respectively) in Lake Eymir. However, annual Secchi disk transparency recorded in 2004 was not significantly lower than that of the clearwater period (216 cm and 247 cm, respectively). This decline in spring Secchi disk transparency was attributed to significant increase in concentrations of especially chlorophyll-a and lesser extend to suspended solids in spring (51 μg l^{-1} and 12.5 mg l^{-1}, respectively). The annual concentrations of chlorophyll-a and to suspended solids were remained lower than those in spring (18.7 μg l^{-1} and 11.3 mg l^{-1} , respectively) (Table 4.1 and Figure 4.6).

Conversly, Spring Secchi disk transparency was high in 2005 (250 cm). However, annual Secchi disk transparency recorded in 2005 was significantly lower than that of the clearwater period (129 cm and 247 cm, respectively). This decline in annual Secchi disk transparency was attributed to significant increase in concentrations of chlorophyll-a and suspended solids in 2005 (91 μg l^{-1} and 22 mg l^{-1}, respectively).

The spring concentrations of chlorophyll-a and to suspended solids were remained lower than annual concentrations (3 μg l^{-1} and 7 mg l^{-1}, respectively) (Table 4.1 and Figure 4.6).

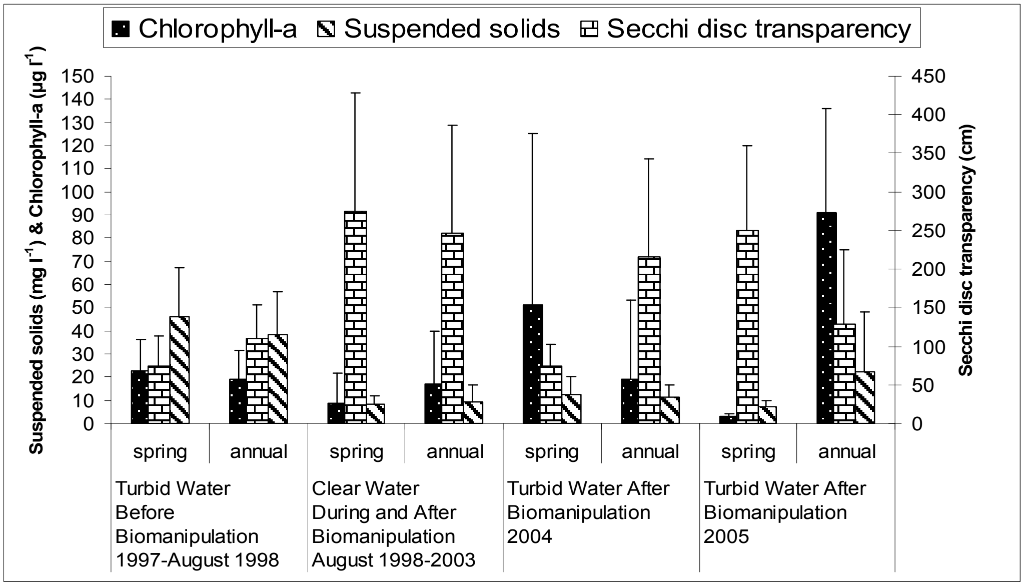

Figure 4.6. Spring and annual changes in the concentrations of chlorophyll-a (mg l^{-1}), suspended solids (mg l^{-1}) and Secchi disk transparency (cm) in Lake Eymir from 1997 to 2005.

4.1.6 Nutrients

Total phosphorous concentrations significantly increased compared to clearwater state (527.8 μg l^{-1} and 254.6 μg l-1, respectively), although annual and spring TP areal loads remained significantly lower than previous years (0.004 and 0.009 g m^{-2} yr^{-1} respectively) in 2004. Soluble reactive phosphate (SRP) concentration was also significantly higher in 2004 than in the clearwater state (445.1 μg l^{-1} and 163.6 μg l^{-1}, respectively) (Figure 4.7). In 2005, TP and SRP concentrations were high but lower than 2004 (452 μg l^{-1} and 308-μg l^{-1}, respectively) (Table 4.1).

No significant difference was recorded for nitrate (NO_3-N): 0.13 mg l^{-1}). However ammonium concentration was found higher (NH_4-N: 0.27 µg l^{-1}) (Table 4.1). Dissolved inorganic nitrogen concentration (DIN) was 0.39 mg l^{-1}. Annual and spring DIN loads were lower than previous years (0.01 and 0.01 g m^{-2} yr^{-1}, respectively) (Table 4.1). Although there were no external loading for DIN, DIN concentration was higher than 2004 (0.39 mg l-1). Nitrate (NO3-N) and ammonium concentrations were found higher than 2004 (0.21 mg l-1 and 0.31 µg l-1 respectively) (Table 4.1). The amount of ammonium in DIN was higher in turbid water state after biomanipulation than clearwater state (Figure 4.10).

The concentrations of silicate were significantly higher than previous years in 2004 and 2005 ($p<0.0001$, 5.4 and 5.5 mg l^{-1} respectively) (Figure 4.11).

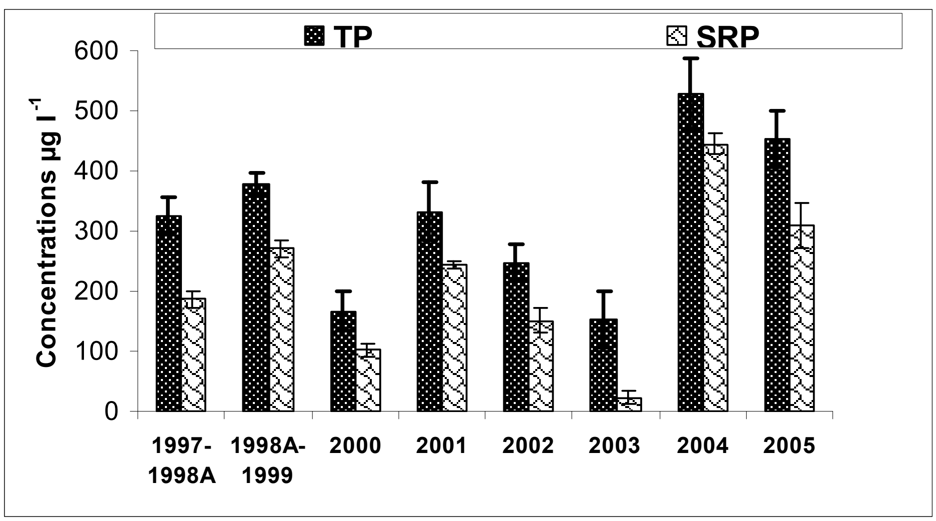

Figure 4.7. Changes in the mean±SD of concentrations of TP and SRP from 1997 to 2005 in Lake Eymir.

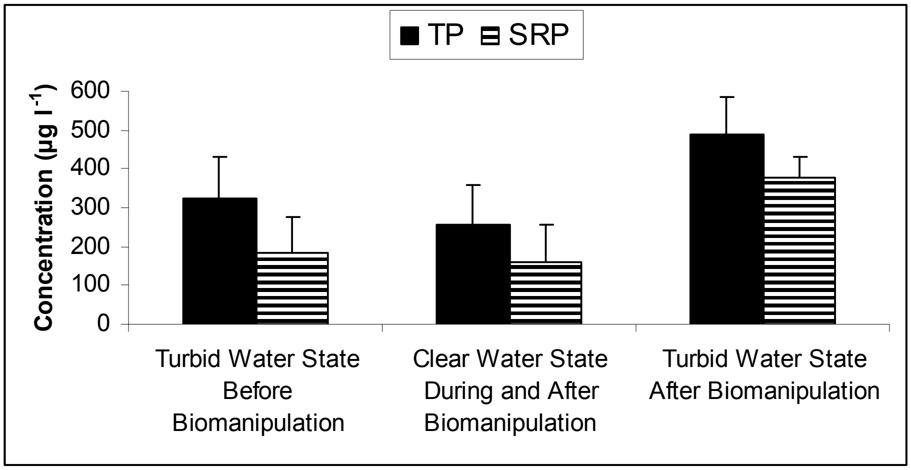

Figure 4.8. Changes in the concentrations of TP and SRP during the study period in Lake Eymir (Turbid water state before biomanipulation: 1997-1998 August, Clearwater state during and after biomanipulation: 1998 August-2003, Turbid water state after biomanipulation 2004-2005).

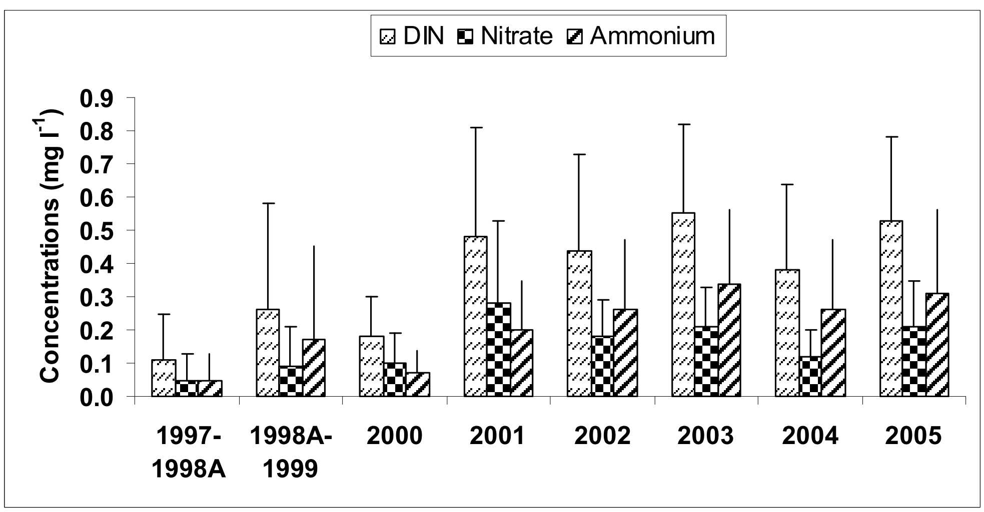

Figure 4.9. Changes in the mean±SD of concentrations of DIN, Nitrate (NO_3-N and Ammonium (NH_4-N) from 1997 to 2005 in Lake Eymir.

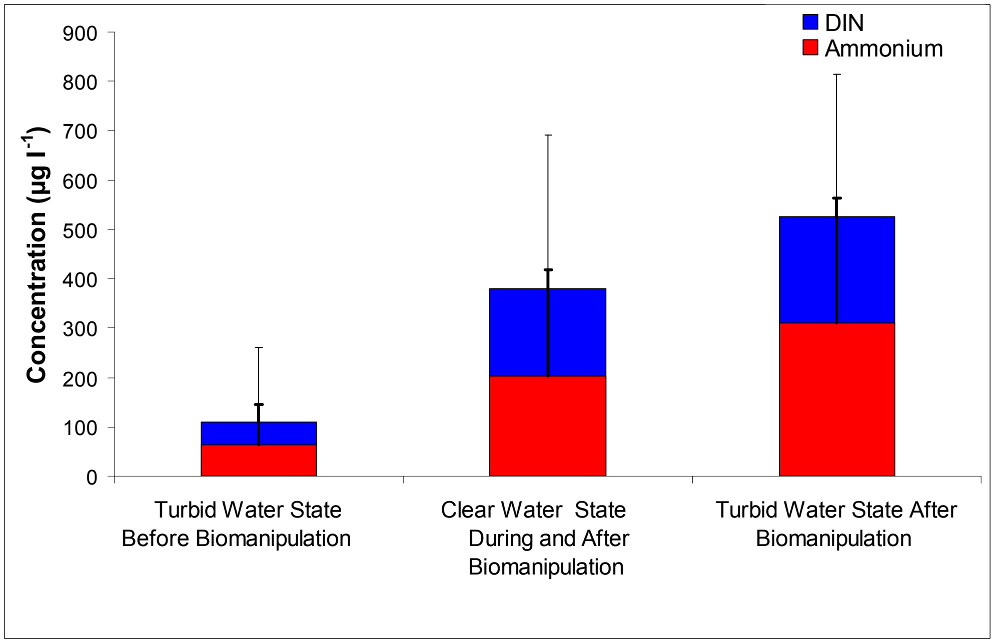

Figure 4.10 The amount of ammonium in DIN during the study period in Lake Eymir (turbid water state before biomanipulation: 1997-1998 August, Clearwater state during and after biomanipulation: 1998 August-2003, turbid water state after biomanipulation 2004-2005).

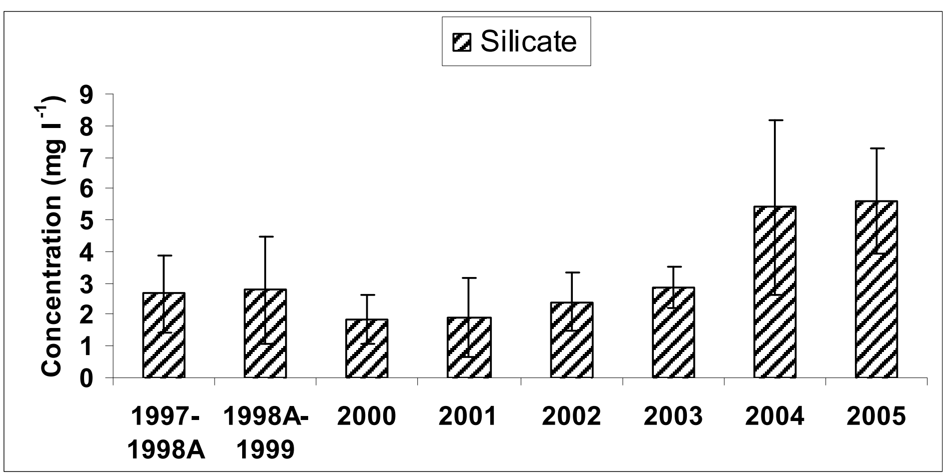

Figure 4.11. Changes in the mean±SD of concentrations of silicate from 1997 to 2005 in Lake Eymir.

Table 4.1. Mean±SD of selected variables measured across 1997-2005 in Lake Eymir. Significance of differences was tested using Kruskal-Wallis test and post hoc comparisons were made by Dunn's procedure with the Bonferroni type adjustments. Water levels are given in meters above sea level. Spring lake level and spring Secchi depth included the measurements of the variable from March to June. The annual areal TP and DIN loadings are estimated as the sum of the loads of the inflows per unit of lake area given in g m^{-2} yr^{1}.

	1997-August 1998	August 1998-1999	2000	2001	2002	2003	2004	2005	Kruskal-Wallis 1997-2005	Dunn's procedure with the Bonferroni type adjustments 1997-2005
Annual lake level (ma.s.l)	968.44±0.3	968.66±0.3	968.58±0.2	968.64±0.4	967.62±0.3	968.12±0.3	967.89±0.3	967.62±0.2	p<0.0001	2001,2004,2005*all years
Spring lake level (m a.s.l)	968.94±0.16	968.83±0.2	969.04±0.2	967.87±0.4	968.50±0.4	968.73±0.4	968.24±0.3	967.75±0.1	p:0.002	2001,2005*9798,9899,2000,2003
H.Residence time (yr^{-1})	1.4	0.38	0.24	7	1.62	1.12	7.01	13.15	13.15	NA
Secchi Depth (cm)	109±44	270±144	309±113	218±95	236±168	158±170	216±127	129±96	p<0.0001	9798,2005*all years
Spring Secchi Depth (cm)	74±39	435±134	302±189	253±119	254±181	243±171	75±27	250±110	p:0.023	9798,2004*9899,2000,2001
Spring Chl-a (µg l^{-1})	25±11	12±6	20±21	8±9	3±4	6±5	51±75	3±1	p:0.003	2002*9798,9899,2000
SS (mg l^{-1})	38±18	9±6	6±3	10±5	8±5	12±11	11±5	22±26	p<0.0001	97*allyears;2003,2005*9798,2000
Summer temperature	20±0.82	21±6.42	22±2.11	25±1.58	22±1.58	21±2.01	24±0.51	23±1.65	p:0.022	2001*all years
Summer Oxygen	2.98±1.5	3.83±0.74	3.71±1.12	5.74±1.73	5.27±1.57	5.19±2.09	2.33±0.26	5.61±1.82	p:0.006	2001,2005*all years
Conductivity (mS cm^{-1})	2.12±0.11	2.24±0.1	2.12±0.12	2.40±0.06	2.06±0.16	2.05±0.22	2.25±0.18	2.75±0.21	p<0.0001	2001,2005*all years
Salinity (‰)	1.09±0.03	1.14±0.06	1.07±0.07	1.2±0	1.04±0.07	1.07±1.11	1.12±0.12	1.41±0.12	p<0.0001	2001,2005*all years
pH	8.52±0.21	8.50±0.14	8.45±0.19	8.67±0.23	8.81±0.05	8.78±0.09	8.55±0.29	8.64±0.33	p<0.0001	2001*9899,2000,2002
TP (µg l^{-1})	326±106	378±81	167±63	332±95	248±147	152±54	528±132	452±157	p<0.0001	2000,2003*all years
SRP (µg l^{-1})	186±91	271±79	102±58	244±123	151±94	23±20	445±148	308±235	p<0.0001	2004*all years;2003*all years
DIN (mg l^{-1})	0.11±0.14	0.26±0.32	0.18±0.12	0.48±0.33	0.44±0.29	0.55±0.27	0.38±0.26	0.53±0.25	p<0.0001	9798,9899,2000*all years
NO_3-N (mg l^{-1})	0.05±0.08	0.09±0.12	0.10±0.09	0.28±0.25	0.18±0.11	0.21±0.12	0.12±0.08	0.21±0.14	p<0.0001	9798,9899,2000*2001,2002,2003,2005
NH_4-N (mg l^{-1})	0.05±0.08	0.17±0.28	0.07±0.07	0.20±0.15	0.26±0.21	0.34±0.22	0.26±0.21	0.31±0.25	p<0.0001	2003,2004,2005*9798,9899,2000
TP Areal Load	0.01±0.21	0.15±0.23	0.19±0.25	0.02±0.06	0.01±0.02	0.03±0.07	0.01±0.01	0±0	p<0.0001	2002,2003,2004,2005*all years
DIN Areal Load	0.15±0.22	0.16±0.29	0.21±0.29	0.05±0.16	0.03±0.07	0.14±0.31	0.01±0.02	0±0	p<0.0001	2002,2004,2005*all years

Table 4.1. Mean±SD of selected variables measured across 1997-2005 in Lake Eymir. Significance of differences was tested using Kruskal-Wallis test and post hoc comparisons were made by Dunn's procedure with the Bonferroni type adjustments. Water levels are given in meters above sea level. Spring lake level and spring Secchi depth included the measurements of the variable from March to June. The annual areal TP and DIN loadings are estimated as the sum of the loads of the inflows per unit of lake area given in g m^{-2} yr^{1}.

	1997-August 1998	August 1998-1999	2000	2001	2002	2003	2004	2005	Kruskal-Wallis 1997-2005	Dunn's procedure with the Bonferroni type adjustments 1997-2005
Annual lake level (m a.s.l)	968.44±0.3	968.66±0.3	968.58±0.2	968.64±0.4	967.62±0.3	968.12±0.3	967.89±0.3	967.62±0.2	p<0.0001	2001,2004,2005*all years
Spring lake level (m a.s.l)	968.94±0.16	968.83±0.2	969.04±0.2	967.87±0.4	968.50±0.4	968.73±0.4	968.24±0.3	967.75±0.1	p:0.002	2001,2005*9798,9899,2000,2003
H.Residence time (yr^{-1})	1.4	0.38	0.24	7	1.62	1.12	7.01	13.15	13.15	NA
Secchi Depth (cm)	109±44	270±144	309±113	218±95	236±168	158±170	216±127	129±96	p<0.0001	9798,2005*all years
Spring Secchi Depth (cm)	74±39	435±134	302±189	253±119	254±181	243±171	75±27	250±110	p:0.023	9798,2004*9899,2000,2001
Spring Chl-a (µg l^{-1})	25±11	12±6	20±21	8±9	3±4	6±5	51±75	3±1	p:0.003	2002*9798,9899,2000
SS (mg l^{-1})	38±18	9±6	6±3	10±5	8±5	12±11	11±5	22±26	p<0.0001	97*allyears;2003,2005*9798,2000
Summer temperature	20±0.82	21±6.42	22±2.11	25±1.58	22±1.58	21±2.01	24±0.51	23±1.65	p:0.022	2001*all years
Summer Oxygen	2.98±1.5	3.83±0.74	3.71±1.12	5.74±1.73	5.27±1.57	5.19±2.09	2.33±0.26	5.61±1.82	p:0.006	2001,2005*all years
Conductivity (mS cm^{-1})	2.12±0.11	2.24±0.1	2.12±0.12	2.40±0.06	2.06±0.16	2.05±0.22	2.25±0.18	2.75±0.21	p<0.0001	2001,2005*all years
Salinity (‰)	1.09±0.03	1.14±0.06	1.07±0.07	1.2±0	1.04±0.07	1.07±1.11	1.12±0.12	1.41±0.12	p<0.0001	2001,2005*all years
pH	8.52±0.21	8.50±0.14	8.45±0.19	8.67±0.23	8.81±0.05	8.78±0.09	8.55±0.29	8.64±0.33	p<0.0001	2001*9899,2000,2002
TP (µg l^{-1})	326±106	378±81	167±63	332±95	248±147	152±54	528±132	452±157	p<0.0001	2000,2003*all years
SRP (µg l^{-1})	186±91	271±79	102±58	244±123	151±94	23±20	445±148	308±235	p<0.0001	2004*all years;2003*all years
DIN (mg l^{-1})	0.11±0.14	0.26±0.32	0.18±0.12	0.48±0.33	0.44±0.29	0.55±0.27	0.38±0.26	0.53±0.25	p<0.0001	9798,9899,2000*all years
NO_3-N (mg l^{-1})	0.05±0.08	0.09±0.12	0.10±0.09	0.28±0.25	0.18±0.11	0.21±0.12	0.12±0.08	0.21±0.14	p<0.0001	9798,9899,2000*2001,2002,2003,2005
NH_4-N (mg l^{-1})	0.05±0.08	0.17±0.28	0.07±0.07	0.20±0.15	0.26±0.21	0.34±0.22	0.26±0.21	0.31±0.25	p<0.0001	2003,2004,2005*9798,9899,2000
TP Areal Load	0.01±0.21	0.15±0.23	0.19±0.25	0.02±0.06	0.01±0.02	0.03±0.07	0.01±0.01	0±0	p<0.0001	2002,2003,2004,2005*all years
DIN Areal Load	0.15±0.22	0.16±0.29	0.21±0.29	0.05±0.16	0.03±0.07	0.14±0.31	0.01±0.02	0±0	p<0.0001	2002,2004,2005*all years

Table 4.1. Mean±SD of selected variables measured across 1997-2005 in Lake Eymir. Significance of differences was tested using Kruskal-Wallis test and post hoc comparisons were made by Dunn's procedure with the Bonferroni type adjustments. Water levels are given in meters above sea level. Spring lake level and spring Secchi depth included the measurements of the variable from March to June. The annual areal TP and DIN loadings are estimated as the sum of the loads of the inflows per unit of lake area given in g m^{-2} yr^{1}.

	1997-August 1998	August 1998-1999	2000	2001	2002	2003	2004	2005	Kruskal-Wallis 1997-2005	Dunn's procedure with the Bonferroni type adjustments 1997-2005
Annual lake level (ma.s.l)	968.44±0.3	968.66±0.3	968.58±0.2	968.64±0.4	967.62±0.3	968.12±0.3	967.89±0.3	967.62±0.2	p<0.0001	2001,2004,2005*all years
Spring lake level (m a.s.l)	968.94±0.16	968.83±0.2	969.04±0.2	967.87±0.4	968.50±0.4	968.73±0.4	968.24±0.3	967.75±0.1	p:0.002	2001,2005*9798,9899,2000,2003
H.Residence time (yr^{-1})	1.4	0.38	0.24	7	1.62	1.12	7.01	13.15	13.15	NA
Secchi Depth (cm)	109±44	270±144	309±113	218±95	236±168	158±170	216±127	129±96	p<0.0001	9798,2005*all years
Spring Secchi Depth (cm)	74±39	435±134	302±189	253±119	254±181	243±171	75±27	250±110	p:0.023	9798,2004*9899,2000,2001
Spring Chl-a (μg l^{-1})	25±11	12±6	20±21	8±9	3±4	6±5	51±75	3±1	p:0.003	2002*9798,9899,2000
SS (mg l^{-1})	38±18	9±6	6±3	10±5	8±5	12±11	11±5	22±26	p<0.0001	97*allyears;2003,2005*9798,2000
Summer temperature	20±0.82	21±6.42	22±2.11	25±1.58	22±1.58	21±2.01	24±0.51	23±1.65	p:0.022	2001*all years
Summer Oxygen	2.98±1.5	3.83±0.74	3.71±1.12	5.74±1.73	5.27±1.57	5.19±2.09	2.33±0.26	5.61±1.82	p:0.006	2001,2005*all years
Conductivity (mS cm^{-1})	2.12±0.11	2.24±0.1	2.12±0.12	2.40±0.06	2.06±0.16	2.05±0.22	2.25±0.18	2.75±0.21	p<0.0001	2001,2005*all years
Salinity (‰)	1.09±0.03	1.14±0.06	1.07±0.07	1.2±0	1.04±0.07	1.07±1.11	1.12±0.12	1.41±0.12	p<0.0001	2001,2005*all years
pH	8.52±0.21	8.50±0.14	8.45±0.19	8.67±0.23	8.81±0.05	8.78±0.09	8.55±0.29	8.64±0.33	p<0.0001	2001*9899,2000,2002
TP (μg l^{-1})	326±106	378±81	167±63	332±95	248±147	152±54	528±132	452±157	p<0.0001	2000,2003*all years
SRP (μg l^{-1})	186±91	271±79	102±58	244±123	151±94	23±20	445±148	308±235	p<0.0001	2004*all years;2003*all years
DIN (mg l^{-1})	0.11±0.14	0.26±0.32	0.18±0.12	0.48±0.33	0.44±0.29	0.55±0.27	0.38±0.26	0.53±0.25	p<0.0001	9798,9899,2000*all years
NO_3-N (mg l^{-1})	0.05±0.08	0.09±0.12	0.10±0.09	0.28±0.25	0.18±0.11	0.21±0.12	0.12±0.08	0.21±0.14	p<0.0001	9798,9899,2000*2001,2002,2003,2005
NH_4-N (mg l^{-1})	0.05±0.08	0.17±0.28	0.07±0.07	0.20±0.15	0.26±0.21	0.34±0.22	0.26±0.21	0.31±0.25	p<0.0001	2003,2004,2005*9798,9899,2000
TP Areal Load	0.01±0.21	0.15±0.23	0.19±0.25	0.02±0.06	0.01±0.02	0.03±0.07	0.01±0.01	0±0	p<0.0001	2002,2003,2004,2005*all years
DIN Areal Load	0.15±0.22	0.16±0.29	0.21±0.29	0.05±0.16	0.03±0.07	0.14±0.31	0.01±0.02	0±0	p<0.0001	2002,2004,2005*all years

4.1.7 Phytoplankton

During biomanipulation, compared to the pre-biomanipulation period in Lake Eymir, the phytoplankton community did not change. Chlorophyta dominated the community ($0.88x10^6$ μm^3 ml^{-1}) but also cryptophyta and cyanobacteria biovolumes were significant ($0.21x10^6$ and $0.24x10^6$ μm^3 ml^{-1}, respectively (Table 4.2, Figure 4.12).

Phytoplankton community of clearwater state included species of chlorophyta, cyanobacteria, dinoflagellates, cryptophyta and diatoms. In 2003, the biovolume of diatom was significantly higher than other years ($p<0.0001$, $0.31x10^6$ μm^3 ml^{-1}) (Table 4.2, Figure 4.12).

In the period of turbid water state after biomanipulation, phytoplankton community included cyanobacteria, cryptophyta and diatom. Contribution of chlorophyta and dinoflagellates to phytoplankton numbers were too low (3% and 1%, respectively). In 2004, cyanobacteria dominated the community during summer ($0.17x10^6$ μm^3 ml^{-1}) and diatoms dominated the community during spring ($0.71x10^6$ μm^3 ml^{-1}) (Table 4.2).

In 2005, cyanobacteria dominated the community and their biovolumes were significantly higher than previous years (p: 0.017, $2.32x10^6$-µm3 ml^{-1}). The biovolume of diatom was significantly lower than other years ($p<0.0001$, $0.01x10^6$ μm^3 ml^{-1}). The biovolume of chlorophyta was significantly lower than other years in both 2004 and 2005 ($p<0.0001$, $0.07x10^6$ μm^3 ml^{-1}, $0.01x10^6$ μm^3 ml^{-1} respectively) (Table 4.2, Figure 4.12). Although the biovolume of cryptophyta decreased in 2004, their biovolume increased in 2005 and they become the second dominant group ($0.15x10^6$ μm^3 ml^{-1} and $0.87x10^6$ μm^3 ml^{-1}, respectively) (Table 4.2, Figure. 4.12).

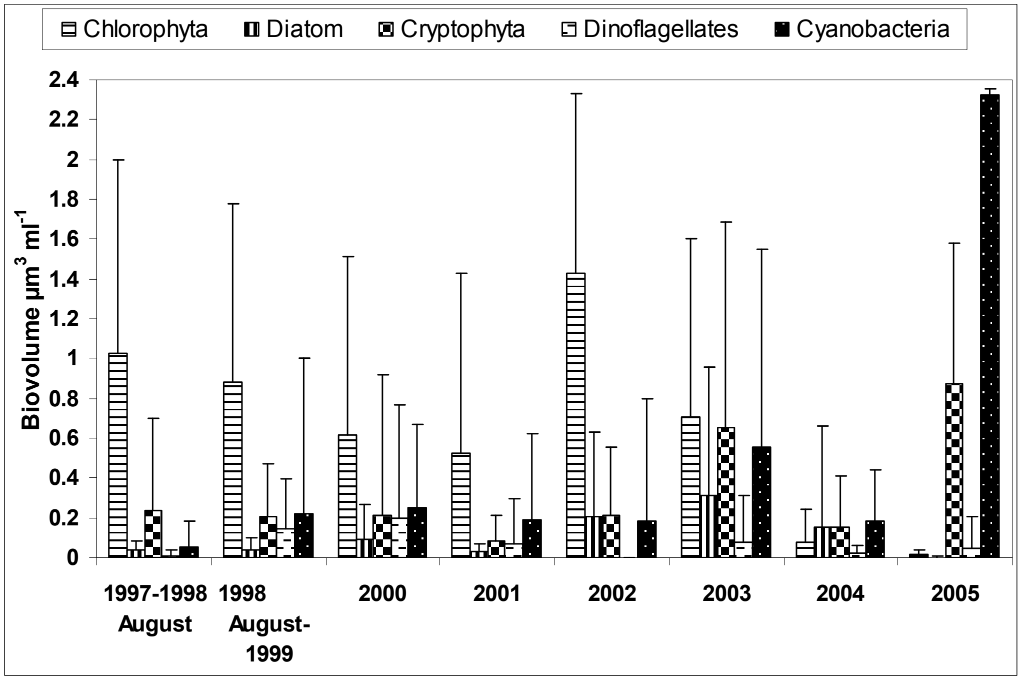

Figure 4.12. Changes in mean±SD of the biovolume of phytoplankton taxa from 1997 to 2005 in Lake Eymir.

4.1.8 Zooplankton community

The pre-biomanipulation zooplankton community of Lake Eymir was species-poor. The density of the large-sized grazers increased significantly in response to the fish removal (Beklioglu et al., 2003).

The fish removal led to a marked increase in the density of the large-bodied zooplankton *D. pulex* and *A. bacillifer*, which also dominated the zooplankton community during the clearwater state in the summer. Daytime samples taken in the hypolimnion, named hypolimnetic *D. pulex*, showed higher hypolimnion than epilimnion densities in both 2000 and 2001 (15 and 11 Indv l^{-1}) (Figure 4.13). Hypolimnetic *D.pulex* densities in 2000 were significantly higher than other years (p: 0.001). Hypolimnetic *D.pulex* densities were lower in both 2002 and 2003 (0.8 and 2.1 Indv l^{-1}, respectively) (Figure 4.13).

The zooplankton community of Lake Eymir was species-poor in 2004. The community was dominated mainly by rotifera, which also contributed 47% of the total biomass (15 Indv l^{-1}) (Figure 4.13). Cyclopoid copepods were the second dominant group after rotifera and contributed 24% to the total zooplankton biomass (7.7 Indv l^{-1}) (Figure 4.13). The density of the large-sized grazers especially the density of Calanoid Copepod *A. bacillifer* was significantly lower than previous years (2.3 ind.l^{-1}) ($p<0.0001$, Table results, fig zoop). Cladocerans were represented by one species, *D.pulex* and contributed 17% to total zooplankton biomass (6.8 Indv l^{-1}). Hypolimnetic *D.pulex* density was low in 2004 (0.4 Indv l^{-1}) (Figure 4.13).

In 2005, the zooplankton community of Lake Eymir was dominated mainly by rotifera, which contributed 98% of the total biomass (128 Indv l^{-1}). Cyclopoid copepods contribution to total zooplankton biomass was lower than 2004 (1%, 6.8 Indv l^{-}). The density of the large-sized grazers especially the density of Calanoid Copepod *A. bacillifer and D.pulex* were significantly lower than previous years (0.5 and 1 ind.l^{-1}, respectively). Hypolimnetic *D.pulex* was not observed in 2005 (Figure 4.13). Body size of the *D.pulex* changed significantly throughout the study (p: 0.000). Their length slightly increased after the biomanipulation (0.95 to 1.3 mm). During the turbid water state after biomanipulation there was no change in *D.pulex* body lenghth compared to clearwater state.

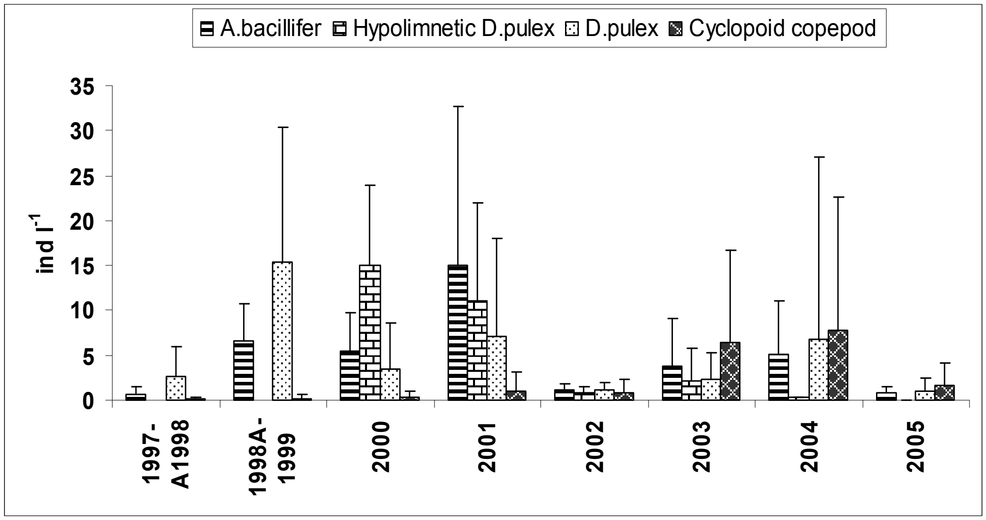

Figure 4.13. Changes in the density of A.bacillifer, Hypolimnetic D.pulex, D.pulex and Cyclopoid copepods in Lake Eymir from 1997 to 2005 in Lake Eymir.

4.1.9 Fish

There were increase in the biomass of both tench and carp (47 and 30 CPUE kg net^{-1}, respectively), in the gill nets ranging from 36 to 90 mm mesh sized nets in Lake Eymir in 2004. The total length of tench caught in the 36-90 mm mesh-sized nets largely varied from 27.8 ± 4 to 38 ± 4 cm and the mean total carp length varied from 24 ± 4.2 and 74.5 ± 9 cm. Pike biomass was 5 CPUE kg net^{-1}. The mean total pike length varied from 45 ± 10 to 65 ± 13 cm.

The young of the year (YOY) density was dominated by bleak (404 ±286 CPUE number net^{-1}) in 2004. The total length of bleak caught in the gill nets ranging from 7-16 mm mesh sizes varied from 6.8 ± 0.1 cm to 10.8 ± 2.6 cm.

During the fish stock estimation in 2005, there were increase in the biomass of both tench and carp (65 and 80 CPUE kg net^{-1}, respectively), in the gill nets ranging from 36 to 90 mm mesh. The total length of tench caught in the 36-90 mm mesh-sized nets largely varied from 35 ± 4 to 85 ± 5 cm and the mean total carp length varied from 37 ± 3 and 85 ± 8 cm. Pike biomass was 1 CPUE kg net^{-1}. The mean total pike length varied from 77 ± 11 to 90 ± 15 cm.

The young of the year (YOY) density was dominated by bleak (548 ±291 CPUE number net^{-1}) in 2005. The total length of bleak caught in the gill nets ranging from 7-16 mm mesh sizes varied from 8 ± 1.1 cm to 12 ± 13 cm.

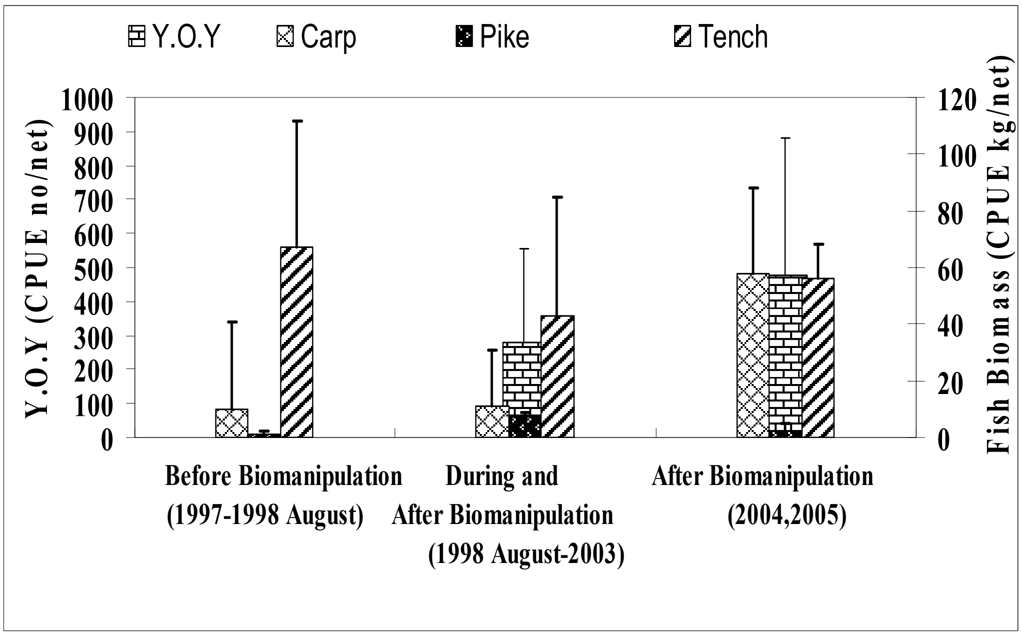

Figure 4.14. Mean number of young of the year fish (YOY) caught in the nets with mesh sizes from 7 to 25 mm and biomass of pike, carp and tench caught in the nets with mesh sizes from 36-90 mm across 1998-2005 in Lake Eymir.

4.1.10 Aquatic Plants

In the plant survey carried out in 2004, distribution of submerged plants was very limited covering less than 5% of the lake's total surface area with a low PVI (9.5%). Unlike the high submerged plant coverage and rich species diversity during the clearwater state, *Ceratophyllum demersum L* was the only species encountered during this survey. The outer depth of colonization was limited to relatively shallow areas (197 cm) (Table 4.3).

In the plant survey carried out in 2005, There was nearly no plant coverage and distribution of submerged plants was very limited covering only 1% of the Lake's total surface area with a lowest recorded PVI (1%) for the study period. Unlike the high submerged plant coverage and rich species diversity in clearwater state, *Ceratophyllum demersum L* was the only species encountered during the latest survey. The outer depth of colonization was higher than 2005 (295 cm) (Table 4.3).

Previous plant survey results and distrubition of aquatic plants between 1998 2005 can be seen from Table 4.3 and Figure 4.15

Table 4.3. Mean percent plant volume infested (PVI %), dry weight (DW: gm^{-2}), percent cover of the total lake area with submerged plants and the outer depth of colonization (cm) for the submerged plants sampled across 1998-2005 in Lake Eymir. Data for clearwater state shows the range of minimum and maximum values (c: Total cover (%), d: outer depth of colonization).

	1998		1999		2000		2001		2002		2003		2004		2005	
	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)	PVI (%)	DW (g/m^2)
P. pectinatus	*	-	*	-	74±15	468±75	55±18	278±35	35±32	136±21	50±13	211±98	-	-	-	-
Ceratophyllum sp.	*	-	*	-	76±22	601±88	37±28	234±201	41±28	332±86	26±27	201±35	9.5±2	159.2±358	3±1	124±51
Najas sp.	*	-	*	-	5±22	13±9	5±32	9±1	5±18	8±2	3±4	5±3	-	-	40	16.7
M. spicatum	*	-	*	-	78±21	34±13	7±2	19±5	4±3	11±3	-	-	-	-	-	-
Chara sp.		-	-	-	20±18	51±42	-	-	-	-	-	-	-	-	-	-
Overall PVI (%)	28±13		23±16		76±22	376±265	46±31	189±192	42±22	138±77	26±27	134±68	10±20	159±358	1±5	135±16
Cover (%)	2.5		6.2		50		90		60		45		5		0.8	
Outer depth of col. (cm)	175		250		560		390		405		350		197		295	

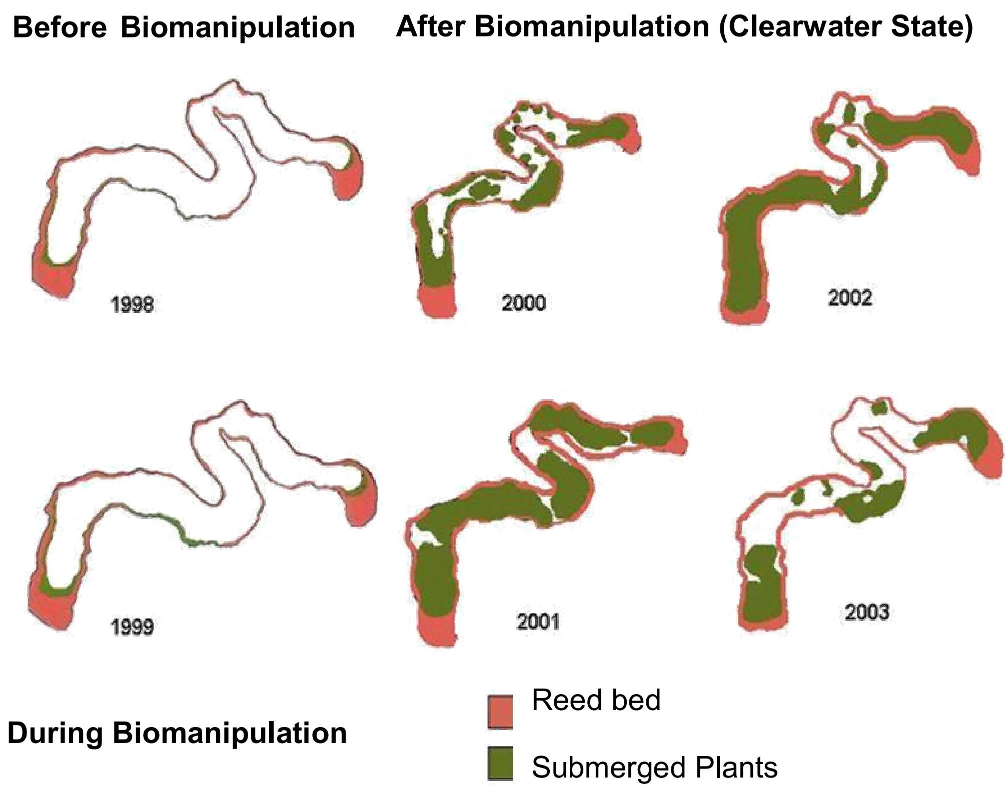

Figure 4.15. Distribution of aquatic vegetation in Lake Eymir. 1998-2003 (Taken from Tan, 2002).

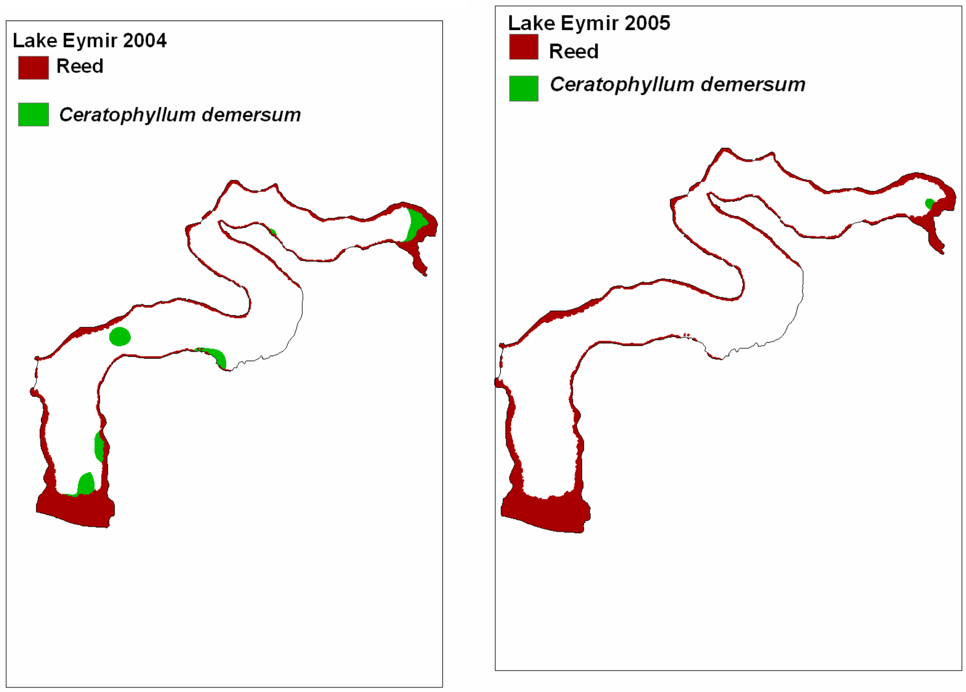

Figure 4.15 (continued). Distribution of aquatic vegetation in Lake Eymir during 2004 and 2005. Maps were prepared using ArcGIS Ver. 9.1 (Taken from Doğan Karabulut Ö).

4.2. Lake Mogan

4.2.1 Hydrology

Lake Mogan had an ice cover between January to March 2004. During this period, the ice cover was thick and stable. In 2005, the lake had an ice cover between January to February, but the ice cover was mostly at shallow shores and did not cover the whole lake.

The longest hydraulic residence time was recorded in 2005 (6.9 yr^{-1}) which differed significantly from the previous years. The shortest hydraulic residence time was 0.7 yr^{-1} recorded in 2000.

The water level fluctuations occurred on different time scales. A typical annual fluctuation was observed every year with increasing water level during the rainy periods which occurring in late winter and spring, and decreasing water level during the dry period in summer but also autumn and winter in some years. The mean amplitude of the annual water level fluctuations (maximum-minimum water levels) recorded from 1997 to 2005 was 0.72±0.25 m a.s.l.

The lake level fluctuated both annually and interannually throughout the study period (Figure 4.15). Z-scores were calculated to standardize the water level data. Firstly, Kruskal-Wallis ANOVA test revealed that the years were significantly different in terms of water levels (p: 0.000). Multiple pairwise comparisons revealed that the water levels in 2001 and 2005 were significantly lower than that of years 2000, 2002 and 2003 (Bonferroni corrected significance level: 0.0014). Multiple pairwise comparisons using the Dunn's procedure constructed three hydrologically different periods. Years were grouped in homogeneous subsets as low, regular and high water levels. 2001 and 2005 were referred to low water level (LWL) years whereas 2002 and 2003 were referred to high water level (HWL) years and the remaining study period was regarded as regular water level (RWL) years (Figure 4.15). On the other hand, if hydraulic residence time was also taken into account, year 2004 could be referred to dry year because it had the second highest hydraulic residence time (4.9 $year^{-1}$), throughout the study period.

However, the spring water levels fluctuated and spring water levels in 2001 and 2005 were significantly lower than other years (Bonferroni corrected significance level: 0,0014). Consequently, 2001 and 2005 were regarded as the driest year throughout the study period due to the lowest lake level and the highest residence time ever recorded. The same non-parametric methods were applied to summer water levels (p: 0.008). Results showed that years 2002 and 2003 were significantly differ than years 2000, 2001 and 2005 (Bonferroni corrected significance level: 0.0014).

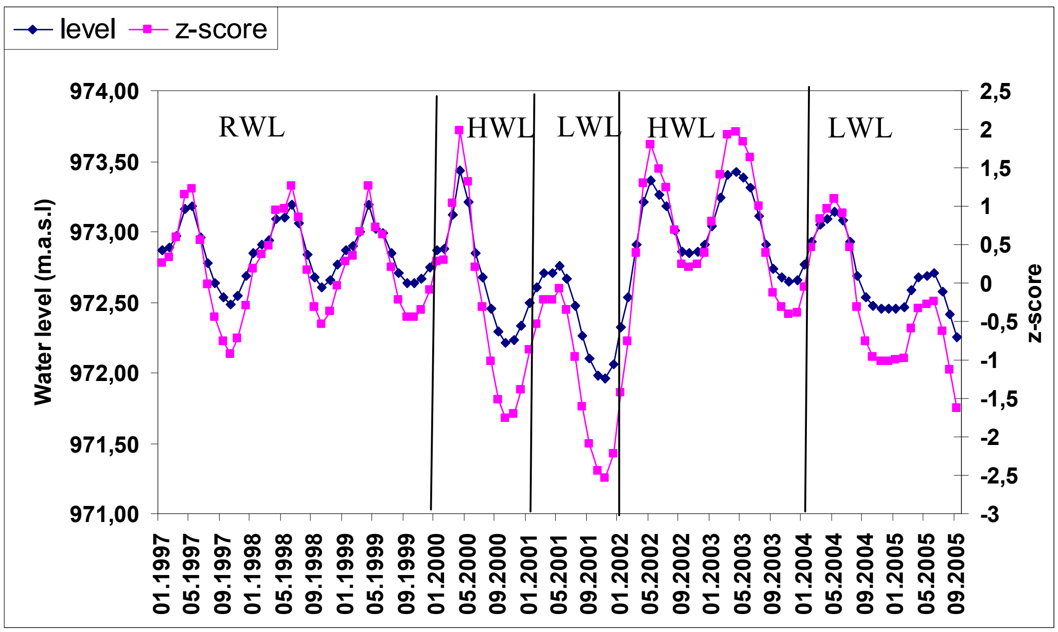

Figure 4.16. Water level fluctuations recorded from 1997 to 2004 in Lake Mogan. (RWL: regular water level, HWL: high water level, LWL: low water level).

4.2.2 Conductivity and Salinity

In Lake Mogan, the conductivity in 2004 and 2005 was significantly higher than previous years ($p<0.0001$, 2.90 and 3.56 $mScm^{-1}$, respectively, Table 4.4 & Figure 4.17). The salinity recorded in 2005 was significantly higher than those of previous years ($p<0.0001$, 1.85 ‰). Both the salinity and conductivity were inversely correlated with the lake levels (r:-0.29, p: 0.005 and r:-0.14, p: 0.04, respectively).

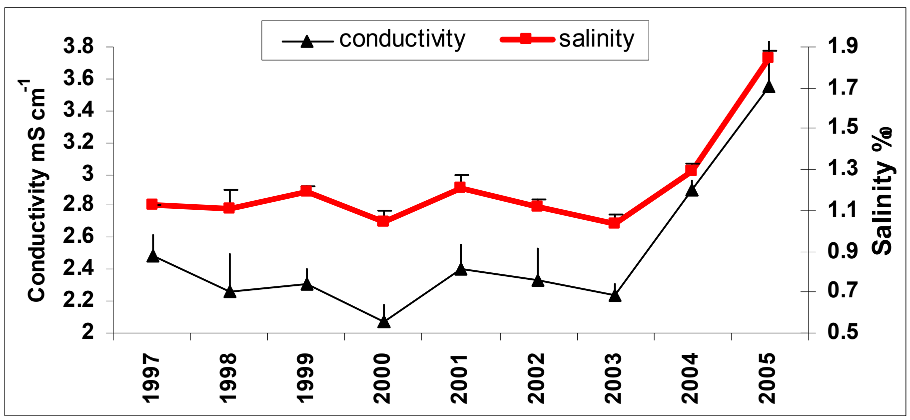

Figure 4. 17. Changes in the conductivity and salinity in Lake Mogan from 1997 to 2005.

4.2.3 pH

pH was significantly lower in 2003 and 2004 in Lake Mogan compared to previous period (p:0.000, 8.58 and 8.59, respectively). However, pH was inversely correlated with the water level (r: -0.68, p: 0.04).

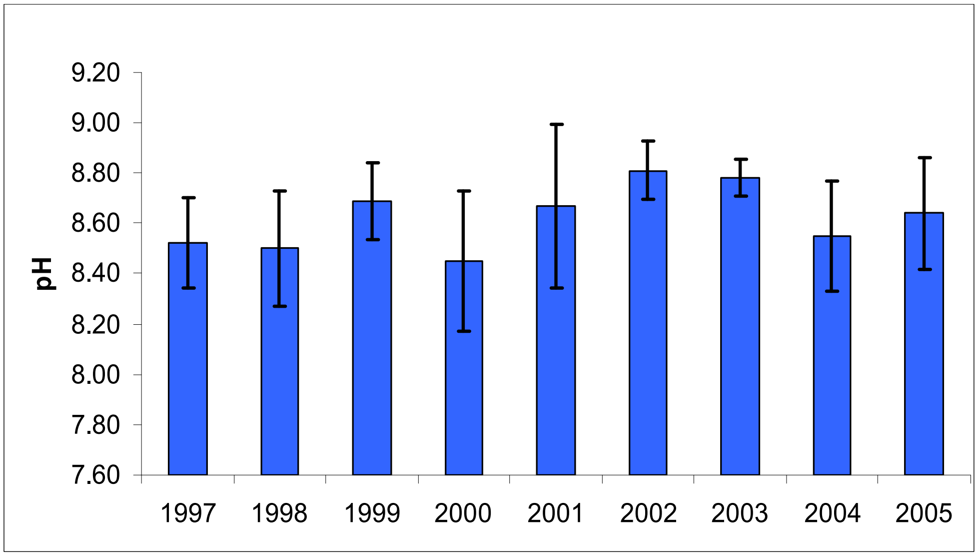

Figure 4.18. Changes in the pH in Lake Mogan from 1997 to 2005.

4.2.4 Temperature and Oxygen

Lake Mogan did not undergo thermal stratification during summer from 1997 to 2005. The summer mean water temperature in 2004 and 2005 was significantly higher than that of other years (p: 0.003, 21.5 and 22.5 ^{0}C, respectively, Figure 4.19). The annual mean water temperature did not significantly differ between years throughout the study period (16.5±2 ^{o}C) even though it was about 2^{o}C warmer in 2004 (18.5±2.2 ^{o}C, Figure 4.19).

The mean dissolved oxygen concentration was high and remained so throughout the study period (7.7±0.5 mg l^{-1}) and summer oxygen concentrations were not significantly differ from annual concentrations throughout the study period (Figure 4.20).

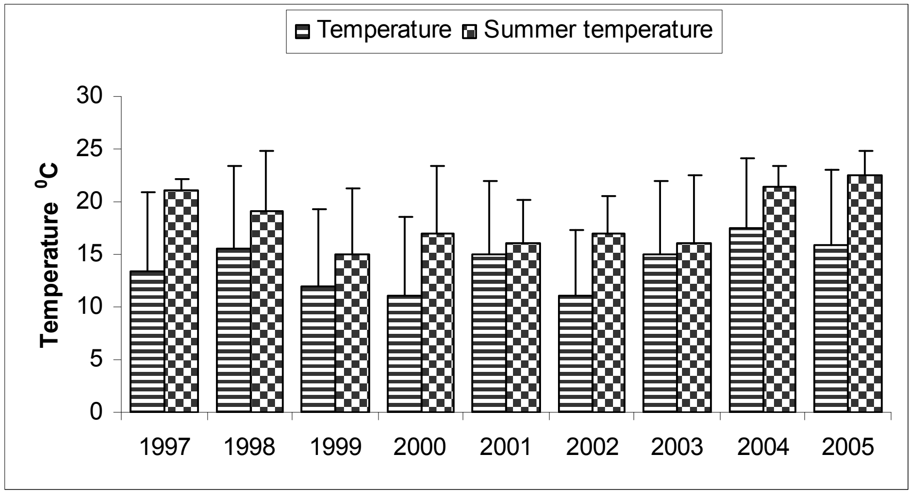

Figure 4.19. Annual and summer mean±SD of temperature from 1997 to 2005 in Lake Mogan.

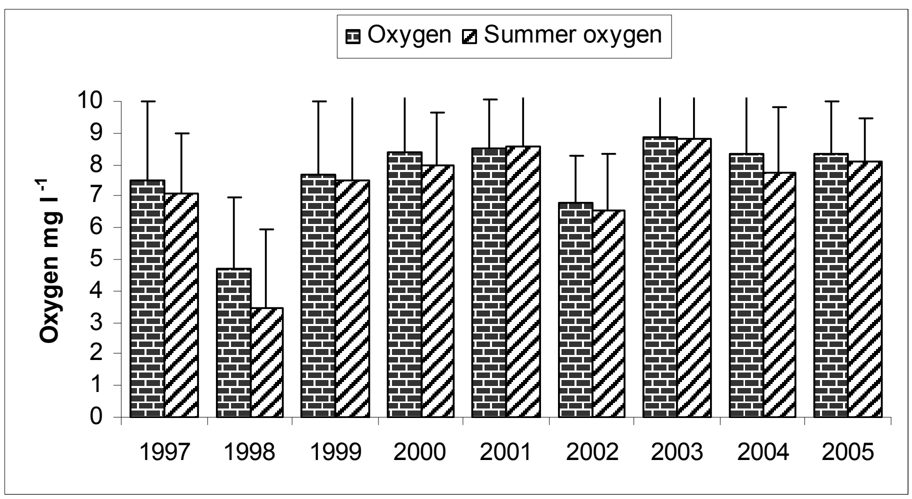

Figure 4.20. Annual and summer mean±SD of oxygen from 1997 to 2005 in Lake Mogan.

4.2.5 Secchi disk transparency, concentration of suspended solids and chlorophyll-a

In Lake Mogan, the concentrations of chlorophyll-a and suspended solids and the Secchi disk transparency changed significantly throughout the study period (p: 0.021, give SS and $p<0.0001$, respectively, Table 4.4 and Figure 4.21). Concentration of chlorophyll-a in 2000 and 2002 were significantly higher than other years (p: 0.021, 16.94 and 18.66 $\mu g\ l^{-1}$, respectively). Furthermore, the concentration of chlorophyll-a correlated inversely with the Secchi disk transparency (r:-0.49, p: 0.000). The concentrations of suspended solids (SS) were also significantly higher than previous years in both 2004 and 2005 ($p<0.0001$, 16.4 and 21 $mg\ l^{-1}$, respectively). In addition, the deteriorations in the Secchi disk transparency were more pronounced in 2004 and 2005 (110 cm and 77 cm, respectively) probably due to the effect of SS (Table 4.4 and Figure 4.21). Furthermore, the spring Secchi disk transparency also significantly changed during the study period (p: 0.001, Table 4.4). The highest spring Secchi disk transparency was observed in 2001 (312 cm) and lowest spring secchi was observed in 2005 (65 cm). In addition, the lowest spring chllorophyll-a concentration was observed in 2001 (1.6 $\mu g\ l^{-1}$) and highest spring chl-a concentration was observed in 2005 (18 $\mu g\ l^{-1}$) (Table 4.4).

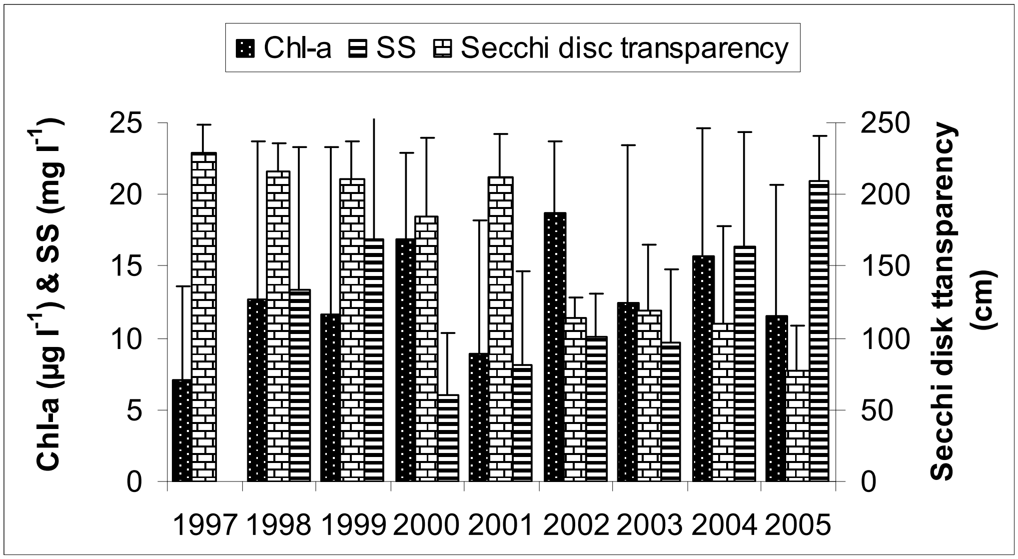

Figure 4.21. Changes in the concentrations of chlorophyll-a (mg l^{1}), suspended solids (SS) (mg l^{-1}) and Secchi disk transparency (cm) in Lake Mogan from 1997 to 2005.

4.2.6 Nutrients

The concentrations of total phosphorous (TP) and soluble reactive phosphorous (SRP) significantly differed between years; however, their mean concentrations always remained below 120 and 25 µgl^{-1}, respectively, throughout the study period (Figure 4.22 & Table 4.4).

TP concentrations significantly increased in 2003, 2004 and 2005 (89,120 and 86 µg l^{-1}, respectively), although annual TP areal loads remained lower than previous years (0.004 and 0.004 g m^{-2} yr^{-1} respectively) in 2004 and 2005. SRP concentration was also significantly higher in 2004 than other years (p: 0.000, 48 µg l^{-1}) (Figure 4.22).

Although DIN areal loads not significantly differ interannually, DIN concentrations were significantly higher than other years in 2003, 2004 and 2005 ($p<0.0001$, 0.67, 0.31 and 0.31 mg l^{-1}, respectively). However, concentrations of nitrate (NO_3-N) and ammonium were also higher in 2003, 2004 and 2005 (Table 4.4).

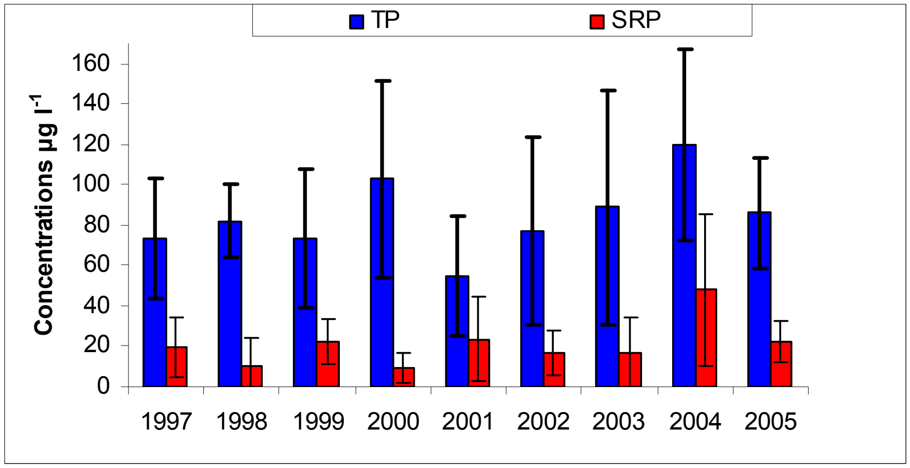

Figure 4.22. Changes in the concentrations of TP and SRP from 1997 to 2005 in Lake Mogan.

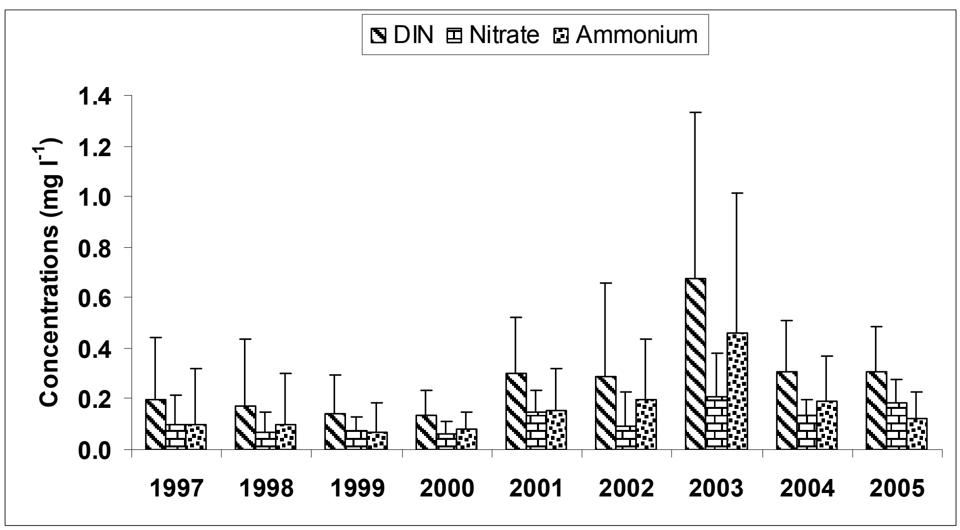

Figure 4.23. Changes in mean±SD of concentrations of DIN, Nitrate (NO_3-N) and Ammonium (NH_4-N) from 1997 to 2005 in Lake Mogan.

The concentrations of silicate in 2004 and 2005 were significantly higher than previous years ($p<0.0001$, 5.08 and 4.23 mg l^{-1}, respectively).

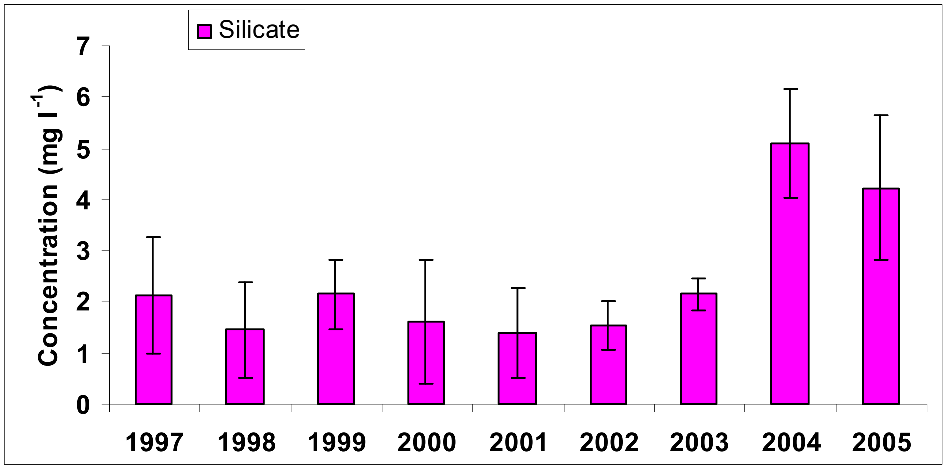

Figure 4.24. Changes mean±SD in the concentrations of silicate from 1997 to 2005 in Lake Mogan.

Table 4.4. Mean±SD of selected variables measured across 1997-2005 in Lake Mogan. Significance of differences was tested using Kruskal-Wallis ANOVA and multible comparisons using Dunn's procedure with Bonferroni's method. Water levels are given in metres above sea level. Spring lake level and spring Secchi depth included the measurements of the variable from March to June. The annual areal TP and DIN loadings are estimated as the sum of the loads of the inflows per unit of lake area given in g m^{-2} yr^{-1}. Algal volumes are given in million of μm^3 ml^{-1} and zooplankton in numbers per litre (ind.l^{-1}). Remaining variables are given as annual mean±SD.

	1997	1998	1999	2000	2001	2002	2003	2004	2005	Kruskal-Wallis	Dunn's procedure with the Bonferroni type adjustments
Annual lake level (m a.s.l)	972.81 ± 0.24	972.89 ± 0.19	972.85 ± 0.18	972.71 ± 0.41	972.40 ± 0.31	972.94 ±0.30	973.05 ±0.31	972.80 ± 0.27	972.53 ±0.15	p:0.000	01 and 05*all years
Spr. lake level (m a.s.l)	973.01 ±0.11	973.04 ± 0.10	973.04 ± 0.10	973.24 ± 0.31	972.70±0.04	973.16±0.23	973.40±0.02	973.10±0.04	972.65±0.05	p:0.021	05*97,00,02,03; 01*00,03
Secchi Depth (cm)	229±100	216±53	211±75	184±54	212±86	113±15	119±46	110±68	77±32	p< 0,0001	02,03,04 and 05*97,98,99,00,01
Spring Secchi depth (cm)	164±126	210±14	243±12	155±30	312±40	116±70	125±50	91±20	65±18	p:0.001	01*97,02,03,04,05;05*98,99,00,01
Chlorophyll-a ($\mu g\ l^{-1}$)	7.13±6.53	12.71±11.01	11.70±11.64	16.94±9.12	8.94±9.25	18.66±6.91	12.41±13.23	15.67±21.32	11.56±9.18	p:0.021	00*97,01;02*97,99,01,03,04,05
Spring Chlorophyll-a ($\mu g\ l^{-1}$)	7.7±3.2	10±3	5±0.4	14±5	1.6±2.1	12±4.9	8.8±9.3	7.6±8.2	18±9.6	p:0.02	05*99,01;01*98,01,02,05
SS (mg l^{-1})		13.35±0.21	16.88±12.81	6.05±4.23	18.06±6.61	10.04±3.05	9.62±5.19	16.40±10.17	21±9.54	p< 0,0001	04,05*99,00,03;99*00;02*05
Conductivity (mS cm^{-1})	2.48±0.14	2.25±0.25	2.30±0.10	2.06±0.12	2.40±0.15	2.33±0.21	2.24±0.06	2.90±0.33	3.56±0.44	p< 0,0001	04 and 05*all years;00*all years
Salinity (‰)	1.12±0.01	1.11±0,10	1.19±0.04	1.05±0.05	1.21±0.06	1.12±0.04	1.03±0.05	1.30±0.34	1.85±0.23	p< 0,0001	05*allyears;00 and 03*all years, 01 and 04*00,02,03,05
TP ($\mu g\ l^{-1}$)	73.46±29.58	81.88±18.12	73.38±33.9	102.65±48.685	54.44±29.68	77.14±46.74	88.78±57.96	120.12±47.25	85.93±27.56	p: 0,000	01*98,00,03,04,05; 04*97,99,01,02,03
SRP ($\mu g\ l^{-1}$)	19.52±14.97	10.58±13.22	21.92±11.18	9.24±7.28	23.65±20.51	16.65±11.45	17.01±17.51	47.85±37.40	22.44±10.52	p: 0,000	04*97,98,00,01,02,03; 98 and 00*99,01,04,05
DIN (mg l^{-1})	0.197±0.243	0.169±0.266	0.140±0.154	0.136±0.100	0.303±0.218	0.289±0.370	0.674±0.658	0.304±0.206	0.307±0.176	p< 0,0001	97,98,99 and 00*01,03,04,05; 03,04 and 05*97,98,99,00
NO_3-N (mg l^{-1})	0.10±0.12	0.07±0.07	0.07±0.05	0.06±0.05	0.15±0.09	0.10±0.13	0.21±0.17	0.14±0.06	0.19±0.10	p: 0,000	01,03,04,05*97,98,99,00,02
NH_4-N (mg l^{-1})	0.10±0.22	0.10±0.20	0.07±0.07	0.07±0.01	0.16±0.16	0.20±0.24	0.46±0.55	0.19±0.18	0.12±0.10	p: 0,000	97,98,99*00, 02,03,04;03*97,98,99,00,01,05

Table 4.4 (continued). Mean±SD of selected variables measured across 1997-2005 in Lake Mogan. Significance of differences was tested using Kruskal-Wallis ANOVA and multible comparisons using Dunn's procedure with Bonferroni's method. Water levels are given in metres above sea level. Spring lake level and spring Secchi depth included the measurements of the variable from March to June. The annual areal TP and DIN loadings are estimated as the sum of the loads of the inflows per unit of lake area given in g m^{-2} yr^{-1}. Algal volumes are given in million of μm^3 ml^{-1} and zooplankton in numbers per litre ($ind.l^{-1}$). Remaining variables are given as annual mean±SD.

	1997	1998	1999	2000	2001	2002	2003	2004	2005	Kruskal-Wallis	Dunn's procedure with the Bonferroni type adjustments
Diatoms	0.06±0.06	0.02±0.04	0.27±0.41	0.24±0.37	0.26±0.47	0.18±0.23	0.16±0.22	0.04±0.09	0.01±0.01	p:0.008	05*99,00,01,02,03;00*04,05
Chlorophyta	0.17±0.27	0.35±0.46	0.30±0.18	0.26±0.29	0.16±0.16	0.45±0.69	0.33±0.32	0.07±0.09	0.13±0.17	p:0.005	04*98,99,00,02,03;05*99,02
Cyanobacteria	1.5±3.7	2.3±4.5	1.6±4.1	0.7±1.6	1.5±3.4	2.6±3.7	1.5±25	0.8±1.2	1±1	p:0.027	97,99*02,03,05
D.pulex	1±1	1±1	2±3	1±1	1±1	1±1	1±1	2±1	1±1	p:0.026	03,05*97,99;99*02,03,05
A.bacillifer	5±5	14±20	10±17	10±18	12±27	3±4	1±1	6±14	2±2	p<0,0001	03*all years;01*97,98,99,03

4.2.7 Phytoplankton

In Lake Mogan, phytoplankton community included species of cyanobacteria, dinoflagellates, Chlorophyta, diatoms and Cryptophyta from 1997 to 2005. In both 2004 and 2005, cyanobacteria dominated the phytoplankton community ($0.8x10^6$ μm^3 ml^{-1} and $1.1x10^6$ μm^3 ml^{-1}, respectively). Diatoms and chlorophyta biovolumes significantly decreased in 2004 and 2005 (p: 0.005, 0.07 and $0.131x10^6$ μm^3 ml^{-1} and p: 0.008, 0.04 and $0.011x10^6$ μm^3 ml^{-1}, respectively).

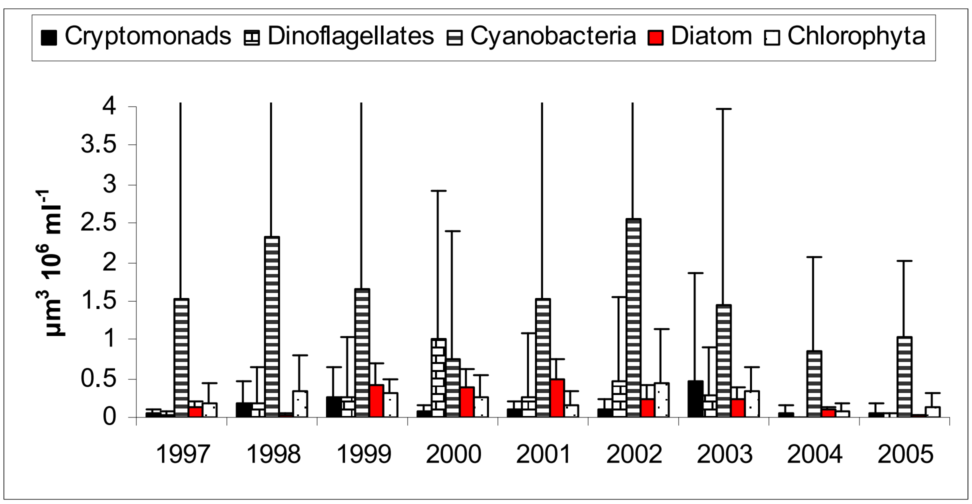

Figure 4.25. Changes in the biovolume of phytoplankton taxa from 1997 to 2005 in Lake Mogan

4.2.8 Zooplankton community

In Lake Mogan, the density of zooplankton community was very low and largely dominated by *A. bacillifer* (Table 4.4, Figure 4.26) from 1997 to 2003. *Daphnia pulex*, was recorded; its abundances being, however, low and with significant interannual differences (Table 4.4).

D.pulex density was higher in both 1999 and 2004 (2 ınd l^{-1}). As for the large-bodied species, *A. bacillifer* and *D. pulex*, only *A. bacillifer* density was noteworthy (Table 4.4 and Figure 4.26). The contribution of rotifera to zooplankton abundance was high in both 2004 and 2005 and they become dominant group (112 and 238 ind l^{-1}, respectively).

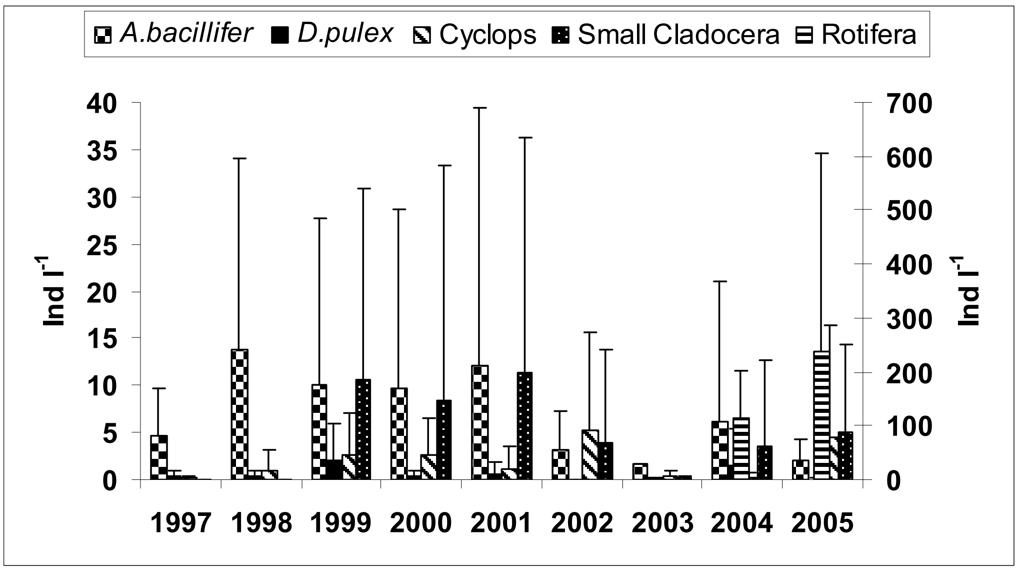

Figure 4.26. Changes in the density of *A.bacillifer, D.pulex* and Cyclopoid copepods in Lake Eymir from 1997 to 2005 in Lake Mogan.

4.2.9. Aquatic Plants

In the plant survey carried out in 2004, distribution of submerged plants was covering about 24% of the Lake's total surface area with an average PVI of 21.2%. *P. pectinatus, Chara sp., Najas spp.* were recorded as submerged plant species (Table 4.5).

In the plant survey carried out in 2005, plant cover was higher than 2004 (35% with a PVI 31%). Same species was observed exept *Chara sp.* (Table 4.5).

Table 4.5. Mean±SD of plant volume infested (PVI %), dry weight (DW: gm^{-2}), percent of the total lake area with submerged plants sampled across 2000-2005 in Lake Mogan.

	2001	**2002**	**2003**	**2004**	**2005**
Cover (%)	90	19	30	24	35
PVI (%)	56±26	42±26	52±23	21±29	31±38
Dry weight (g/m^{-2})	155±215	74±173	144±176	151,5±96	260.4±96

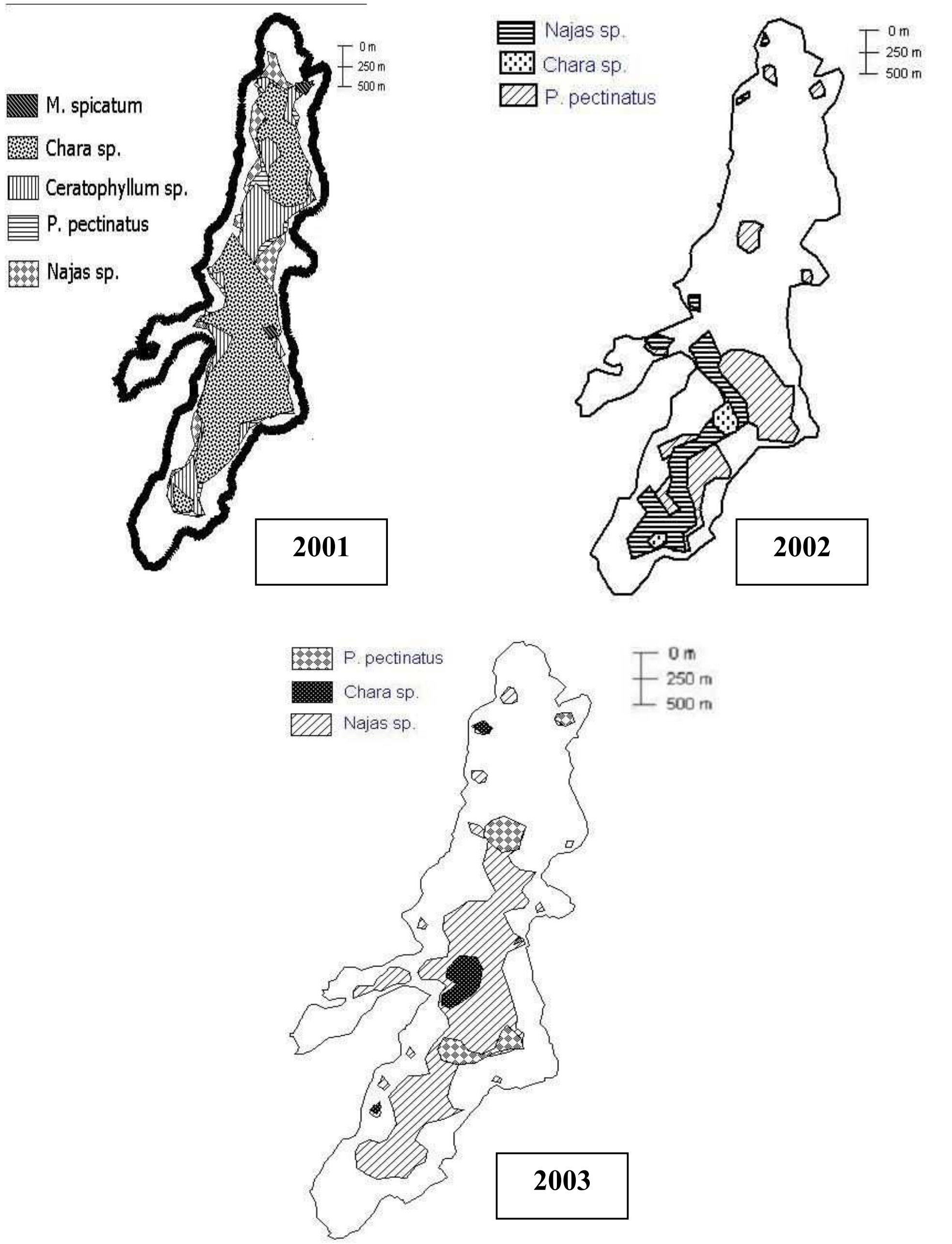

Figure 4.27. Distribution of aquatic vegetation in Lake Mogan. 1998-2003 maps are taken from Tan, 2002

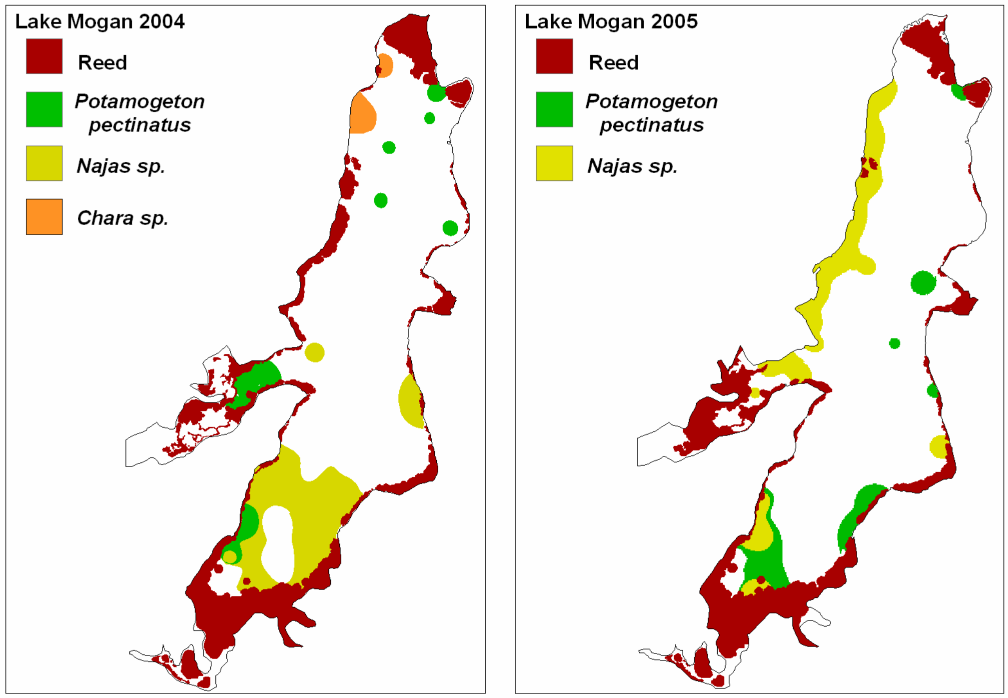

Figure 4.27 (continued). Distribution of aquatic vegetation in Lake Mogan. 2004 and 2005 maps were prepared using ArcGIS Ver. 9.1 (Doğan Karabulut Ö, unpublished data).

5. DISCUSSION

5.1 Lake Eymir

Ecological changes in lakes should be evaluated in relation with hydrology, nutrient availability, especially external loading, and trophic state. The present study provides a distinct opportunity to take a step forward to a better understanding of the nutrient dynamics of an eutrophic shallow lake located in semi arid Mediterrenean and it was restored by biomanipulation which has been the first biomanipulation in Turkey and also in the region. Furthermore, marked hydrological changes which are inheret feature of the region significantly alter outcome of the restoration project.

Lake Eymir was in turbid water state in 1997 with very low Secchi disk transparency owing to the high especially SS and chlorophyll-a concentrations. The fish stock was dominated by planktivorous tench (*Tinca tinca L.*) and the benthivorous common carp (*Cyprinus carpio L.*). Due the turbid water condition, submerged plant cover was poor (Beklioğlu et al., 2000; Beklioglu et al., 2003).

Restoration of Lake Eymir by fish manipulation was performed by removal of benthi-planktivorous fish between 1998 and 1999. The fish removal led to significantly improved light conditions. This improvement in the light climate was mainly attributed to the decrease in the concentrations of suspended solids and chlorophyll-a following the fish removal (Beklioglu et al., 2003). The initial increase of transparency after biomanipulation is typically followed in shallow lakes by a strong development of submerged vegatation in the following years and Lake Eymir also followed the same pattern (Beklioglu et al., 2003; Scheffer, 1998; Drenner & Hambright 1999).

In clear water state between 2000 and 2003, submerged plant cover was between 40% to 90%. with a high PVI %, especially for *Potamogeton pectinatus* (ranging from 35 to 74 PVI %) *L.* and *Ceratophyllum demersum L* (ranging from 26 to 76 PVI %).

Since biomanipulation experiments offer an outstanding opportunity to study the response of the shallow lake community to perturbations (Scheffer & Carpenter, 2003, Ameniya et al., 2005), the changes in the community of fish, zooplankton and phytoplankton were observed after biomanipulation. After biomanipulation, the biomass of planktivorous tench (*Tinca tinca L.*) and the benthivorous common carp (*Cyprinus carpio L.*) decreased compared to pre-biomanipulation and the biomass of pike *(Esox lucius L.)* increased. Due to decreasing grazing pressure on large bodied zooplankton *D.pulex* and *A.bacillifer* become dominant during the clear-water state. Phytoplankton community composition did not change after biomanipulation. Chlorophyta was dominant phytoplankton taxa during this period. Since Chlorophyta was edible phytoplankton the increasing grazing pressure of *D. pulex* may supressed chlorophyta (Beklioğlu & Tan, accepted). However, in 2003, several features started to deteriorate, with decrease in especially summer Secchi depth, increase concentrations of chlorophyll-a and suspended solids. However, the lake retained its macrophyte dominated state probably owing to high spring clear water. Whereas in 2004, submerged plants were completely lost due to low Secchi disk transparency especially in spring High spring chlorophyll-a and suspended solids concentration led to decrease in water transparency. As a consequence of this, submerged plants become very limited covering less than 5% of the Lake's total surface area with a low PVI (9.5%). In Lake Eymir the recorded submerged plant was *Ceratophyllum demersum* L whish is a typical survivor growing in eutrophic conditions as it was observed elsewhere (James et al., 2005). Ecological state of the lake was similar to that of recorded before biomanipulation (Beklioglu et. al., 2003). In turbid water state before biomanipulation, the poor water conditions were due to the high concentration of SS and whereas in 2004 the poor water condition were largely by high chlorophyll-a concentration. The former was due to especially carp stiring up the sediment while feeding. During latter period, chlorophyll-a concentration increased in the lake.

This was probably due to the lake being released from the former N-limited state resulting from the latter increase in ammonium concentration. In both turbid water state, spring Secchi disk transparency was low compared to clearwater state. Because of this, submerged plant was almost lost in both turbid water state. We conclude that the clear water phase in spring was crucial leading to increase submerged macrophyte and this is in concert with the Meijer et al. (1999) and Van Nes et al. (2002).

Meijer et al., (1990) reported from several Ducth shallow lakes that vegetation loss was due to benthivorous fish feeding induced turbidity. In Lake Eymir, fish stock was dominated by planktivorous fish tench and benthivorous fish carp in the turbid water period in 1997-1998 and 2004-2005. During the clearwater state initiated by about 50% removal of existing carp and tench stock, biomass of tench and carp were lower than both of the turbid states. Pike biomass was higher from 1999 to 2003 than both of the turbid states. Increase in the tench and carp not only increased suppression of *Daphnia* by predation pressure but it also triggered a number of mechanisms by which they contributed to the turbid water state including resuspension of sediment, and nutrient pomping (Breukelaar et al., 1994; Parkos et al., 2003). Furthermore, increase in SS amount and nutrients concentrations in 2004 and 2005, supported this idea. These results also support the observation that lakes, which were initially biomanipulated successfully but reverted to pre-implementation conditions, were often lakes whose fish population returned to pre-implementation levels (Drenner & Hambright 1999). Our results indicates that top-down control of fish on zooplankton and nutrients were important since zooplankton grazing effect on phytoplankton decreased with high predation pressure during turbid water state after biomanipulation.

Water level fluctuation has been identified as one of the mechanisms triggering a shift to a submerged macrophyte dominated state in shallow lakes owing to an improved underwater light climate over the lake bottom (Blindow 1992; Coops et al. 2003).

Recent studies by Beklioglu et al. (2006a) and Tan & Beklioglu (2006) identified water level change as a one of the key mechanisms for submerged plant development in five Turkish shallow lakes; though the low water level condition also increased the concentrations of major ions and nutrients. Our results implicate that the effect of drop in Lake water level may be different in clear and turbid water state in semi-arid region. Lake level drop in 2001 led to significant changes in the lake. Submerged plant cover expanded to 90% of lake total surface area. This result is in concert with with the idea that low water levels may enhance submerged plant coverage in shallow Mediterrenean lakes (Romo et al., 2004; Beklioglu et al., 2006). Whereas in 2004, when the water level was significantly low, and it was as low as the level recorded in 2001, but we could not observed the same vegatation expansion in 2004. On contrary, submerged plants completely dissaperared and the lake shifted to turbid water state. This may be related with decrease in water clarity, especially spring water clarity resulted from the increased chlorophyll-a and suspended matter concentrations. Increase in the biomass of benthi-planktivorous fish may have partly initiated this. On the other hand late increase in the concentrations of DIN largely by ammonium most probably relaeased the lake form the formerly nitrogen-limited state since TP level in the lake was already supressed the critical TP levels suggested for stable long lasting recovery (Jeppesen et al., 1990, 2003). Combination of top-down and bottom up factors working together stabilized the turbid water conditions in the lake. Since nitrogen limited conditions was disappeared from the lake in 2004 and 2005, recovery of submerged plant did not occured as it was observed in clearwater state in high concentrations of TP. Most restorations of shallow lakes by phosphprous control and biomanipulation have produced temporarily clearwater and a macrophyte community of modest diversity. These have proved unstable and there has often been a shift to turbid water state (James et al., 2005). However, nitrogen may play a far more important role than previously appreciated in the loss of submerged macrophytes at relatively high phosphorus levels (Moss 2001; James et al., 2005; Sagrario et al., 2005).

Biomanipulation was undertaken successfully in Lake Eymir but concentrations of DIN, ammonium and nitrate were respectively higher during the clearwater state and turbid water state after biomanipulation. However, DIN concentration significantly increased largely by ammonium which might have resulted from slowed denitrification processes at the low availability of dissolved oxygen concentration during turbid water state in 2004 and 2005. The instability of Lake Eymir may be related with this high concentrations and It may be concluded that high nitrogen may be important in the loss of submerged macrophytes during the course of eutrophication in Lake Eymir.

Thus, N-control might be a new challenge in long-term stable lake restoration with rich submerged plant diversity (Gonzales Sagrario et al. 2005; James et al. 2005).

Concentrations of nutrients (nitrogen and phosphorus) vary with water levels (Nöges et al., 1999; Talling, 2001) and significantly increase during drought periods in the Mediterranean region (Hambright et al., 2000; Tan, 2002; Tan & Beklioglu, 2005). Drought-induced decrease in water level and increase in water residence time may provide longer contact with sediment that may enhance internal release of nutrients, such as phosphorus (Romo et al., 2005; Karapınar & Beklioglu, 2006). Our results are in accordance with these results. Although there was no external loading in 2004 and 2005, TP concentration increased with high hydraulic residence time. Thus, in shallow warm lakes, global warming may require an establishment of even lower thresholds not only for P but also for also N loadings than those so far suggested to obtain a stable submerged plant dominated clearwater state (Romo et al. 2004, Karapinar & Beklioglu, 2006).

The dry conditions in 2001, 2004 and 2005 with the low lake level, higher retention time caused an increase in the salinity and conductivity to the highest values. Due to relatively higher water levels in 2002 and 2003, salinity and conductivity decreased to their regular values.

Although drop in water level may enhance submerged vegetation in temperate lake due to less significant effects on water chemistry, the affect of water level drop in semi-arid regions may be negative due to increase salinity and nutrients and trophic state of lake. Hydrological extremes (floods and dry periods) are predicted to follow from the global climatic changes. Semi-arid or arid Mediterranean lakes are predicted to receive lower input of water due to shorter precipitation seasons coupled with higher incidence of summer dry periods (Sanchez et al. 2001). Extrapolation of these results implies that if such conditions occur, many freshwater lakes located in the Mediterranean region would turn saline in the future. Evidently, in Turkish lakes changes in salinity level have frequently triggered shifts between saline and freshwater conditions (Karabiyikoglu et al. 1999).

Due to increasing turbidity and nutrients and lost of submerged plants buffer mechanisms, cyanobacteria bloom occurred in lake. However, drop in water level and high hyraulic residence time favoured cyanobacteria over other phytoplankton taxa and this is in concert with other observations in Mediterranean shallow lakes (Romero et al., 2002; Romo et al., 2004). The zooplankton community of Lake Eymir was species-poor in 2004. The density of the large-sized grazers was significantly low (especially the density of Calanoid Copepod *A. bacillifer* and *D.pulex*). The decline in daphnids may be related with increasing fish predation, loss of submerged vegetation refuge mechanism and increased turbid conditions which prevented them from grazing efficiently. However, increase in cyanobacteria and decrease in edible algae may affect them, negatively. A.bacillifer decreased due to increasing turbidity may prevent them from fish grazing. In 2005, Rotifers dominated the zooplankton community due to increasing eutrophication and decrease of large bodied zooplankton. Turbid conditions favoured rotifers since they were better competetors and had high reproduction rate than crustaceans. However, decrease in crustaceans was another reason for increasing rotifers since they had a competition with crustaceans for food (Kirk, 1991).

The average success rate of food-web manipulations is about 60% (Hansson et al., 1998; Drenner & Hambright, 1999). General criteria for a successful biomanipulation mainly would include decrease in algal turbidity, concentrations of total phosphorous and chlorophyll-a and plankti-benthivorous fish abundance, increase in Secchi depth transparency and macrophyte cover. Biomanipulation may produce positive results early in the study and can be categorized as successfull, given to at least 4 to 5 year time span after the intervention (Hannson et al., 1998; Meijer et al., 1999). Lake Eymir met the criteria and results in Lake Eymir suggest that biomanipulation was a very effective measure in arid climates as it observed temperate lakes (Beklioğlu & Tan accepted). In many cases, lakes returned to turbid state as it was observed in Lake Eymir (Meijer et al., 1999). However, processes, which were responsible for shift, were relatively same such as reestablishment of dense populations of planktivorous or benthivorous fish (Kasprzak et al., 2002), increased in benthi-planktivorous fish ratio, increase resuspension and loss of submerged plants, nutrient increases (Hansson et al. 1998; Meijer et al. 1999; Gulati & Van Donk 2002).

Recently Beklioglu & Tan (2006) contradicted the argument put forward by Jeppesen et al. (2005) argued that fish stock manipulations probably would not have the same positive effect on warm temperate and subtropical lakes as in north temperate lakes due to their extremely rich fish fauna characterised by omnivorous feeding habits and high densities resulting from the creation of several cohorts per year and lack of strong predators (Branco et al. 1997, Lazzaro 1997, Meerhoff et al. 2003). Furthermore, Blanco et al. (2003) reported dominance of omnivorous fish communities with lack of strong predators in Spanish Mediterranean lakes resembling tropical lakes. Consequently, it may be hypothesised that biomanipulation may not be as effective a restoration method in tropical and Mediterranean lakes. However, the response of Lake Eymir to a ca. 50% removal of carp and tench contradicts this anticipation since fish removal improved the trophic conditions for pike growth and biomass in the lake and generated a shift in water transparency sufficient for macrophytes to develop with a coverage of 40-90% of the total surface area.

This could be due to the fact that trophic structure varies among warm lakes, the distribution and composition of the fish community depending, for instance, on altitude. This difference may account for variations in the outcome of biomanipulation owing to the strong impact of fish composition and diet (Moss et al. 1996; Jeppesen et al. 2005). A comparison among low altitude lakes in Greece and Spain and high altitude lakes in Turkey, all located at 39^{o} N latitude, denoted a lack of piscivorous fishes in the former with communities mostly composed of omnivores, whereas in high altitude Turkish shallow lakes piscivorous (e.g. pike, perch) and omnivorous species co-existed (Blanco et al. 2003; Burnak & Beklioglu 1999; Beklioglu et al. 2003). The differences in fish community composition were partly due to the effect of winter ice-cover and the restricted biogeographical distribution of fish at higher altitudes, but may also be explained by other factors, such as water temperature and water visibility, facilitating predation by visual predators (Vijverberg et al. 1990; Liao et al. 2004). Thus, not only latitude, but also altitude, should be included in our perspective for understanding the trophic structure of shallow lakes.

In conclusion, the long-term stability of biomanipulated restoration is very poorly elucidated, locally and internationally. One of the future challenges within this field is to determine the stability of clearwater state, taking into account fish recruitment and growth of submerged macrophye mediated by variations in climate, water level fluctuations and role of nitrogen. Lake Eymir may present a case between temperate and subtropical lakes and may contribute the understand of governing mechanisms for alternative stable states in eutrophic lakes located in semi arid Mediterrenean climatic region which are prone to drought due to global warming.

5.1 Lake Mogan

Functioning of shallow lakes located in regions ranging from north temperate to semi-arid and arid zones can be sensitive to hydrology and water level fluctuations (WLF), which emerges a decisive element of hydrology (Blindow, 1992; Blindow et al., 1997; Gafny & Gasith, 1999;, Coops et al., 2004; Beklioglu, et al., 2006).

Extend of WLF in effecting submerged plant development largely depends on the periodicity of the fluctuations, including the range, frequency and regularity of WLF (Riis & Hawes, 2002) and depth profile of a lake (Beklioğlu et al., 2006). In shallow lakes, strong WLF can be disadvantageous to many submerged plants through either reduced light availability or drying out/freezing of shallow lakes, which may exceed the physiological limits of many submerged plant species (Hill & Keddy, 1992, Coops et al., 2004).

In Lake Mogan, the clearwater state with submerged plant dominated appeared to be very sensitive to the water level fluctuations. In 2001, the lake shifted to the dense submerged plant dominated state in connection with a 40 cm drop in the spring water level followed by a 145 cm increase in the corresponding Secchi disk transparency compared to 2000. The lake was in turbid water state with very low vegetation cover in 2000. The vegetation shift also lowered phosphorus amount by reduced resuspension and the phosphorus release from the sediment (Granéli & Solander, 1988) and the concentration of chlorophyll-a and TP nearly decreased two fold compared to 2000 (8.94 and 54.44 $\mu g\ l^{-1}$, respectively). The total surface coverage of the submerged plants was high (77.5%) macrophytes being dominated by *P. pectinatus* and *Chara sp.* Additionally, a recent review by Beklioglu et al. (2006) on Turkish shallow lakes located in semi-arid to arid regions indicated that the water level fluctuations were the most decisive factor for a shift to the vegetated state. Beklioglu et al. (2006) found that Beyşehir, Marmara and Uluabat Lakes shifted to a clearwater state dominated by submerged plants with the submerged plant coverage of 50% to 100% of the lake total surface areas after the drops of the spring water levels. Furthermore, the state changes recorded in these lakes occurred also independent of the changes in the availability of nutrients (Beklioglu et al., 2006).

In 2005, a 45 cm drop in spring lake water level resulted in an increase in plant cover compared to 2004 (24%, in 2004, 35%, in 2005). The water level decline did not lead to shift to high submerged plant cover as it was observed in 2001. The possible reasons for that is low spring Secchi disk transparency (77 cm) in 2005 due to high spring chl-a and suspended solids concentrations (18 μg l^{-1} and 21 mg l^{-1}, respectively). However, these results are in accordance with Lake Eymir where submerged plant did not grow in low spring water level due to high spring chl-a and low spring Secchi Disk in 2004.

In years 2000, 2002 and 2004, the spring lake level was very high and there were significant deteriorations in the Secchi disk transparency and the chlorophyll-a concentration due to the relatively low submerged plant development. The reasons for the decreased submerged plant biomass were probably the high spring water level and cooler water temperature in spring. The cooler spring water temperature recorded with high spring water level in 2000, 2002 and 2004 (16, 17 and 16 ^{o}C, respectively) and low submerged plant cover recorded in these years. Water temperature is an important physical factor in controlling aquatic plant development. Thus, an early warming of water temperature significantly enhanced submerged plant growth, and overriding the effect of underwater irradiance compared to the later warmer years (Rooney and Kalff, 2000). It can be suggested that in Lake Mogan the combination of factors including the high spring water level, the cooler water temperature and the decreased Secchi disk transparency might have suppressed submerged plants development.

Blindow et al. (1997) claimed that the same factors affected the submerged vegetation and the observed switch to turbid water in Lake Takern.

In arid tropical shallow lakes, sensitivity to hydrological conditions also has dramatic costs especially for salinity (Talling, 2001). In Lake Mogan, the salinity was high throughout the study especially in 2001, 2004 and 2005 which were the low water level years. Our results indicate that there was a inverse relationship between lake water level and conductivity and salinity.

Water level also influences nutrient dynamics. In Lake Mogan, inflows were the major sources of nutrients in the high water level and regular water level years while the importance of inflows decreased significantly in the low water level years. Significant increase in hydraulic residence time (3 yr^{-1}) took place in 2001 due to low inflows. However, second longest hydraulic residence time was recorded in 2004 because the water level was kept higher by closing sluice gate for maintaining the lake volume high not to allow the reduction in lake level despite of low inflows. The hyraulic residence time was highest in 2005 (6.8 yr^{-1}). In 2000, in Lake Mogan, TP amount in the lake was higher than previous years (1997 to 1999). Because TP load via inflows was significantly high and the lake was lack of submerged vegetation due to high spring level in 2000. However, TP amount in the lake declined 2-fold in 2001 when the water level was low owing to low TP loading from the catchment via the inflows. Moreover, the low spring water level increased the submerged vegetation cover (90% surface coverage). In 2005, low water level resulted a 1.5 fold decline in TP concentrations compared to 2004. Increase in submerged plants also lowered phosphorus amount by reduced resuspension and the phosphorus release from the sediment (Granéli & Solander, 1988). This is in accordance with Lake Võrtsjärv in Estonia that precipitation-poor period resulted in decline in external nutrient loading and lower water level year yielded better light conditions causing increase in growth of the submerged macrophyte (Nõges & Nõges, 1999). Furthermore, drops in spring lake level coupled with high evaporation in summer enhanced vegetation development leading to a large expansion of the littoral zone in Mediterranean and subtropical lakes (Gaffny & Gasith, 1999; Beklioglu et al., 2006; Havens et al., 2004). Furthermore, in 2004, the TP amount in Lake Mogan reached to a highest amount despite the lowest external TP load of the study period was recorded in this year. Possible explanation for high in-lake TP amount may be the slight decrease in vegatation compared to 2003 (24 and 30%, respectively).

In Lake Mogan, DIN amount increased 2-fold in 2001, which was a low water level year, despite the fact that there were a lower external DIN load to the lake and the high submerged plant cover. Several studies in freshwater systems support the idea that submerged macrophytes can enhance nitrogen removal by offering surfaces that can hold populations of both nitrifiers and denitrifiers (Reddy & Busk, 1985; Eighmy & Bishop, 1989; Korner, 1997; Eriksson & Weisner, 1997). In Lake Mogan, DIN amount continued to increase onward 2002. High water level years observed in 2002 and 2003 were associated with high in-lake DIN amount and low submerged plant coverage. This result is similar with subtropical Lake Okeechobee, where DIN concentrations were higher at high water levels (James & Havens, 2005). Furthermore, lower submerged vegetation may lead to reduction in denitrification in the lake. In 2004, low external DIN load to the lake led to a 2 fold decrease in the DIN amount in Lake Mogan. In 2005, increase in DIN was not observed as in 2001, although 2005 was a low water level year. Denitrification or biological consumption by plant and algae can be responsible for this. In Lake Mogan, phytoplankton biovolume was high and increase in consumption by algae may have resulted in decrease in the in-lake DIN.

In low water level year 2004, the in-lake TP increased despite the fact that TP loads via inflows were significantly low in these years. Therefore, the results showed that the in-lake phosphorus amount was controlled by internal processes rather than external loading in the years with low water levels which coincided with the high hydraulic residence times. But DIN seems mostly depend on external loading and submerged plant cover in Lake Mogan.

The higher water level resulted in the higher phytoplankton density and, in turn, declining Secchi disk transparency in 2000, 2002 and 2003. In the case of high macrophyte biomass, phytoplankton biomass is naturally suppressed (Søndergaard & Moss, 1997). The allelopathic effects and reduced light conditions might have been important for suppressing phytoplankton (Wium-Andersen et al., 1987; Sand-Jensen, 1989).

During the studied period in Lake Mogan *A.bacillifer* and *D.pulex* densitiy was insignificant and cyanobacteria was dominant phytoplankton species in Lake Mogan. Our data suggest that maintenance of the clearwater in Lake Mogan probably occurred regardless of the top down impact of the large-bodied zooplankton, since their density was insignificant. During the study periods TP concentrations were low and lake submerged plant cover was high, it seems that nutrients were limited factor for phytoplankton in Lake Mogan. In 2005, due to reduction in light and decreasing in TP concentrations, bottom up factors become important and edible phytoplankton decreased significantly (p:0.02, 0.2x106 $\mu m^3\ ml^{-1}$) and favoured cyanobacteria (0.2x106 $\mu m^3\ ml^{-1}$) , in Lake Mogan.

Finally, the water level fluctuations, especially in spring, were also a very critical factor, along with water temperature, affecting submerged plant density. This, in turn, has an effect on the concentration of nutrients. Therefore, in shallow lakes in arid regions, the presence of submerged plants appeared to be just as crucial as in north temperate shallow lakes. Consequently, the sensitivity of submerged plants of shallow lakes, which are located in semi-arid to arid regions, to water level fluctuations and water temperature indicates that the littoral plant communities are particularly very susceptible to long term climatic changes. Furthermore, if the global climate change as anticipated leads to further increases in temperature and drier conditions, many freshwater lakes located in the Anatolian basin will become more vulnerable to dry conditions and to become saline in the future (Loaiciga et al., 1996). Since hydrology and submerged plants were important in determining the ecology of Lake Mogan, more importance should be given to hydrology and submerged plants for management programme in the future regarding the global warming in the area.

6. CONCLUSION

Studies on shallow lakes from the North Temperate Zone show that they alternate between clear and turbid water states in response to control factors. However, the ecology of semi-arid to arid shallow Mediterranean lakes is less explored (Beklioglu et al., 2003; Romo et al., 2004; Romo et al., 2005, Beklioglu et al., 2006a, Beklioglu et al., 2006b, Beklioglu & Tan accepted).

Water levels in shallow lakes naturally fluctuate intra- and interannually depending largely on regional climatic conditions. As it was stressed by Coops et al., 2003, water level fluctuations may become as significant as nutrients on functioning of shallow lakes through global climate change. Climate, on a local or catchment scale is of great importance for lake hydrology as it determines both the inputs and outputs of water, and the retention time. Shallow lakes located in arid to semi-arid regions are particularly sensitive to water stock. As a result, sensitivity to hydrological conditions also has significant consequences for the in-lake concentration of major ions and nutrients, especially for determining the salinity and conductivity (Wetzel 1983, Talling 2001). In Lakes Eymir and Mogan, the concentrations of salinity and nutrients increased especially in drought years 2004 and 2005 related with low water levels and increasing hydroulic retention time. Due to increasing salinity and nutrients, the ecology of lakes deterioted in 2004 and 2005. Rotifers and cyanobacteria become dominant in two lakes related with deteriotion in both lakes in 2004 and 2005. This results indicates the effect of water level fluctuations, on the ecology of shallow lakes (Blindow 1992, Talling 2001, Coops et al. 2003, Beklioglu et al. 2006b). However, most research on climate change impacts has focused on high latitude or temperate lakes (Carvalho & Moss 1999, Jeppesen et al. 2006), while less is known about low latitude lakes especially in arid climates region. Furthermore, the implications of climate warming may be even more challenging than in high latitude lakes since shallow lakes in the Mediterranean region are among the most sensitive to extreme climate changes. There is an urgent need for data on the ecology of shallow lakes in the region. However, more strong argument to support more research on Mediterranean lakes comes from the fact that global warming will probably make cold temperate limnosystems more similar to what Mediterranean lakes are now (Arnell et al., 1996). So further improvement of Mediterrenean studies such as Lake Eymir and Mogan may be very useful to improve our understanding about the role of hydrology in semi arid region and providing reference conditions for cold temperate lakes.

During the study period, combination of top-down and bottom up factors working together stabilized the turbid water conditions in the Lake Eymir. Despite the relatively clear relationship in Lake Eymir between *D.pulex* and phytoplankton grazing, this top-down control was sustained until planktivorous fish increased in the lake again. Since submerged plant become dominant and restricted phytoplankton for nutrients in clearwater state, bottom up factors especially nitrogen become limiting factor in clearwater state. During the study in Lake Eymir, increases in phytoplankton were followed by increases in zooplankton, but the impact of zooplankton feeding rates decreased due to dominance of large bodied zooplankton replacing by small bodied zooplankton. In turbid water state after biomanipulation increased in rotifers and lost of submerged plants resulted with decreasing the limiting effect of top down and bottom up factors and phytoplankton crop especially cyanobacteria increased in lake.

Since large bodied *A.bacillifer* and *D.pulex* densitiy was low and cyanobacteria was dominant phytoplankton species in Lake Mogan, it seems that top down control of phytoplankton was less effective in lake. During the study periods TP concentrations were low and lake submerged plant cover was high, it seems that nutrients were limited factor for phytoplankton in Lake Mogan.

In 2005, due to reduction in light and decreasing in TP concentrations, bottom up factors become important and favoured cyanobacteria, in Lake Mogan.

Ecology of submerged plants appears to have a crucial role for maintaining clear water state in Lake Eymir and Mogan. Dense submerged plants in Lake Eymir and Mogan increased water clarity. However, at high nutrient levels (TP> 0.25 mg L^{-1}) submerged plants did not prevent deterioration in water transparency in Lake Eymir. Drops in Spring Lake level enhanced vegetation development leading to a large expansion of the littoral zone in Lakes Eymir and Mogan. Due to increasing hydraulic residence time and low external loading in both lakes, internal processes become more important than external loading in 2004 and 2005. In Lakes Mogan and Eymir, high hydraulic residence time whether it is caused by high and low water level led to increase in the nutrients. Therefore, the present study provided a great example on strong impacts of submerged plant dominated clearwater state triggered by biomanipulation, governing mechanism for returning the turbid water state after biomanipulation and impact of hydrology on nutrients and ecology of shallow Lakes Eymir and Mogan. If hydrological changes through global warming in shallow lakes located especially in semi-arid or arid Mediterranean lakes are taken into consideration (Loaiciga et al., 1996), even lower thresholds of nutrient loading may be necessary for obtaining submerged vegetation dominated clearwater state in Mediterranean lakes since drought induced low water level years are associated with high TP, DIN, and salinity amounts.

CHAPTER 2

ROLE OF ENVIRONMENTAL VARIABLES IN DETERMINING PHYTOPLANKTON AND ZOOPLANKTON DYNAMICS OF LAKES EYMIR AND MOGAN

1. INTRODUCTION

The relationship between zooplankton and phytoplankton is an important base of the food web in lakes. Phytoplankton and zooplankton are two of the common biological parameters collected because they form the base of the aquatic food web and influence other aspects of the lake including color and clarity of the water and fish production (Wetzel, 2001). Phytoplankton (algae) are microscopic plants that are an integral part of the lake community. Phytoplankton use nutrients in the water and sunlight to grow and are the base of the aquatic food web (Lund, 1998). Zooplankton are small animals that float freely in the water column of lakes and oceans and whose distribution is primarily determined by water currents and mixing (Grant, 2001). Zooplankton play a vital role in aquatic food webs because they are important food for fish and invertebrate predators and they graze heavily on algae, bacteria, protozoa, and other invertebrates (Lampert, 1988; Sommer et al. 2000).

Ecosystems are regulated by an array of biotic and abiotic factors that are classified as top-down (predation) or bottom up (nutrients availability) processes (Roughgarden, 1994). Top down hypothesis (Carpenter et al., 1985) is based on the fundamental assumption that a change in predator biomass at the highest trophic level of an aquatic food web (piscivorous fish) cascade down to the lowest level (phytoplankton).

The competing bottom-up/top-down hypothesis (McQuenn et al., 1986) predicts that top-down effects are strong at the top of food web and weaken towards the bottom, because phytoplankton biomass is thoght to be more strongly controlled by resources (bottom-up) than by grazing (top-down). Top-down and bottom-up processes both occur and their relative importance varies between systems and across scales in time and size (Hairson et al., 1960; Cottingam & Carpenter,1998).

The abundance and diversity of phytoplankton and zooplankton vary in freshwater bodies according to their limnological features and trophic state (Jensen et al., 1994; Jeppessen et al., 2000; 2002; 2003). When nutrient loading is low, lakes present a clearwater phase with low algae biomass and a dense community of submerged macrophytes (Vanni & Temte, 1990; Jeppesen et al., 1999). On the other hand, the increase of nutrient loads in lakes may be responsible for an increase of the phytoplankton biomass. As the primary producers community undergoes changes, the structure of communities along the trophic web is subsequently affected. The phytoplankton increase leads to the high turbidity of water, which in turn, may contribute to the dissaperance of the submerged macrophytes community (Klinge et al., 1995). This fact enables changes in community structure, as it reduces plant surface area for algal fixation, the habitat and refuge for zooplankton, macroinvertebrates and fish, as well as impairing the slow-growing biomass of macrophytes, which is responsible for long term storage of nutrients by removing them from water column and sediment (McDougal et al., 1996). Moreover, some algal taxa that respond more swiftly than others to increase in nutrients, as cyanobacteria, are favoured. It is known that cyanobacteria are low quality food for zooplankton, due to their filamentous or colonial structure, low digestibility and toxin production, inducing limitations to the growth and reproduction of zooplanktonic organisms (e.g. Arnold, 1971; Haney, 1987; Lampert, 1987; Reinikainen et al., 1994; Repka, 1996; 1997; DeMott, 1999; DeMott et al., 2001; Rohrlack et al., 2001; Ghadouani et al., 2003). Cyanobacteria can affect zooplankton grazers by their colonial and filamentous morphology (Fulton & Paerl, 1987), toxicity (Fulton & Paerl, 1988), or lack of nutritional value (Von Elert & Wolffrom, 2001).

Changes in zooplankton community structure associated with the presence of filamentous and colonial cyanobacteria include the replacement of large cladocerans by smaller zooplankton species (Orcutt & Pace, 1984), possibly due to inhibition of the cladocerans' filtering machinery (Leonard et all 2005). Ultimately, the increase of nutrient concentrations can promote the massive growth of phytoplankton, which eventually die and are decomposed by microorganisms that are responsible for the elevated rates of organic matter decomposition and the subsequent anoxia, leading to large kills of fish and other aquatic life (Brönmark and Weisner, 1992).

In aquatic environments, phytoplankton growth may be controlled by the supply of limiting nutrients, usually nitrogen or phosphorus (Wetzel, 2001). Nutrient limitation could be inferred from the water column total nitrogen: total phosphorus (TN: TP) ratio. Phytoplankton biomass, as indicated by Chl a, was strongly dependent upon TP in the lakes, and there was a weaker relationship with TN. Of the major hypotheses that explained the success of cyanobacteria, the most prevalent and disputed explanation may be the TN: TP (total phosphorus) ratio. In an analysis based on 17 lakes located worldwide, Smith (1983) Smith (1983) found a negative association between the ratio of total N to total P (TN: TP) and the percent dominance of blue-greens (cyanobacteria). This conclusion has led to the so-called ''TN: TP rule'' that increasing the mass ratios above 29 will reduce the proportion of cyanobacteria as a fraction of the total algal biomass. There have been many substantial reviews to discuss the impact of TN: TP ratios on phytoplankton populations, some support the ''TN: TP rule'' (Takamura et al., 1992; Aleya et al., 1994), but others hold the reverse view. Scheffer et al, (1997) have recognized that even when such a response is observed, it may be due to the increasing P concentrations rather than a decrease in the N: P ratio. However, shallow lakes contradicts this assumption (Beklioğlu & Tan, accepted, Jensen et al., 1994, Havens et al., 2003).

Zooplankton can influence the phytoplankton community through preferential grazing (Shapiro, 1995) and by the resupply of nutrients through extcretion (Carney and Elser, 1990). The study of phytoplankton dynamics regarding environmental physical disturbances was analyzed and evaluated (Reynolds, 1984; Harris, 1986; Padisák, 1992, 1993).

Zooplankton community structure and biomass are determined by food availability, temperature, predation and interactions between different zooplankton, including inter-specific and interference competition (Gliwicz, 1985; Sterner, 1989; Gliwicz, 2003). Other investigations (Deneke & Nixdorf, 1999) and minimal models (Scheffer et al., 1997) also lend support to the hypothesis that decreasing zooplankton grazing occurs at high fish densities. Conversely, fish kills or low fish recruitment may lead to high grazing pressure on phytoplankton and clear water conditions throughout the summer (Kubecka & Duncan, 1994; Gliwicz, 2003). Control of grazing zooplankton by zooplanktivorous fish, tend to preserve algal domination even where the nutrient concentration is suitably low for stable macrophyte populations to exist (Irvine et al., 1989; Scheffer et al., 1993). It is now widely accepted that biomanipulation, the top-down manipulation of fish communities (i.e. enhancement of piscivores or reduction of zooplanktivores and/or benthivores) is required as an extreme perturbation to push one stable equilibrium to another (Perrow et al., 1997). Although several mechanisms have been suggested for the maintenance of macrophyte communities in restored lakes (Perrow et al., 1997), research has thus far tended to focus on the provision of macrophytes as refuges for zooplankton (Phillips et al., 1996; Schriver et al., 1995; Burks et al., 2002), as this mechanism may be applicable both when macrophytes have become rapidly established or where they suffer some lag in development. The migration of grazing Cladocera from macrophyte refuges to open water has the potential to limit phytoplankton growth outside of macrophyte beds, thus providing a potential mechanism for the stabilisation of macrophytes (Timms & Moss, 1984). Conversly, when fish reduced the biomass of large crustaceans, there was a complementary increase in the biomasses of smaller crustacean species and rotifers and increase in cyanobacteria (Vakkilainen et al., 2004).

Hydrology directly determines the temperal pattern of zooplankton community (Badosa et al., 2006) and also affects nutrient dynamics and phytoplankton species and biomass and this indirectly affect on zooplankton (Rennella et al., 2006).

A common feature of most natural ecosystems is the pronounced variation in space and time. In order to separate natural variations from impacts of human interventions, we need to understand the natural variation in time and space. This understanding can only be achieved by analysing long term data sets. Long-term studies are important in relation to the effects of e.g. climatic changes, acid precipitation, eutrophication, and hydropower regulation of rivers and lakes (Halvorsen et al., 2004). Since a number of physical, chemical and biological factors influence the zooplankton and phytoplankton community and the interaction between them are complicated, and the relative importance of the factors varies both seasonally and between years, it is therefore difficult to identify the most important factors at any time.

In recent decades, a large number of multivariate techniques have been developed to aid the interpretation of complex data sets. These techniques typically identify groups of samples that have similar community composition based on the abundance or biomass of different taxa. Multivariate statistics allow for the analysis of complex zooplankton (Sprules, 1977, 1984; Stemberger and Lazorchak, 1994; Stemberger and Miller, 1998) and phytoplankton (Van tongeren et al., 1992, Romo & Tongeren, 1995) data sets against a number of environmental variables to identify the nature of zooplankton and phytoplankton variation along ecologically significant environmental gradients. Dominant patterns of community variation can then be correlated with physical, chemical and biological parameters to further aid interpretation.

1.1. SCOPE OF THE STUDY

Zooplankton and phytoplankton abundance, composition and environmental parameters (nutrients, salinity, temperature, Secchi disk transparency etc) were monitored in Lake Eymir and Lake Mogan from 1997 to 2005. The data were subjected to Canonical correspondence analyses (CCA) to determine the patterns that group zooplankton and phytoplankton in relation to environmental parameters. This large data set provided a unique opportunity to examine relationships with great statistical power. The aim of this study to understand zooplankton and phytoplankton community structure relation to environmental conditions. Also this study will improve our understanding of the trophic linkage between zooplankton and phytoplankton in Lake Eymir and Lake Mogan.

2. STATISTICAL METHODS

Canonical Correspondence Analysis (CCA) has been developed to relate the abundance of species to environmental variables. The anaylsis allows us to detect patterns of species distribution related to physical and chemical parameters CCA provides an integrated description of species environment relationships by assuming a response model that is common to all species and the existence of a single set of underlying environmental gradients to which all the species respond (Ter Braak, 1986). The same strong assumption is implicit in all ordination techniques. CCA has the advantage over other techniques in that it focuses on the relationships between species and measured environmental variables and so provides an automated interpretation of the ordination axes (Ter Braak & Verdanschot, 1995).

Canonical Correspondence Analysis (CCA) used the XLSTAT (Addin Software Design, U.S.A, 2006) for performing multivariate analysis of the data.

In order to describe relations among environmental variables and phytoplankton communities in Lake Eymir, CCA was carried using a matrix data with environmental variables (TP, SRP, ammonium, nitrate, silicate, Secchi disk transparency, temperature and SS) and using a matrix with main phytoplankton taxa. Dinoflagellates, Diatoms, Cryptophyta, Cyanobacteria and Chlorophyta (all species encountered from 1997 to 2005) were selected as main phytoplankton taxa for CCA. In order to describe relations among environmental variables and phytoplankton communities in Lake Mogan, CCA was carried using a matrix data with environmental variables (Secchi disk transparency, temperature, SS, salinity, ammonium and nitrate) and using a matrix with main phytoplankton taxa. copepods Dinoflagellates, Diatoms, Cryptophyta, Cyanobacteria and Chlorophyta (all species encountered from 1997 to 2005) were selected as main phytoplankton taxa for CCA.

In order to describe relations among environmental variables and zooplankton communities in Lake Eymir, CCA was carried using a matrix data with environmental variables (TP, SRP, ammonium, nitrate, Secchi disk transparency, temperature, oxygen and chlorophyll-a) and using a matrix with main zooplankton taxa. Largebodied zooplankton *D. pulex* and *A. bacillifer,* Cyclopoid copepods (*Eucyclops sp, Mesocyclops sp., Megacyclops sp*) and rotifers (all species encountered from 1997 to 2005) were selected as main zooplankton taxa for CCA. In order to describe relations among environmental variables and zooplankton communities in Lake Mogan, CCA was carried using a matrix data with environmental variables (Secchi disk transparency, temperature, SS, salinity, conductivity and chlorophyll-a) and using a matrix with main zooplankton taxa. Largebodied zooplankton *D. pulex* and *A. bacillifer* and Cyclopoid copepods (*Eucyclops sp, Mesocyclops sp., Megacyclops sp*) were selected as main zooplankton taxa for CCA for the period of 1997-2005.

In order to describe relations among environmental variables and phytoplankton communities in Lake Mogan, CCA was carried using a matrix data with environmental variables (Secchi disk transparency, temperature, SS, salinity, ammonium and nitrate) and using a matrix with main phytoplankton taxa. Dinoflagellates, Diatoms, Cryptophyta, Cyanobacteria and Chlorophyta (all species encountered from 1997 to 2005) were selected as main phytoplankton taxa for CCA.

In order to describe relations among environmental variables and zooplankton communities in Lake Mogan, CCA was carried using a matrix data with environmental variables (Secchi disk transparency, temperature, SS, salinity, conductivity and chlorophyll-a) and using a matrix with main zooplankton taxa. Largebodied zooplankton *D. pulex* and *A. bacillifer* and Cyclopoid copepods (*Eucyclops sp, Mesocyclops sp., Megacyclops sp*) were selected as main zooplankton taxa for CCA for the period of 1997-2005. Due to the lack of data for rotifers for 1997 to 2003, they were excluded for CCA.

In order to reach highest varience, all analysis were performed with different combinations of variables and the variables, which gave best results, choose for analysis.

The results contain the environmental variables plotted as arrows emanating from the center of the graph along with points for taxa. The arrows representing the environmental variables indicate the direction of maximum change of that variable across the diagram. The position of the species points indicates the environmental preference of the species.

In order to describe relations among environmental variables and plankton communities in Lake Eymir and Lake Mogan, CCA was carried out seperately for each lake using a matrix data with environmental variables and using a matrix with zooplankton and phytoplankton tax seperately.

3. RESULTS

3.1 Lake Eymir

3.1.1 Phytoplankton Community dynamics of Lake Eymir

During biomanipulation, compared to the pre-biomanipulation period in Lake Eymir, the phytoplankton community composition did not show a great change. Chlorophyta dominated the community ($0.88x10^6$ μm^3 ml^{-1}) but also Cryptophyta and Cyanobacteria biovolumes were significant ($0.21x10^6$ and $0.24x10^6$ μm^3 ml^{-1}, respectively). *Chlamydomonas globosa* and *Oocystis parva* were the dominant species during biomanipulation (0.27 and 0.26 $24x10^6$ μm^3 ml^{-1}, respectively).

Phytoplankton community of clearwater state included species of Chlorophyta, Cyanobacteria, Dinoflagellates, Cryptophyta and Diatoms. In 2003, the biovolume of diatom was significantly higher than other years ($p<0.0001$, $0.31x10^6$ μm^3 ml^{-1}). The diatom species encountered were *Cyclotella meneghiniana, Fragilaria sp, Navicula sp., Amphora ovalis*, *Synedra ulna, Stephanodiscus sp., Melosira sp*. and *Asterionella formosa*. *Synedra ulna* was the dominant diatom species in 2003 ($0.280.31x10^6$ μm^3 ml^{-1}).

In the period of turbid water state after biomanipulation, phytoplankton community included cyanobacteria, cryptophyta and diatom. Contribution of Chlorophyta (*Oocystis parva, Ankistrodesmus falcatus, Ankistrodesmus sp. Schroederia setigera, Chlamydomonas globosa, Mallomonas acaroides, Phacus sp*) and Dinoflagellates (*Peridinium sp*) to phytoplankton numbers was too low (3% and 1%, respectively). In 2004, Cyanobacteria dominated the community during summer ($0.17x10^6$ μm^3 ml^{-1}) and diatoms dominated the community during spring ($0.71x10^6$ μm^3 ml^{-1}).

Chorococcus sp was dominant Cyanobacteria during summer ($0.16x10^6$ μm^3 ml^{-1}). However, *Cyclotella meneghiniana* was dominant diatom during spring (0.06x106 µm3 ml^{-1}).

In 2005, Cyanobacteria dominated the community and their biovolumes were significantly higher than previous years (p: 0.017, $2.32x10^6$ μm^3 ml^{-1}). The biovolume of diatom was significantly lower than other years ($p<0.0001$, $0.01x10^6$ μm^3 ml^{-1}).

The biovolume of Chlorophyta was significantly lower than other years in both 2004 and 2005 ($p<0.0001$, $0.07x10^6$ μm^3 ml^{-1}, $0.01x10^6$ μm^3 ml^{-1} respectively). Although the biovolume of Cryptophyta decreased in 2004, their biovolume increased in 2005 and they become the second dominant group ($0.15x10^6$ μm^3 ml^{-1} and $0.87x10^6$ μm^3 ml^{-1}, respectively). *Cryptomonas ovata* was the dominant Cryptophyta species in 2005 ($0.86x10^6$ μm^3 ml^{-1}).

In August 2005, Cyanobacteria bloom, which caused fish kills in lake, was observed. *Anabaena sp.* and *Anabaenopsis sp.* were the observed species during the bloom ($9.8x10^6$ μm^3 ml^{-1} and $0.24x10^6$ μm^3 ml^{-1}, respectively).

Table 3.1. The list of phytoplankton species in Lake Eymir observed from 1997 to 2005.

CHLOROPHYTA		CYANOPHYTA
Gleocystis major	*Planktospheria sp.*	*Coelospherium sp.*
Botryococcus sp.	*Carteria cordiformis*	*Merismopedia elegans*
Oocystis crassa	*Chlorella vulgaris*	*Anabaena sp.*
Oocystis parva	*Crucigenia tetrapedia*	*Aphanocapsa sp.*
Ankistrodesmus falcatus	*Scenedesmus quadriqauda*	*Anabaenopsis sp.*
Ankistrodesmus sp.	*Mallomonas acaroides*	*Microsystis sp.*
Schroederia setigera	*Ankistrodesmus fractus*	*Spirulina sp.*
Chlamydomonas angulosa	*Eudorina elegans*	*Chorococcus sp.*
Chlamydomonas globosa	*Kirchneriella lunaris*	*Oscillatoria sp.*
Chlamydomonas sp.	*Phacus sp.*	*Phormidium tenue*
Tetraedron sp.	*Astereococcus sp.*	*Gleocapsa sp.*
DIATOMS	**DINOFLAGELLATES**	**CRYPTOMONADS**
Cyclotella meneghiniana	*Ceratium hidruniella*	*Cryptomonas ovata*
Fragilaria sp.	*Glenodium sp.*	*Rhodomonas sp.*
Navicula sp.	*Peridinium sp*	
Amphora ovalis	*Gymnodium sp.*	
Synedra ulna		
Stephanodiscus sp.		
Melosira sp.		
Pinnularia sp		
Gomphonema		
Cymbella sp.		
Asterionella formosa		

3.1.2 CCA of phytoplankton in Lake Eymir

For Lake Eymir, CCA F1 and F2 explained 68.85% of the variance in the species environment biplot (Figure 3.1). The p value is 0.03. F1 was mostly related with nutrients (TP, SRP, silicate and ammonium), F2 was related with Secchi disk depth, temperature, and concentrations of SS. Cyanobacteria and Cryptomonads were positioned on left upper part of matrix and had a great correlation with higher concentrations of SS and ammonium. Chlorophyta were positioned on the right upper part of matrix and they thrive in water with higher TP and SRP concentrations and lower SS. Diatoms had a great correlation with higher silicate concentrations.

Dinoflagellates were positioned on the left lower part of matrix and they thrive in water with warmer temperature and moderate nutrient concentrations.

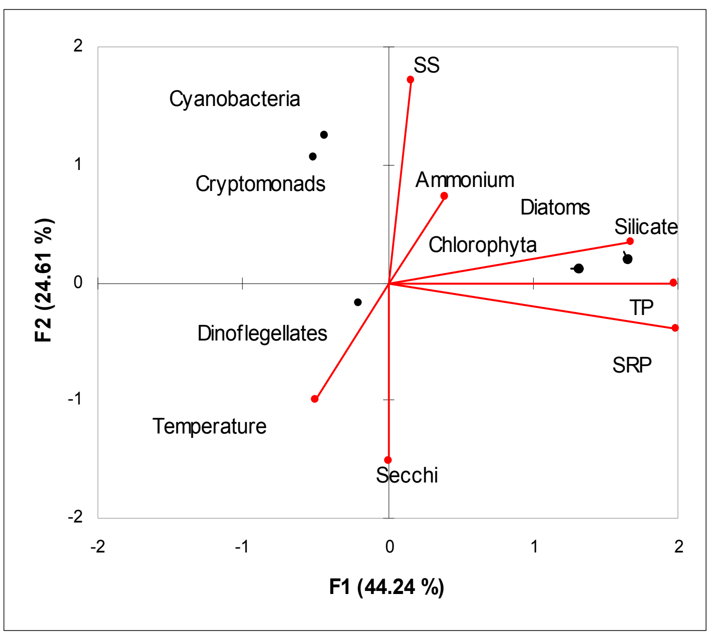

Figure 3.1. CCA ordination with phytoplankton (points) and environmental variables (arrows) in Lake Eymir.

3.1.3 Zooplankton Community dynamics of Lake Eymir using CCA

Zooplankton community of Lake Eymir was species poor before biomanipulation. The zooplankton community composed of one species of calonoid copepods (*A.bacillifer*), 3 species of cladocerans *(Daphnia pulex* and *Ceriodaphnia reticulata),* one species of cyclopoid copepods (*Eucyclops sp*), three species of rotifers (*Keratella quadrata, Brachionous calyciflorus and Filinia longiseta*). Rotifers were the dominant group and their contribution to the zooplankton community was 30%.

Biomanipulation led to a marked increase in the density of the largebodied zooplankton *D. pulex* and *A. bacillifer*, which also dominated the zooplankton community from 1999 to 2003. Furthermore, the density of the species peaked in spring and early summer and decreased in late summer. The clearwater state zooplankton community was richer than turbid water state. New species of cyclopoid copepods (mesocyclops sp, megacyclops sp), caladoceran species (*Diaphanosoma branchyrum, Bosmina longristis, Chydorus sphaericus, Alona affinis),* and rotifers *(Hexarthra sp., Polyarthra sp.,* and *Asplancha sp*) encountered during the clearwater state.

In both 2004 and 2005, rotifers become dominant again and *A.bacillifer* and *D.pulex* densities decreased compared to clearwater state. The small cladoceran were not observed during the turbid water state. Keratella quadrata was the dominant species in 2004 (17 ind l^{-1}) and *Brachionous calyciflorus* and Polyarthra sp. were dominant in 2005 (87 and 37 ind l^{-1}, respectively).

For Lake Eymir, CCA F1 and F2 explained 94.16% of the variance in the species environment biplot (Figure 3.2). The p value was less than 0.0001. Secchi disk depth, concentrations of TP and SRP had a great correlation with the distribution of *D.pulex*. Zooplankton that thrived in water with a somewhat lower temperature and high concentrations of chl-a, ammonium, lower concentrations of nitrate values as Rotifera were positioned on the left. Cyclopoid copepods were positioned left lower part and they thrived at water with higher oxygen and lower chl-a.

A.bacillifer was positioned right lower part and it had a correlation with high concentrations of SRP, lower concentrations of ammonium and nitrate.

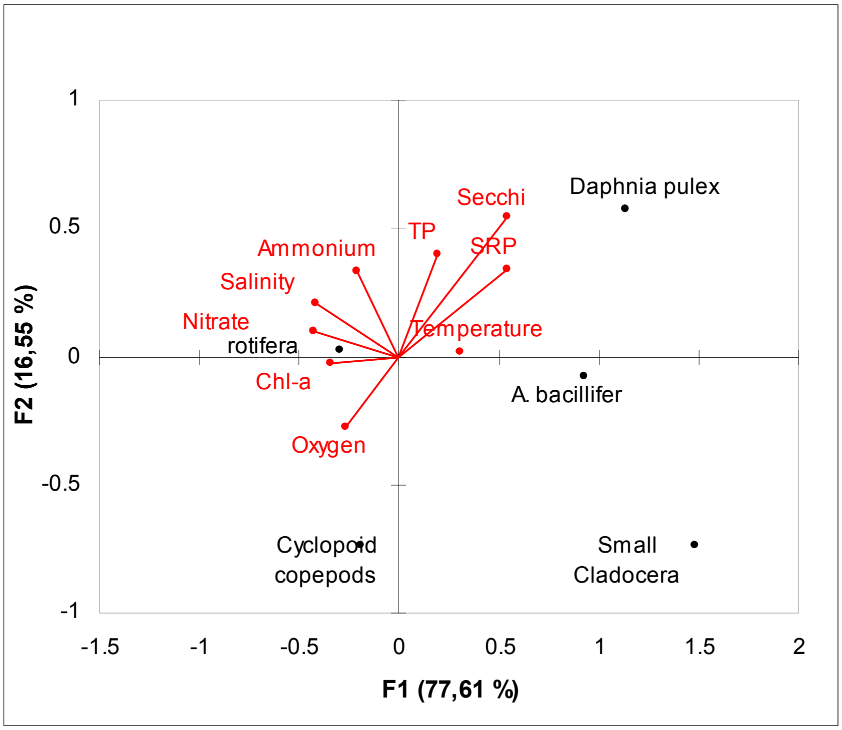

Figure 3.2. CCA ordination with zooplankton (points) and environmental variables (arrows) İn Lake Eymir.

3.2. Lake Mogan

3.2.1 Phytoplankton Community of Lake Mogan

In Lake Mogan, phytoplankton community included species of Cyanobacteria, Dinoflagellates, Chlorophyta, Diatoms and Cryptophyta from 1997 to 2005 (Table 3.2). In both 2004 and 2005, Cyanobacteria dominated the phytoplankton community ($0.8x10^6$ μm^3 ml^{-1} and $1.1x10^6$ μm^3 ml^{-1}, respectively). Diatoms and Chlorophyta biovolumes significantly decreased in 2004 and 2005 (p: 0.005, 0.07 and $0.131x10^6$ μm^3 ml^{-1} and p:0.008, 0.04 and $0.011x10^6$ μm^3 ml^{-1}, respectively).

Anabaena sp. and *Merismopedia elegans* were the dominant species in 2004 (0.49×10^6 μm^3 ml^{-1} and 0.18×10^6 μm^3 ml^{-1}, respectively). *Anabaena sp.*and *Anabaenopsis sp.* were dominant in 2005 (0.41×10^6 μm^3 ml^{-1} and 0.17×10^6 μm^3 ml^{-1}, respectively).

Table 3.2 The list of phytoplankton species in Lake Mogan observed from 1997 to 2005.

DIATOMS	**CHLOROPHYTA**	
Cyclotella meneghiniana	*Gleocystis major*	*Carteria cordiformis*
Fragilaria sp.	*Sphaerocystis sp.*	*Chlorella vulgaris*
Navicula sp.	*Oocystis crassa*	*Crucigenia tetrapedia*
Amphora ovalis	*Oocystis parva*	*Scenedesmus quadriqauda*
Synedra ulna	*Ankistrodesmus falcatus*	*Mallomonas acaroides*
Stephanodiscus sp.	*Ankistrodesmus sp.*	*Ankistrodesmus fractus*
Melosira sp.	*Schroederia setigera*	*Treubaria setigerum*
Pinnularia sp.	*Chlamydomonas angulosa*	*Lagerheima sp.*
Gomphonema sp.	*Chlamydomonas globosa*	*Eudorina elegans*
Cymbella sp.	*Chlamydomonas sp.*	*Kirchneriella lunaris*
Diatoma sp.	*Tetraedron sp.*	*Phacus sp.*
Asterionella formosa	*Planktospheria sp.*	*Astereococcus sp.*
		Botryococcus sp.
DINOFLAGELLATES	**CRYPTOMONADS**	**CYANOPHYTA**
Ceratium hidruniella	*Cryptomonas ovata*	*Coelospherium sp.*
Glenodium sp.	*Rhodomonas sp.*	*Merismopedia elegans*
Peridinium sp		*Anabaena sp.*
Gymnodium sp.		*Aphanocapsa sp.*
		Anabaenopsis sp.
		Microsystis sp.
		Spirulina sp.
		Chorococcus sp.
		Oscillatoria sp.
		Gleocapsa sp.

3.2.2 CCA of phytoplankton in Lake Mogan

For Lake Mogan, CCA F1 and F2 explained 71.19% of the variance in the species environment biplot (Figure 3.3) (p: 0.01). F1 was mostly related with Secchi disk depth, SS and salinity. F2 was correlated with temperature and concentrations of ammonium and nitrate.Cyanobacteria and Cryptomonads were related with higher SS, salinity and higher temperature. Chlorophyta thrived at water with high secchi disk transparency and higher concentrations of nitrate and ammonium. Dinoflagellates prefer the water with higher secchi depth, higher ammonium and lower concentrations of SS and lower salinity. Diatoms thrived in water lower concentrations of ammonium and nitrate and warmer temperature.

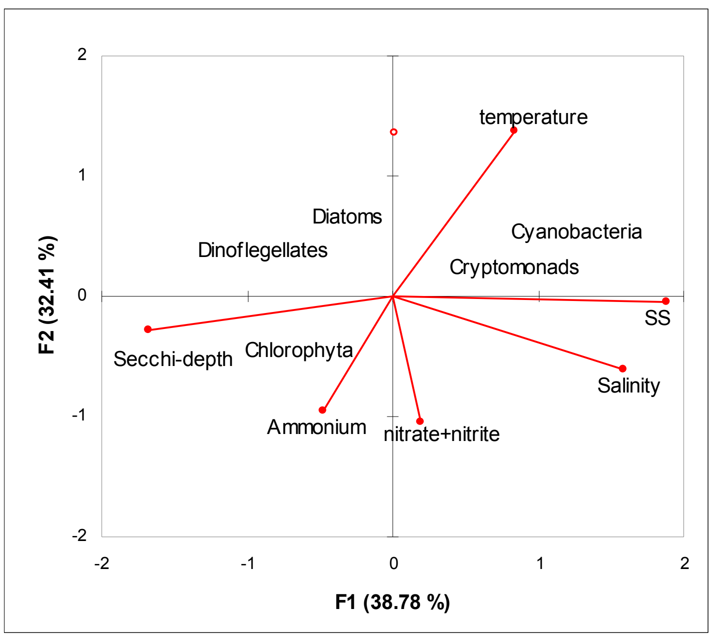

Figure 3.3. CCA ordination with zooplankton (points) and environmental variables (arrows) İn Lake Mogan.

3.2.3 Zooplankton Community dynamics of of Lake Mogan using CCA

The zooplankton community of Lake Mogan was composed of one species of calonoid copepods (*A. bacillifer*), three species of cyclopoid copepods (*Eucyclops sp, Mesocyclops sp, and Megacyclops sp*), six species of cladocerans (*D.pulex*, *Diaphanosoma lacustris, Ceriodaphnia reticulata, Bosmina longristis, Chydorus sphaericus, Alona affinis*).

In Lake Mogan, the density of zooplankton community was very low largely dominated by *A. bacillifer*. *Daphnia pulex*, *Diaphanosoma lacustris Sars, and Cyclops spp.* were recorded; their abundances being, however, low and without significant interannual differences. Moreover, the density of all the species peaked in early summer.

In 2005, cyclopoid copepods density significantly increased (p:0.001, 5 ind l^{-1}). Rotifers become dominant in 2004 and 2005 (112 and 238 ind l^{-1}, respectively). *Keratella quadrata, Filinia sp, Brachionous calcyflorus, B. Quadritentatus, B.angularis, Hexarthra sp., Asplancha sp., Polyarthra sp.* and *Anuruepsis fissa* were the rotifer species encountered in 2004 and 2005. Filinia sp. was the dominant species in 2004 (59 ind l^{-1}). *Keratella quadrata* and *Brachionous calcyflorus* were dominant species in 2005 (58 and 56 ind l^{-1}, respectively).

For Lake Mogan, CCA F1 and F2 explained 71.6% of the variance in the species environment biplot (Figure 3.4). The p value is 0.001. Zooplankton that thrived in water with high transparency as Rotifera were positioned near the F2 since F2 related with secchi depth, chl-a and temperature. *A.bacillifer* and *D.pulex* have similar environmental preferences. They both prefer lower salinity and conductivity, low SS, warmer temperature and low concentrations of chl-a. Cyclopoid copepods were positioned right upper part and they thrive in water with lower salinity and conductivity and higher SS concentrations than other zooplankton taxa.

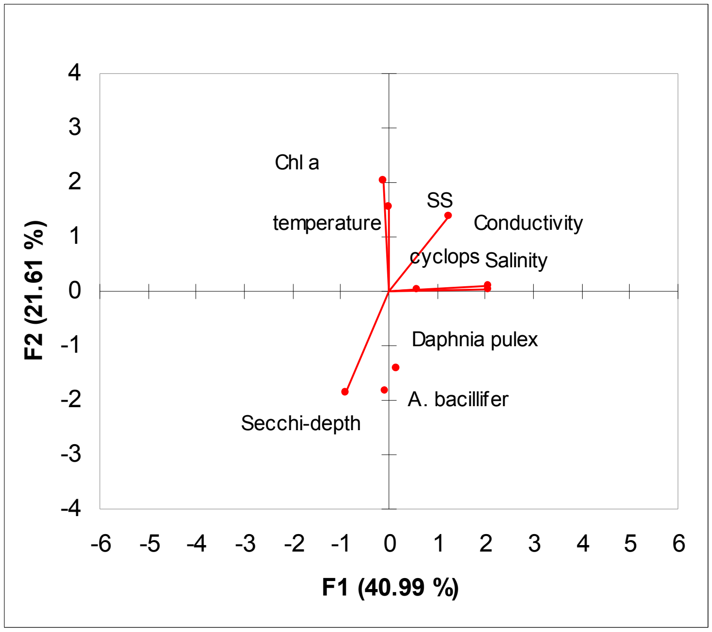

Figure 3.4. CCA ordination with zooplankton (points) and environmental variables (arrows) İn Lake Mogan.

4. DISCUSSION

The relationship between zooplankton and phytoplankton is an important base of the foodweb in lakes (Abrantes et al., 2005). The abundance and diversity of phytoplankton and zooplankton vary in freshwaters according to their limnological features and trophic state (Jensen et al., 1994, Jeppesen et al., 2000, 2002 and 2003).

The canonical correspondence analyses (CCA) performed on the phytoplankton data in Lakes Eymir and Mogan to understand the relationships between environmental parameters and phytoplankton communities. In Lake Eymir, CCA revealed that phytoplankton was mainly regulated by nutrients. However, water transparency (SS and Secchi disk transparency) were important to determine which taxa become dominant in lake. Our field data shows that in clearwater state was dominated by chlorophyta and turbid water state after biomanipulation was dominated by cyanobacteria. These results were in accordance with results of CCA. Chlorophyta biovolume had correlation with low SS while cyanobacteria biovolume had correlation with high SS and low Secchi disk transparency.

In Lake Mogan, CCA revealed that phytoplankton was mainly regulated by nutrients and water transparency. In Lake Mogan, cyanobacteria was dominant during the study period and it seems that water transparency was less important in determining phytoplankton taxa than Lake Eymir. It may be related with temperature and salinity are the other factors determine phytoplankton in Lake Mogan. In 2004 and 2005, cyanobacteria increased with increased salinity. This coincided with CCA because cyanobacteria had correlation with high salinity.

Furthermore, the responce of phytoplankton to temperature was different between lakes. In Lake Mogan, CCA revealed that cyanobacteria was related with warmer temperatures, although same responce was not observed in Lake Eymir. This result coincides with Kagalov et al., (2003) that temperature is main factor in producing effects of cyanophytes abundance in Lake Pamvotis, Greece. However, Cryptophyta and cyanobacteria had same environmental preferences both in Lakes Eymir and Mogan.

The canonical correspondence analyses (CCA) performed on the zooplankton data in Lakes Eymir and Mogan to understand the relationships between environmental parameters and zooplankton communities. In Lake Eymir, CCA revealed that zooplankton was mainly regulated by nutrients. *Daphnia pulex* was the taxa most positively correlated with high Secchi disk transparency. This result seems reasonable because Warner (1999) reported that abundance of cladoceran zooplankton correlated with high water transparency because of ability to effectively crop phytoplankton. The location of rotifers and *Daphnia pulex* was consistent with their environmental preferences in Lake Eymir. In Lake Eymir, *Daphnia pulex* was dominant during clearwater state with high Secchi disk transparency excluding high planktivorous fish periods. CCA results implicates that rotifers highly correlated with high ammonium and high salinity. Our field data in 2004 and 2005 are inacordance with this that rotifers were dominant in high concentrations of ammonium and high salinity.

Since high ammonium concentration is not favourable for many zooplankton species as it prevents their colonization (Rocha et al., 1999), CCA are reasonable. Because only rotifers seems associated with high concentrations of ammonium.

In Lake Mogan, CCA revealed that zooplankton was mainly regulated by water transparency (Secchi disk transparency and SS). *D. pulex* and *A. bacillifer* were highly correlated with high Secchi disk transparency, although cyclopoid copepods were related with high SS. *A.bacillifer* was dominant from 1997 to 2003 and due to deteriotion of water transparency, rotifers become dominant in Lake Mogan and this implicates that water transparency was important for abundance of *A. bacillifer*.

Other parameters that were not included in these CCA, such as fish, zooplankton, phytoplankton interactions and submerged plants probably also affected the spatial and temporal distribution of zooplankton and phytoplankton in Lakes Eymir and Mogan. In both for zooplankton and phytoplankton, predation and interaction with submerged plants are important factors conditioning phytoplankton and zooplankton communities.

As their population development is slower, cladocerans are more likely to be controlled by a top down processes (predator) while rotifers life histories are strongly influenced by bottom up (nutrients) mechanisms (Walz, 1997). In Lake Eymir, planktivorous and benthivorous fish were dominant in 2004 and 2005. Due to high grazing pressure, zooplankton community shifted from large bodied zooplankton dominated state to rotifer dominated state.

Generally, rotifers were dominant in Lakes Eymir and Mogan, in 2004 and 2005. it may be related with rotifers high reproductive rate (Allan, 1978) and decrease in the abundance of *A.bacillifer* and *D. pulex* since rotifers and crustaceans are known to compete for many of the same phytoplankton species (Kirk, 1991) and larger cladocerans also interfere mechanically with rotifers (Gilbert, 1988).

In many lakes, *Daphnia* do not tolerate even low levels of salinity (Moss, 1994; Jeppesen et al., 1994) and increases salinity leads to increases in mortality (Teshner, 1995, Hall et al., 2002). Furthermore, cladocerans, in general, are poorly represented in marine environments (Frey, 1993). It was reported by Schallenberg et al., 2002, that the zooplankton community structure in Lake Waihola changed with increasing salinity from cladoceran dominated to rotifer dominated state. This result is in accordance with observed results in Lake Eymir. As a conclusion, the shift from large bodied zooplankton to small-bodied zooplankton may be related with increasing salinity in 2004 and 2005 in Lake Eymir.

There were significant relationship between zooplankton, phytoplankton and nutrients which suggest that nutrients controlled by phytoplankton while phytoplankton controlled by zooplankton in Lake Eymir. However, top down control of zooplankton may reduce the grazing pressure of zooplankton on phytoplankton.

The role of zooplankton on phytoplankton was not strong in Lake Mogan since the *A.bacillifer* and *D.pulex* numbers were significant during the study period.

Submerged plants are important for both zooplankton and phytoplankton. Submerged plants serve as a refuge for zooplankton from fish grazing. They also counteract increases in algal turbidity by compete for nutrients, allelopathic affects and reduce resuspension (Moss, 1998, Jeppesen et al., 2002). In 2004, the complete return of Lake Eymir to turbid water state dominated by cyanobacteria seems to have depended upon turnover of the limiting nutrient, which was retarded by submerged plants and stimulated by planktivorous fish and loss of refuge for cladocerans due to lost of buffer mechanisms of submerged plants.

Daphnia species often decrease during cyanobacterial blooms (Gliwicz & Lampert, 1990). Due to increasing nutrient availability and deteriotion of water transparency, cyanobacteria increased. D.pulex decreased in Lake Eymir, in 2004 due to increase in cyanobacteria. The reasons for that are cyanobacteria interfere with filtration, have low digestibility, and release toxins (Lampert, 1987, Demott et al., 2001).

Typically, it assumed that high phosphorous availability and low N: P ratios are favourable in nitrogen fixing cyanobacteria blooms (Reynolds, 1987) but in Lake Eymir the DIN: SRP ratio was below 29:1 by mass (Smith, 1983) but bloom did not occur in clearwater state. In contrast, Cyanobacterial bloom of *Anabaena sp.* was recorded in lake when DIN: SRP ratio was above 29:1 and DIN ammount increased largely by ammonium (Beklioğlu & Tan, accepted). In 2004, DIN: SRP ratio decreased to 0.97 and cyanobacteria *Anabaena sp.* bloom observed. This may be not related with low DIN: SRP ratio but may be related with increasing SRP concentrations in the lake (Scheffer et al, 1997) because the lake DIN ammount was increased largely by ammonium. In 2005, cyanobacterial Anabaena sp. bloom was recorded when DIN: SRP ratio was above 29:1 ratio. Cyanobacterial bloom and abundance may be related with other factors in Lake Eymir.

Cyanobacteria promoted turbidity, also better competitors under turbid conditions (Carter et al., 2005), and during turbid water state they dominant community in Lake Eymir. Due to increasing hydroulic residence time in both lake, phosphorous release from sediment become an important source for phytoplankton and this may favour cyanobacteria in both lakes. In addition, increase in water temperature and pH, reduced transparency and lack of top-down control by zooplankton may be the other explanations for increase in cyanobacterial abundance.

In conclusion, different zooplankton and phytoplankton taxa corresponding to environmental conditions in Lake Eymir and Lake Mogan were identified. The indicator properties of zooplankton and phytoplankton (eg. rotifers and cyanobacteria indicator for turbid conditions) can be used to identify different physical and chemical gradients in lakes and therefore it can be employed in environmental monitoring programmes.

REFERENCES

Abrantes, N, 2002, Population dynamics of Ceriodaphnia pulchella (Crustacea, Cladocera). M. Sc. Thesis. University of Aveiro. Department of Biology, Aveiro. 120 pp.

Abrantes, N. J., Antunes, S. C., Pereira, M. J., Goncalues, F., 2005. Seasonal succession of cladocerans and phytoplankton and their interactions in a shallow eutrophic lake (Lake Vela, Portugal). Acta Oeecologica 27: 1-11.

Aleya L, Desmolles F, Michard M, Bonnet MP, Devaus J., 1994. The deterministic factors of the Microcysitis aeruginosa blooms over a biyearly survey in the hypereutrophic reservoir of Villerest (Roanne, France). Arch Hydrobiol 4:489–515.

Allan, J. D., 1976. Life History patterns in zooplankton. Amer. Natur. 110: 165-189.

Altinayar, G., 1988. Su Yabancıotları. DSI Basım ve Foto-Film İşletme Müdürlüğü Matbaası, Ankara: 239 pp. (in Turkish).

Altınbilek, D., Usul, N., Yazıcıoglu, H., Kutoglu, Y., Merzi, N., Gögüs, M. Doyuran, V. & Günyaktı, A. (1995) Gölbası Mogan-Eymir Gölleri için su kanakları ve çevre yönetim planı projesi. Technical Report. No. 93-03-03-04-01. Middle East Technical University, Ankara: 680 pp. (inTurkish).

Alvarez, M. C., Castello S. C., Rojo, C., Rodrigo M. A., 2004. La contaminacion de las lagunas de ruidera. Actas de las II Jornadas sobre el Medio Natural Albacetense: 293-301.

Alvarez, M. C., Rojo, C., Angeler, D. G., 2005. Mediterranean limnology:current status, gaps and the future. J. Limnol. 64: 13-29.

Amemiya, T., T. Enomoto, A. G. Rossberg, N. Takamura and K. Itoh. 2005. Lake restoration in terms of ecological resilience: a numerical study of biomanipulations under bistable conditions. Ecology and Society 10: 3-15.

Angeler D. G., M. C. Alvarez, S. C., Carrillo & M. A. Rodrigo, 2002. Assesstment of exotic fish impacts on water quality and zooplankton in a degraded semi-arid floodplain wetland. Aquat. Sci. 64: 76-86.

Arnell, N.W., Reynard, N.S. (1996). The effects of climate change due to global warming on river flows in Great Britain. Journal of Hydrology 183, 397–424.

Arnold, D.E., 1971. Ingestion, assimilation, survival, and reproduction by Daphnia pulex fed seven species of blue-green algae. Limnol. Oceanogr. 16, 906–920.

Bachmann, R. W., M. V. Hoyer & D. E. Canfield Jr, 1999. The restoration of Lake Apopka in relation to alternative stable states. Hydrobiologia 394: 219-232.

Badosa, A., Boix, D., Brucet S., Rocío L., Xavier D., 2006. Nutrients and zooplankton composition and dynamics in relation to the hydrological pattern in a confined Mediterranean salt marsh (NE Iberian Peninsula). Hydrobiologia 66: 513-522.

Barber, H.G., Haworth, E.Y., 1981. A guide to the morphology of the diatom frustule FBA Publications, U.K.: 112 pp.

Beklioglu, M., L. Carvalho & B. Moss, 1999. Rapid recovery of a shallow hypertrophic lake following sewage effluent diversion: lack of chemical resilience. Hydrobiologia 412: 5-15.

Beklioglu M., Burnak L., Ince O., 2000. Benthi-planktivorous Fish-Induced Low Water Quality of Lake Eymir before Biomanipulation. Turk J Zoo 24: 315-326.

Beklioglu, M., G. Altınayar & C. O. Tan, 2001. Role of water level fluctuations, nutrients and fish in determining the macrophyte-dominated clear, water states in five Turkish shallow lakes. Shallow Lake Wetlands: Ecology, Eutrophication and Restoration. International Workshop. 28-30 October 2001, Ankara, Turkey.

Beklioglu, M., O. Ince & I. Tuzun, 2003. Restoration of the eutrophic Lake Eymir, Turkey, by biomanipulation after a major external nutrient control I. Hydrobiologia 489: 93-105.

Beklioglu, M., G. Altinayar & C. O. Tan, 2004. Ülkemiz sıg göllerinin suiçi bitki gelisiminin belirlenmesinde, su seviyesi degisimi, besin tuzları ve bentik balık beslenmesinin degisen rolleri, pp:23-29. Sıg Göl Sulakalanlar: Ekoloji, Ötrofikasyon ve Restorasyon, M. Beklioglu (Ed.),pp:126. (in Turkish).

Beklioglu, M. Altınayar, G. Tan C.T. 2006a . Water level control over submerged macrophyte development in five shallow lakes of Mediterranean Turkey. Archive für Hydrobiologia 305:1-22.

Beklioglu, M. Susana Romo, Ifigenia Kagalou, Xavier Quintana, Eloy Bécares., 2006b. Shallow Mediterranean Lakes: Variation in the Theme – Workshop Conclusion. Hydrobiologia, (in press).

Beklioğlu, M., & C.O. Tan, 2006. Drought complicated restoration of a Shallow Mediterranean Lake by Biomanipulation. Archive für Hydrobiologia 166:535-556.

Blanco S., Romo S., Villena M.J. & Martínez S. 2003. Fish communities and food web interactions in some shallow Mediterranean lakes. Hydrobiologia. 506: 473-480.

Blindow, I., 1992. Long- and short-term dynamics of submerged macrophytes in two shallow eutrophic lakes. Freshw. Biol. 28: 15-27.

Blindow, I., A. Hargeby & G. Andersson, 1997. Alternative stable states in shallow lakes: what causes a shift? In Jeppesen, E., M. Søndergaard, M. Søndergaard & K. Christoffersen (eds), The structuring role of submerged macrophytes in lakes. Ecological Studies 131, Springer, New York: 353-360.

Boström, B., M. Jansson & C. Forsberg, 1982. Phosphorus release from lake sediments. Arch. Hydrobiol. Beih. Ergebn. Limnol. 18: 5-59.

Bottrell, H. H., A. Duncan, Z. M. Gliwicz, E. Grygiereg, A. Herzig, A. Hillbricht-Ilkowska, H. Kurasawa, P. Larsson & T. Weyleleuska, 1976. A review of some problems in zooplankton production studies. Norw. J. Zool. 24: 419-456.

Branco, C.W. C., Aguiro, T., Esteves, F.A., Caramaschi, E. P., 1997. Food sources of the Teleost *Eucinostomus aganteus* in two coastal lagoons of Brazil. Stud. Neutrop. Fauna & Environm. 32: 33-50.

Breukelaar, A.W., E.H.R.R. Lammens, J.G.P.K. Breteler, and I. Tatrai. 1994. Effects of benthivorous bream (Abramis brama) and carp (Cyprinus carpio) on sediment resuspension and concentrations of nutrients and chlorophyll a. Freshwat. Bio. 32: 113-121.

Brook, A.J., John, D.M., Whitton B.A., 2005. The freshwater Algal Flora of the British İsles Cambridge University Press, 702 pp.

Brönmark, C. & Weisner S.E.B. (1992) Indirect effects of fish community structure on submerged vegetation in shallow, eutrophic lakes: an alternative mechanism. Hydrobiologia, 243/244, 293–301.

Burks R.L., Lodge D.M., Jeppesen E. & Lauridsen T.L., 2002. Diel horizontal migration of zooplankton: costs and benefits of inhabiting littoral zones. Freshwater Biology, 47, 343–365.

Burnak, S. L., 1998. Pelagic Trophic Interactions: Clear versus Turbid in Lake Mogan and Lake Eymir. MSc Thesis. Middle East Technical University.

Burnak, S. L. & M. Beklioglu, 2000. Macrophyte-dominated clearwater state of Lake Mogan. Turk. J. Zool. 24: 305-313.

Canfield, D. E. Jr., J. V. Shireman, D. E. Colle, W. T. Haller, C. E. Watkins & M. J. Maceina, 1984. Prediction of chlorophyll-a concentrations in Florida Lakes: Importance of aquatic macrophytes. Can. J. Fish.Aquat. Sci. 41: 497-501.

Carpenter, S.C., Kitchell, J.F., Hodgson, J.R., 1985. Cascading trophic interactions and lake productivity. Bioscience 35, 634–639.

Carpenter, S.R., R.C. Lathrop, P. Nowak, E.M. Bennett, T. Reed and P.A. Soranno. 2005. The ongoing experiment: Restoration of Lake Mendota and its watershed. Pp. 236-256 in J.J. Magnuson, T.K. Kratz and B.J. Benson (eds.), Long-Term Dynamics of Lakes in the Landscape. Oxford University Press, London, England.

Carvalho L and Moss B, 1999. Climate sensitivity of a lowland acid lake. Freshwater Biology, 42, 585-591.

Chaney, A. & E. P. Morbach, 1962. Modified reagents for the determination of urea and ammonia. Clin. Chem. 8: 130-132.

Coops, H., M. Beklioglu & T.L. Crisman, 2003. The role of water-level fluctuations in shallow lake ecosystems – workshop conclusions.
Hydrobiologia 506-509: 23-27.

Coops, H., T.J. Vulink and E.H. van Nes, 2004. Managed water levels and the expansion of emergent vegetation along a lakeshore. Limnologica 34: 57-64.

Cottingham, K.L., S.R. Carpenter, and A. St. Amand. 1998. Responses of epilimnetic phytoplankton to experimental nutrient enrichment in three small seepage lakes. Journal of Plankton Research 20:1889-1914.

DeMott, W.R., 1999. Foraging strategies and growth inhibition in five daphnids feeding on mixture of a toxic cyanobacterium and green alga. Freshwater Biol. 42, 263–274.

DeMott, W.R., Gulati, R.D., Van Donk, E., 2001. Daphnia food limitation in three hypereutrophic Dutch lakes: evidence for exclusion of largebodied species by interfering filaments of cyanobacteria. Limnol. Oceanogr. 46, 2054–2060.

Deneke, R. & B. Nixdorf, 1999. On the occurrence of clear-water phases in relation to shallowness and trophic state: a comparative study. Hydrobiologia 408/409: 251–262.

Dent L. C., Gumming G.S., Carpenter S., 2002. Multiple states in river and lake ecosystems. Phil. Trans. R. Soc. Lond. B 357: 635–645.

Drenner R.W., Hambright K.D., 1999. Review: Biomanipulation of fish assemblages as a lake restoration technique. Arch. Hydrobiol. 146(2): 129-165.

DSI, 1993. Mogan Gölü Limnolojik Etüt Raporu. Technical Report. T.C. Bayındırlık ve İskan Bakanlıgı Devlet Su İsleri Genel Müdürlügü Isletme ve Bakım Dairesi Baskanlıgı, Ankara: 212 pp. (in Turkish).

Dugan, P. J., 1994. Wetlands management: A critical issue for conservation in Africa, p 1-8. In: T. Matiza and H. N. Chabwela (eds). Wetlands conservation conference for Southern Africa. Proceedings of the Southern African development co-ordination conference held in Gaborone, Botswana, 3-5 June 1991. IUCN, Gland, Switzerland.

EIE, 2004. Mogan ve Eymir Gölleri havzasının hidrometeorolojik özellikleri. Technical Report. Turkish General Directorate of Electrical Power, Resources Survey and Development Administration, Ankara: 159 pp. (in Turkish).

Eighmy, T. T. & P. L. Bishop, 1989. Distribution and role of bacterial nitrifying populations in nitrogen removal in aquatic treatment systems. Water Research 25: 947-955.

Eriksson, P. G. & S. E. B. Weisner, 1997. Nitrogen removal in a waste water reservoir: the importance of denitrification by epiphytic biofilms on submerged vegetation. J. Environ. Qual. 26: 905-910.

Fernandez-Alaez M., Fernandez-Alaez C., Be´cares E., Valentın M., Goma J. & Castrillo P., 2004. A 2-year experimental study on nutrient and predator influences on food web constituents in a shallow lake of north-west Spain. Freshwater Biology, 49, 1574–1592.

Frey D. G.,1993. The penetration of cladocerans into saline waters. Hydrobiologia 267: 233-248.

Fulton III, R. S. and H. W. Paerl. 1987. Toxic and inhibitory effects of the blue-green alga *Microcystis aeruginosa* on herbivorous zooplankton. Journal of Plankton Research 9: 837-855.

Fulton, R. S. & Paerl, H. W.,1988. Effects of the blue-green alga Microcystis aeruginosa on zooplankton competitive relations. Oecologia 76, 383–389.

Gafny, S. & A. Gasith, 1999. Spatially and temporally sporadic appearance of macrophytes in the littoral zone of Lake Kinneret, Israel: taking advantage of a window of opportunity. Aquat. Bot. 62: 249-267.

Garcia, P.R., Nandini, S., Sarma, S.S.S., Valderrama, E.R., Cuesta, I., Hurtado, M.D., 2002. Seasonal variations of zooplankton abundance in the freshwater reservoir Valle de Bravo (Mexico). Hydrobiologia 467: 99-108.

Geldiay, R., 1949. Çubuk barajı ve Eymir Gölü'nün makro ve mikro faunasının mukayeseli incelenmesi. Ankara Üniversitesi Fen Fakültesi Mecmuası, Cilt 2. (in Turkish).

Ghadouani, A., Pinel-Alloul, B., Prepas, E.E., 2003. Effects of experimentally induced cyanobacterial blooms on crustacean zooplankton communities. Freshwater Biol. 48, 363–381.

Glibert, P. M., 1988. Primary productivity and pelagic nitrogen cycling. In T. H. Blackburn & J. Sorensen (eds). Nitrogen cycling in coastal marine environments. Scope 33, J. Willey & Sons 3-31.

Gliwicz, Z. M., 1985. Predation or food limitation: an ultimate reason for extinction of planktonic cladoceran species. Archiv für Hydrobiologie, Beihefte Ergebnisse der Limnologie 21: 419-430.

Gliwicz, Z. M., 2003. Between hazards of starvation and risk of predation: the ecology of offshore animals. In: Kinne O (ed) Excellence in ecology, Book 12. International Ecology Institute, Oldendorf/Luhe.

Gliwicz, D. S. & Lambert W., 1990. Food thresholds in Daphnia species in the absence and presence of bluegreen filaments. Ecology 73: 910-926.

Golterman, H., L. Clymo & M. A. M. Ohnstad, 1978. Methods for chemical and physical analyses of freshwaters. 2nd edition. Blackwell Scientific Publishers, Oxford.

Görgün, E. D., 1994. The avifauna of Mogan Lake (Turkey). M.Sc. Thesis. Gazi University.

Granéli, W. & D. Solander, 1988. Influence of aquatic macrophytes on phosphorus cycling in lakes. Hydrobiologia 170: 245-266.

Grant, S.B, B.F. Sanders, J.A. Redman, J.H. Kim, R. Mrse, C.McGee, N.Gardiner, B.H. Jones, J. Svejkovsky, V. Leipzig, A. Brown (2001) "Generation of enterococci bacteria in a coastal salt water marsh and its impact on surf zone water quality", Environmental Science and Technology, 35:2407-2415.

Grimm, M. P., 1989. Northern pike (*Esox lucius L.*) and aquatic vegetation, tools in the management of fisheries and water quality in shallow waters. Hydrobiological bulletin 23: 59-65.

Gulati D. R., Van Donk E., 2002. Lakes in the Netherlands, their origin, eutrophication and restoration: state-of-the-art review. Hydrobiologia 478: 73–106.

Hairston, N.G., Smith, F.E. & Slobodkin, L.B., 1960. Community structure, population control, and competition. Am. Naturalist, 94: 421-425.

Hall J. C & Burns C. W., 2002. Mortality and growth responces of Daphnia carinata To increases in temperature and salinity. Freshwater Biology, 47: 451-458.

Halvorsen G, Dervo BK, Papinska K., 2004. Zooplankton in Lake Atnsjoen 1985-1997. Hydrobiologia: 521: 149-175.

Hansson, L. A., H. Annadotter, E. Bergman, S. F. Hamrin, E. Jeppesen, T. Kairesalo, E. Luakkanen, P. A. Nilsson, M. Søndergaard & J. Strand, 1998. Biomanipulation as an application of food chain theory: constraints, synthesis and recommendations for temperate lakes. Ecosystems 1: 558-574.

Haney, J.F., 1987. Field studies on zooplankton–cyanobacteria interactions. N. Z. J. Mar. Freshwat. Res. 21, 467–475.

Harris, GP 1986. Phytoplankton Ecology. Structure, Function, and Fluctuation. Chapman and Hall, London. 384 pp.

Havens, K. E., B. Sharfstein, M. A. Brady, T. L. East, M. C. Harwell, R. P. Maki & A. J. Rodusky, 2004. Recovery of submerged plants from high water stress in a large subtropical lake in Florida, USA. Aquatic Botany 78: 67-82.

Haslam, S. M., C. A. Sinker & P. A. Wolseley, 1982. British Water Plants. Reprinted from Filed Studies (1975), 4: 243-351.

Havens, K. E., B. Sharfstein, M. A. Brady, T. L. East, M. C. Harwell, R. P. Maki & A. J. Rodusky, 2004. Recovery of submerged plants from high water stress in a large subtropical lake in Florida, USA. Aquatic Botany 78: 67-82.

Hill, N. and PA Keddy. 1992. Predicting numbers of rarities from habitat variables: coastal plain plants of Nova Scotian lakeshores. Ecology 73: 1852-1859

Hillebrand H., Durselen C., Kirschtel. D., and Zahary T., 1999. Biovolume calculation for pelagic and benthic microalgae. Journal of Phycology, 35: 403-424.

Holling CS 1973. Resilience and stability of ecological systems. Annu Rev Ecol Syst 4: 1–24.

Irvine, K., B. Moss, & H. Balls, 1989. The loss of submerged plants with eutrophication II. Relationships between fish and zooplankton in a set of experimental ponds, and conclusions. Freshwat. Biol. 22: 89–107.

James, R. T. & K. E. Havens, 2005. Outcomes of extreme water level on water quality of offshore and nearshore regions in a larfe shallow subtropical lake. Arch. Hydrobiol. 163: 225-239.

Jensen, JP, E. Jeppesen, K. Olrik, and P. Kristensen. 1994. Factors on the Shift From Cyanobacterial to Chlorophyte Dominance in Shallow Danish Lakes. Can. J. Fish Aquat. Sci. 51:1692-1699.

Jeppesen, E., M. Sondergaard, and E. Mortensen. 1994. Nutrient Dynamics and Biological Structure in Shallow Fresh- Water and Brackish Lakes - Preface. Hydrobiologia 276: 15-30.

Jeppesen, E., J. P. Jensen, M. Søndergaard, T. Lauridsen, 1997. Top-down control in freshwater lakes: the role of nutrient state, submerged macrophytes and water depth. Hydrobiologia 342/343: 151-164.

Jeppesen, E., 1998. The ecology of shallow lakes – Trophic interactions in the pelagial. Doctor's dissertation (DSC). NERI Technical Report No.247. Silkebørg. pp. 358.

Jeppesen E., Søndergaard M., Kronvang B., Jensen J.P., Lars M. Svendsen & Lauridsen T.L., 1999. Lake and catchment management in Denmark. Hydrobiologia 395/396: 419–432.

Jeppesen, E., J. P. Jensen, M. Søndergaard, T. Lauridsen & F. Landikildehus, 2000. Trophic structure, species richness and biodiversity in Danish lakes: changes along a phosphorus gradient. Freshwater Biology 45: 201-213.

Jeppesen, E., J. P. Jensen & M. Søndergaard, 2002. Response of phytoplankton, zooplankton, and fish to re-oligotrophication: An 11 year study of 23 Danish lakes. Aquat. Ecosys. Health Manage. 5: 31- 43.

Jeppesen E., Søndergaard M., Jensen J.P. & Lauridsen T.L.,2003a. Recovery from eutrophication. In: Freshwater Management. Global versus Local Perspectives (Eds M. Kumagai & W.F. Vincent), pp. 1–16. Springer-Verlag, New York.

Jeppesen, E., J. P. Jensen, M. Søndergaard, K. S. Hansen, P. H. Møller, H. U. Rasmussen, V. Norby & S. E. Larsen, 2003b. Does resuspension, prevent a shift to a clear state in shallow lakes during reoligotrophicaion? Limnol. Oceanogr. 48(5): 1913-1919.

Jeppesen, E., M. Søndergaard , J. P. Jensen & T.L. Lauridsen, 2005. Restoration of Eutrophic lakes: a Global perspective. In press.

Jespersen, A-M. & K. Christophersen, 1987. Measurements of chlorophyll-afrom phytoplankton using ethyl alcohol as extraction solvent. Arch.Hydrobiol. 109: 445-454.

Kagalou, I. and Petridis, D.(2001), Seasonal variations of water quality parameters in a shallow lake (Lake Pamvotis) and effects on phytoplankton, zooplankton communnities, Hydrobiologia, (submitted for publication).

Kagalou, I. Papastergiadou E., Tsimarakis G., and Petridis, D., 2003. Evaluation of the trophic state of Lake Pamvotis Greece, a shallow urban lake. Hydrobiologia 506-509: 745-752.

Karapinar, B. & Beklioglu, M. 2006. Impact of sewage effluent diversion, biomanipulation and hydrology on nutrients mass balance in a shallow eutrophic Mediterranean lake Hydrobiologia, (in press).

Kasprzak P., Benndorf J., Mehner T. and Koschel R., 2002. Biomanipulation of lake ecosystems: an introduction. Freshwater Biology 47:2227-2281.

Kıraç, C. O., S. Karauz, A. Aykurt, M. Gürsan & O. Arihan, 1995. Birds of Lake Mogan. Bird Count Report (in Turkish).

King, A. J., A. I., Robertson & M. R., Healey, 1997. Experimental manipulations of the biomass of introduced carp (*Cyprinus carpio*) in billabongs. I. Impacts on water column properties. Mar. Freshwat. Res. 48: 435-443.

Kirk, K.L. 1991. Suspended clay reduces Daphnia feeding rate: Behavioural mechanisms. Freshwat. Biol. 25: 357–365.

Kisambira, M., 2003. The impacts of water level fluctuations on the development of submerged plants and their role in the clearwater state of Lake Mogan, Turkey. MSc. Thesis. METU, Ankara. p. 76.

Klinge, M. M., M. Grimm & S. Hosper, 1995. Eutrophication and ecological rehabilitation of Dutch lakes: presentation of a new conceptual framework. Wat. Sci. Technol. 31: 201–218.

Knowlton N. 1992. Thresholds and multiple stable states in coral reef community dynamics. Am Zool 32: 674–82.

Körner, S., 1997. Nutrient and oxygen balance of a highly polluted treated sewage channel with special regard to the submerged macrophytes.Acta Hydrochim. Hydrobiol. 25: 34-40.

Kubecka, J., Duncan, A., 1994. Diurnal changes in fish behaviour in lowland river monitored by a dual-beam echosounder. Fish. Res. 35, 55–63.

Lampert,W., 1987. Laboratory studies as zooplankton–cyanobacteria interactions. N. Z. J. Mar. Freshwat. Res. 21, 483–490.

Lampert, W., 1988. The relationships between zooplankton biomass and grazing. A review. Limnologica, 19: 11-20.

Lazzora, X., 1997. Do the trophic cascade hypothesis and classical biomanipulation approaches apply to tropical lakes and reservoirs?. Ver. Int. Verein. Limnol. 26: 719-730.

Leonard, P.R. and J. Cheever. 2005. Summer 2004 trap net monitoring of the fish community in the littoral zone at Brookwood Point and Rat Cove. In 37th Annual Report (2004). SUNY Oneonta Bio. Fld. Sta. SUNY Oneonta.

Lewontin, R. C., 1969. The meaning of stability. Brookhaven Symp Biol 22: 13–24.

Liao, H., C. L. Pierce, and J. G. Larscheid. 2004. Consumption dynamics of the adult piscivorous fish community in Spirit Lake, Iowa. North American Journal of Fisheries Management 24:890–902.

Loaiciga , H. A., J. B. Valdes, R. Vogel, J. Garvey & H. Schwarz, 1996. Global warming and hydrological cycles. J. Hydrol. 174: 83-127.

Lund H.C., Lund J.WG., 1998.Freshwater Algae their microscopic worl explored Biopress Ltd: 360 pages.

Mackereth, F.J., H. J. Heron & J. F. Talling, 1978. Water analyses: some methods for limnologists. Freshwater Biological Assoc. Scientific Publication No: 36.

May RM. 1977. Thresholds and breakpoints in ecosystems with a multiplicity of states. Nature 269: 471–77.

McDougal, R.L., L.G. Goldsborough, and B.J. Hann 1996. Responses of a prairie wetland to press and pulse additions of inorganic nitrogen and phosphorus: production by planktonic and benthic algae. Archiv fur Hydrobiologia (submitted).

McQueen, D. J., J. R. Post, and E. L. Mills. 1986. Trophic relationships in freshwater pelagic ecosystems. Canadian Journal of Fisheries and Aquatic Sciences 43:1571-1581.

Meerhoff M., Mazzeo N., Moss B. & L. Rodríguez-Gallego., 2003. The structuring role of free-floating versus submerged plants in a shallow subtropical lake. Aquatic Ecology 37: 377-391

Mehner, T., J. Benndorf, P. Kasprzak & R. Koschel, 2002. Biomanipulation of lake ecosystems: successful applications and expanding complexity in the underlying science. Freshwater Biology 47: 2453-2465.

Meijer, M. –L., M. W. de Haan, A. W. Breukelaar & H. Buiteveld, 1990. Is reduction of benthivorous fish an important cause of high transparency following biomanipulation in shallow lakes? Hydrobiologia 200/201: 301-315.

Meijer M.L., Jeppesen E., van Donk., Moss B., Scheffer M., lammens E., Van Nes E., Berkum J.A., De Jong G.J., Faafeng B.A. & Jensen J.P.,1994. Long-term responces to fish-stock reduction in small shallow lakes: Interpretation of five years results of four biomanipulation cases in the Netherlands and Denmark. Hydrobiologia 275/276: 457–466.

Meijer, M.-L., I. de Boois, M. Scheffer, R. Portielje & H. Hosper, 1999. Biomanipulation in shallow lakes in The Netherlands: an evaluation of 18 case studies. Hydrobiologia 408/409: 13-30.

Moss, B., 1990. Engineering and biological approaches to the restoration from eutrophication of shallow lakes in which aquatic plant communities are important components. Hydrobiologia 200/201: 367- 378.

Moss B., 1994. Brackish and freshwater shallow lakes — different systems or variations on the same theme?. Hydrobiologia 275/276: 1- 14.

Moss, B., J. Madgewick & G. Phillips, 1996. A guide to the restoration of nutrient-enriched shallow lakes, 177 pp.

Moss, B., 1998. Ecology of Freshwaters: Man and Medium and Past to Feature. 3rd edition. Blackwell Science, Oxford: 557 pp.

Moss, B., Stephen, D., Balayla, D., Becares, E., Collings, S., Fernandez-Alaez, C., M., Ferriol, C., Garcia, P., Goma, J., Gyllstrom, M., Hansson, L., Hietala, J., Kairesalo, T., Miracle, M., Romo, S., Rueda, J., Russell, V., Stahl-Delbanco, A., Svensson, M., Vakkilainen, K., Valentin, M., Van de Bund, W., Van donk, E., Vicente, E., Villena, M., 2004. Continental-Scale Patterns of Nutrient and Fish Effects on Shallow Wetland Lakes. Synthesis of a Pan-European International Mesocosm Experiment. Freshwater Biology, Vol. 49: 1633-1649.

Naselli-Flores, L., 2003. Man-made lakes in Mediterranean semi-arid climate: the strange case of Dr Deep Lake and Mr Shallow Lake. Hydrobiologia 506-509: 13-21.

Nõges, T. & P. Nõges, 1999. The effect of extreme water level decrease on hydrochemistry and phytoplankton in a shallow eutrophic lake. Hydrobiologia 408/409: 277-283.

Obali, O., 1978. Mogan gölü fitoplanktonunun nitesel ve nicesel olarak incelenmesi. Ph. D. Thesis. Ankara University, Ankara, Turkey (in Turkish).

Orcutt, J.D., Jr. and M.L. Pace. 1984. Seasonal dynamics of rotifer and crustacean zooplankton populations in a eutrophic, monomictic lake with a note on rotifer sampling techniques. Hydrobiology 119:73-80.

Ozesmi, U., 1999. Ecology and politics of rehabilitation: Mogan Lake wetland system, Ankara, Turkey. In Streever, W. (ed.), An International Perspective on Wetland Rehabilitation. Kluwer Academic Publishers, the Netherlands: 181-187.

Ozimek, T., R. D. Gulati & E. van Donk, 1990. Can macrophytes be useful in biomanipulation of lakes? The Lake Zwemlust example. Hydrobiologia 200/201: 399-408.

Padisak, J. 1992. Seasonal succession of phytoplankton in large shallow lake (Balaton, Hungary) - a dynamic approach to ecological memory, its possible role and mechanisms. Journal of Ecology, 80: 217-230.

Padisak, J., 1993. The influence of different disturbance frequencies on the species richness, diversity and equitability of phytoplankton in shallow lakes. Hydrobiologia 249: 135–156.

Parkos III J. J., Victor J. Santucci, Jr., and David H. Wahl 2003.Effects of adult common carp (Cyprinus carpio) on multiple trophic levels in shallow mesocosms. Can. J. Fish. Aquat. Sci. 60: 182–192.

Perrow M.R., Meijer M.L., Dawidowicz P. & Coops H., 1997. Biomanipulation in shallow lakes: state of the art. Hydrobiologia, 342/343, 355–365.

Phillips, G. L.,M. R. Perrow & J. H. Stansfield, 1996. Manipulating the fish-zooplankton interaction in shallow lakes: a tool for restoration. In S. P. R. Greenstreet & Tasker, M. L. (eds), Aquatic Predators and their Prey. Blackwell Scientific Publications Ltd., Oxford, England: 174–183.

Quirós, R. 1998. Fish effects on pelagic - trophic relationships in a comparative lake study. Hydrobiologia 361: 101-111.

Reddy, K. R. & W. F. De Busk, 1985. Nutrient removal potantial of selected aquatic macrophytes. J. Environ. Qual. 148: 459-462.

Rejas, D., K. Muylaert & L. De Meester, 2000. Trophic interactions within the microbial food web in a tropical floodplain lake (Laguna Bufeos, Bolvia). Revista de Biologia Tropical (in press).

Reinikainen, M., Ketola, M.,Walls, M., 1994. Effects of the concentrations of toxic *Microcystis aeruginosa* and an alternative food on the survival of *Daphnia pulex.* Limnol. Oceanogr. 39, 424–432.

Rennella, A.M. & R. Quiros, 2006. The effects of hydrology on plankton biomass in shallow lakes of the Pampa Plain. Hydrobiologia 556:181–191.

Repka, S., 1996. Inter-and intraspecific differences in Daphnia life histories in response to two food sources: the green alga Scenedesmus and the filamentous cyanobacterium Oscillatoria. J. Plankton Res. 18, 1213– 1224.

Repka, S., 1997. Effects of food type on the life history of Daphnia clones from lakes differing in trophic state. I. Daphnia galeata feeding on Scenedesmus and Oscillatoria. Freshwater Biol. 37, 675–683.

Reynolds, CS (1984) Phytoplankton periodicity: the interactions of form, function and environmental variability. Freshwater Biol., 14, 111–142.

Rocha, O., Matsumura-Tundisi, T., Espindola, E.L. G., Roche, K. F and Ritzeler, A. C., 1999. Ecological theory applied to reservoir zooplankton, pp. 457-476. In: J. G. Tundisi, & M. Straškraba (eds.), Theoretical reservoir ecology and its applications. Internat. Inst. Ecol./Backhuys Publishers, São Carlos.

Rohrlack, T., Dittmann, E., Borner, T., Christoffersen, K., 2001. Effects of cell-bound microcystins on survival and feeding of Daphnia spp. Appl. Environ. Microbiol. 67, 3523–3529.

Romero J. R., I. Kagalou, J. Imberger, D. Hela, M. Kotti, A. Bartzokas, T. Albanis, N. Evmirides, S. Karkabounas, J. Papagiannis & A. Bithava, 2002. Seasonal water quality of shallow and eutrphic Lake Pamvotis, Greece: Implications for restoration. Hydrobiologia 474: 91-105.

Romo S., Tongoren O., 1995. Multivariate analyses of phytoplankton assemblages of a polymictic hypertrophic lake. Archiv für Hydrobiologia 132:363-384.

Romo S., Miracle M., Villena M., Ruena J., Ferriol C., Vicente E., 2004. Mesocosm experiments on nutrient and fish effects on shallow lake food webs in a Mediterranean climate. Freshwater Biology 49: 1593-1607.

Romo S. & María-José Villena**,** 2005. Phytoplankton strategies and diversity under different nutrient levels and planktivorous fish densities in a shallow Mediterranean lake Journal of Plankton Research 27: 1273-1286.

Rooney, N. & J. Kalff, 2000. Inter-annual variation in submerged macrophyte community biomass and distribution: the influence of temperature and lake morphometry. Aquat. Bot. 68: 321-335.

Roughgarden, J., Pennington, T. & Alexander, S. (1994). Dynamics of the rocky intertidal zone with remarks on generalizations in ecology. Philos. Trans. R. Soc. Lond. Ser. B., 343, 79–85.

Sagrario M.A.G., Jeppesen E., Goma J., Søndergaard M., Jensen J.P., Lauridsen T., and Landkildehus F., 2005. Does high nitrogen loading prevent clerwater conditions in shallow lakes moderately high phosphorous concentration? Freshwater Biology, 50: 27-41.

Sanchez-Carrillo, S & M. Alvarez-Cobeleas, 2001. Nutrient dynamics and eutrophication patterns in a semi-arid wetland: the effects of fluctuating hydrology. Wat. Air Soil Pollut. 131: 97-118.

Sand-Jensen, K., 1989. Environmental variables and their effect on photosynthesis of aquatic plant communities. Aquat. Bot. 34: 5-25.

Sandsten, H., M. Beklioglu & O. Ince, 2005. Effects of waterfowl, large fish and periphyton on the spring growth of Potamogeton pectinatus L. in Lake Mogan, Turkey. Hydrobiologia 537: 239-248.

Schallenberg M., Hall J., Burns C. W., 2002. Climate changes alters zooplankton community structure and biodiversity in coastal wetlands. Limnology report, University of Otogo: 1-9.

Scheffer, M., 1989. Alternative stable states in eutrophic shallow freshwater systems: a minimal model. Hydrobiol. Bull. 23: 73-85.

Scheffer, M., S. H. Hosper, M. L. Meijer, B. Moss & E. Jeppesen, 1993. Alternative equilibria in shallow lakes. TREE 8: 275-279. Scheffer, M., 1998. Ecology of shallow lakes. Chapman & Hall, London: 356 pp.

Scheffer M, Rinaldi S, Gragnani A, Mur LR, van Nes EH., 1997. On the dominance of filamentous cyanobacteria in shallow, turbid lakes. Ecology 78:72–282.

Scheffer M, Carpenter SR, Foley JA, et al. 2001. Catastrophic shifts in ecosystems. Nature 413: 591–96.

Scheffer, M. & S. R. Carpenter, 2003. Catastrophic regime shifts in ecosystems: linking theory to observation. Trends Ecol. Evol. 18: 648-656.

Schriver, P., J. Bøgestrand, E. Jeppesen, & M. Søndergaard, 1995. Impact of submerged macrophytes on fish-zooplankton– phytoplankton interactions: large-scale enclosure experiments in a shallow lake. Freshwat. Biol. 33: 255–270.

Scourfield D. J., Harding J. P., 1966. A key to British Species of Freshwater Cladocera. FBA Publications, U.K. 55 pp.

Shapiro, J., 1995. Lake restoration by biomanipulation, a personel view. Environ. Rev. 3: 83-93.

Smith D. G., 2001. Pennak's Freshwater invertebrates of the United States. John Willey & Sons, Inc, Canada, 638 pp.

Smith, V. H. 1983. Low nitrogen to phosphorus favors dominance by bluegreen algae in Lake Phytoplankton. Science 221: 669-671.

Sommer, F., Stibor, H., Sommer, U. & Velimirov, B., 2000. Grazing by mesozooplankton from Kiel Bight, Baltic Sea, on different sized algae and natural size fractions. Mar. Ecol. Prog. Ser., 199, 43-53.

Sprules, W.G., 1977. Crustacean zooplankton communities as indicators of limnological conditions: an approach using principal component analysis. J. Fish. Res. Board Can. 34, 962–975.

Sprules,W.G., 1984. Towards an optimal classification of zooplankton for lake ecosystem studies. Verh. Internat. Verein. Limnol. 22, 320–325.

Stemberger, R.S., Lazorchak, J.M., 1994. Zooplankton assemblage responses to disturbance gradients. Can. J. Fish. Aquat. Sci. 51: 2435–2447.

Stemberger, R.S., Miller, E.K., 1998. A zooplankton-N:P-ratio indicator for lakes. Environ. Monitor. Assess. 51, 29–51.

Sterner, R. W. 1989. The role of grazers in phytoplankton succession, p 107-170. In: Sommer U (ed) Plankton ecology: succession in plankton communities.

Sun, J. & Liu D., 2003. Geometric models for cell biovolume and surface area. Journal of Plankton Research, 25, 1331-1346.

Sunaga, T., Verani, J. R., 1997. The fish community of four lakes. In: Tundisi, J. G. & Sauji J., (eds), Limnological Studies on the Rio Doce Valley Lakes, Brazil, pp. 359-369. Brazilian Acad. Sci. Uni. of San Paulo and Center for Water Research and Appl. Ecol., Sao Carlos SP.

Søndergaard, M. & B. Moss, 1997. Impact of submerged macrophytes on phytoplankton in shallow freshwater lakes. In Jeppesen, E., M. Søndergaard, M. Søndergaard & K. Christoffersen (eds), The structuring role of submerged macrophytes in lakes. Ecological Studies 131, Springer, New York: 115-132.

Søndergaard M., Jensen J.P. & Jeppesen E.,1999. Internal phosphorous loading in shallow Danish lakes. Hydrobiologia, 408/409, 145–152.

Takamura N, Otsuki A, Aizaki M, Nojiri Y., 1992. Phytoplankton species shift accompanied by transition from nitrogen dependence to phosphorus dependence of primary production in lake Kasumigaura, Japan. Arch Hydrobiol 124:129–48.

Talling, J. F., 2001. Environmental controls on the functioning of shallow tropical lakes. Hydrobiologia 458: 1-8.

Tan, C. O., 2002. The roles of Hydrology and nutrients in alternative equilibria of two shallow lakes of Anatolia, Lake Eymir and Lake Mogan: using monitoring and modelling approaches. MSc. Thesis. METU, Ankara, p.144.

Tan. C. O. and Beklioglu, M. 2005. Catastrophic-like shifts in Shallow Turkish Lakes: A Modelling Approach. Ecological Modelling, 183:425-434.

Tan, C. O. and Beklioglu, M. 2006. Modeling Complex Nonlinear Responses of Shallow Lakes to Fish and Hydrology using Artificial Neural Networks. Ecological Modelling. 196:183-194.

Tanyolaç, J. & M. Karabatak, 1974. Mogan Gölü'nün biolojik ve hidrolojik özelliklerinin tesbiti. TUBITAK VHAG-91, Ankara, Turkey. (in Turkish).

Ter Braak, C. J. F., 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. Ecology 67, 1167-1179.

Ter Braak, C. J. F., and P. F. M. Verdonschot., 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. Aquatic Sciences 57:255-289.

Teschner M., 1995. Effects of salinityon the life history and fitness of *Daphnia magna*: variability within and between populations.Hydrobiologia,307: 33-41.

Timms, R. M. & B. Moss, 1984. Prevention of growth of potentially dense phytoplankton by zooplankton grazing, in the presence of zooplanktovorous fish, in a shallow wetland ecosystem. Limnol. & Oceanogr. 29(3): 472-486.

Vakkilainen K., Kairesalo T., Hietala J. et al., 2004. Response of zooplankton to nutrient enrichment and fish in shallow lakes: a pan-European mesocosm experiment. Freshwater Biology, 49, 1619–1632.

Van der Molen D.T. & Portielje R., 1999. Multi-lake studies in the Netherlands: trends in eutrophication. Hydrobiologia, 408/409: 359-365.

Van Donk, E., R. D. Gullati, A. Iedama & J. T. Meulemans, 1993. Macrophyte related shifts in the nitrogen and phosphorus contents of the different trophic levels in a biomanipulated shallow lake. Hydrobiologia 251: 19-26.

Van Geest, G. J., H. Wolters, F. C. J. M. Roozen, H. Coops, R. M. M. Roijackers, A. D. Buijse & M. Scheffer, 2005. Water-level fluctuations affect macrophyte richness in floodplain lakes. Hydrobiologia 539: 239-248.

Van Nes, E. H., M. Scheffer, M. S. Van Den Berg, and H. Coops. 2002. Dominance of Charophytes in Eutrophic Shallow Lakes - When Should We Expect It to Be an Alternative Stable State? Aquatic Botany 72, no. 3-4: 275-96.

Van Tongeren E. F.,Van Liere L. R., Gulati R. D., Postema G. & Bouse Winkel De Bruyn P., 1992. Multivariate analysis of the plankton communities in the Lousdrechct Lakes: relation with the chemical and physical environment. Hydrobiologia 233:105-117.

Vanni M. J. & G. J. Temte. 1990. Seasonal patterns of grazing and nutrient limitation of phytoplankton in a eutrophic lake. Limnol. Oceanogr. 35: 697-709.

Vijverberg, J., Boersma, M., Van Densen, W.L.T., Hoogenboezem, W., Lammens, E.H.R.R. & Mooij, W.M., 1990. Seasonal variation in the interactions between piscivorous fish, planktivorous fish and zooplankton in a shallow eutrophic lake. Hydrobiologia 207: 279-286.

Vollenveider, R. A., 1975. Input-output models; special reference to the phosphate loading concept in limnology. Schweiz Z. Hydrol. 37: 53-84.

Von Elert, E. and Wollfrom, T. 2001. Supplementation of cyanobacterial food with polyunsaturated fatty acids does not improve growth of Daphnia. Limnol. Oceanogr. 46: 1552–1558.

Wallsten, M. & P. O. Forsgren, 1989. The effects of increased water level on aquatic macrophytes. J. Aquat. Plant Manag. 27: 32-37.

Walz, N. 1997. Rotifer life history strategies and evolution in freshwater plankton communities. In B. Streit, T. Statdler, and C. M. Lively (eds.), Evolutionary ecology of freshwater animals, pp. 119–149, Birkha¨user Verlag, Basel.

Warner, D. M. 1999. Alewives in Otsego Lake, NY: a comparison of their direct and indirect mechanisms of impact on transparency and chlorophyll *a.* Occas. Pap. No.32. SUNY Oneonta Bio Fld. Sta., SUNY Oneonta.

Wetzel, R. G. & G. E. Liekens, 1991. Limnological analyses. 2nd edition. Springer-Verlag, New York: 391 pp.

Wetzel, R. G., 1983. Limnology. 2nd edition. Saunders College Publishing: 753 pp.

Wetzel, R. G., 1990. 1990, Detritus, macrophytes and nutrient cycling in lakes. Mem. Ist. Ital. Idrobiol., 47: 233-249.

Wetzel, R.G. 2001. Limnology. Lake and River Ecosystems. Third Ed. Academic Press, San Diego: 1006 pp.

Wium-Andersen, S., 1987. Allelopathy among aquatic plants. Arch. Hydrobiol. Bull. 18: 157-158.

Zalidis, G Stamatiadis, S Takavakoglou, V Eskridge, K Misopolinos, N., (2002) Impacts of agricultural practices on soil and water quality in the Mediterranean region and proposed assessment methodology. Agric. Ecosyst. Environ. 88, 137-146.