THE ROLE OF HYDRAULIC LOADING AND NUTRIENTS IN THE ECOLOGY OF MEDITERRANEAN SHALLOW LAKES

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

BY

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IN PARTIAL FULFILLMENT OF THE REQUIREMENTS
FOR
THE DEGREE OF DOCTOR OF PHILOSOPHY
IN
BIOLOGY

MAY 2016

Approval of the thesis:

THE ROLE OF HYDRAULIC LOADING AND NUTRIENTS IN THE ECOLOGY OF MEDITERRANEAN SHALLOW LAKES

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ABSTRACT

THE ROLE OF HYDRAULIC LOADING AND NUTRIENTS IN THE ECOLOGY OF MEDITERRANEAN SHALLOW LAKES

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May 2016, 176 pages

In this Ph.D.-research three approaches were used to study the relationship between external nutrient loading, hydraulic loading and in-lake nutrient processes in shallow lakes under different climate conditions: a space-for-time mesocosm experiment on a climate gradient across Europe; time series analysis of long-term water and nutrient budgets of Lakes Eymir and Mogan; a combination of catchment and lake modelling of Lake Mogan under the projected climate conditions for 2020-2090.

The mesocosm experiments showed that nutrient retention in lakes depended on external nutrient loading but plant and phytoplankton biomass production played an important role and were influenced by climate. Phytoplankton production increased with temperature, but warmer temperatures also led to lower water levels which favoured macrophyte development in shallow mesocosms leading to higher nutrient retention. Both the time series analysis and the modelling projections showed the sensitivity of shallow lakes to changes in precipitation and evaporation, leading to periods with low water levels and a risk of drying out. During wet years with high water levels nutrient concentrations were low and determined by external nutrient loading in winter and spring. During dry years, external loading in winter and spring was lower, due to reduced runoff, but evaporative water loss and internal loading from the sediment during summer caused peaks in lake nutrient concentrations. The modelling approach showed that under a future drier and warmer climate, nutrient concentrations can increase, even when external loading decreased.

Keywords: Water level fluctuation; eutrophication; arid region; nutrient retention; lake model

ÖZ

HİDROLİK YÜKLEME VE GÖL BESİN TUZUNUN AKDENİZ SIĞ GÖL EKOLOJİSİNDEKİ ROLÜ

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Mayıs 2016, 176 sayfa

Bu doktora çalışmasında farklı iklim koşullarına maruz kalan sığ göllerdeki dış kaynaklı besin tuzu yüklemesi, hidrolik yükleme ve göl içi besin tuzu süreçleri arasındaki ilişkiler üç farklı yaklaşım kullanılarak çalışılmıştır: Avrupa iklim kuşağı boyunca zaman yerine mekan yaklaşımı kullanılan mezokozm deneyleri; Eymir ve Mogan Gölleri'nin uzun zamanlı su ve besin tuzu bütçelerinin zaman serisi analizleri; Mogan Gölü'nün, havza ve göl modeli kullanılarak, 2020-2090 yılları arası öngörülen iklim koşullarında modellenmesi.

Mezokozm deneyleri, göllerdeki besin tuzu tutulumunun dış kaynaklı besin tuzu yüklemesine bağlı olduğunu ve aynı zamanda iklim koşullarından etkilenen bitki ve fitoplankton biyokütle artışının da besin tuzu tutulumunda önemli bir role sahip olduğunu göstermiştir. Fitoplankton büyümesi sıcaklıkla artış göstermiş fakat sıcaklık artışı aynı zamanda düşük su seviyelerine neden olarak, sığ

mezokozmlarda makrofit gelişimini arttırmış ve yüksek besin tuzu tutulumuna yol açmıştır. Zaman serisi analizleri ve modelleme projeksiyonları sığ göllerin yağış ve buharlaşmaya karşı hassas olduğunu ve düşük su seviyesi ve kuruma riskinin olduğu dönemlere yol açacağını göstermiştir. Su seviyesinin yüksek olduğu bol yağışlı yıllarda besin tuzu derişiminin daha düşük olduğu ve bahar ve kış aylarındaki dış kaynaklı besin tuzu yüklemesine bağlı olarak değişiklik gösterdiği görülmüştür. Kurak geçen yıllarda ise azalan su miktarı sebebiyle kış ve bahar aylarındaki dış kaynaklı besin tuzu yüklemesi düşük sevivede olmasına rağmen yaz aylarında gerçekleşen buharlaşma ve iç besin tuzu yüklemesi göl besin tuzu derişiminde önemli bir artışa sebep olmuştur. Modelleme yaklaşımı gelecekte daha kuru ve sıcak iklim şartlarında dış kaynaklı besin tuzu yüklemesinde azalma gerçekleşse dahi besin tuzu derişiminin artabileceğini yol açabileceğini göstermiştir.

Anahtar Kelimeler: Su seviyesi değişimi, ötrofikasyon; kurak bölge; besin tuzu tutulumu; göl modeli

To my wife and son

ACKNOWLEDGEMENTS

I am very proud and happy that, 9 years after writing the acknowledgements in my master thesis, I have the chance to write another acknowledgments section in this PhD thesis.

I first and foremost want to thank my supervisor Prof. Dr. Meryem Beklioğlu for accepting me as a student in her lab, for providing all the valuable opportunities in these past 5 years and for her guidance and support. I also want to express my gratitude to Prof. Dr. Erik Jeppesen for his help and guidance with my work and for providing his valuable comments on my papers during his busy schedule.

I am also glad I had the opportunity to work with Dr. Boris Jovanovic on the titanium oxide mesocosm experiment, with the REFRESH mesocosm experiment team, with the MARS heatwave experiment team in Denmark and with Dr. Dennis Trolle on the SWAT and PCLake modelling.

I also want to acknowledge the TÜBITAK 2215 program and the EU-MARS-project that provided financial support during my PhD.

I want to thank all the students in the lab over these past five years. Special thanks to Arda, Deniz, Zeynep, Serhan, Esra and Jennifer for their help with the fieldwork and the laboratory work and to Tuba, Eti, Nur, Şeyda, Gizem, İdil and Nihan for their help with my many, many questions.

I want to thank my parents for all their understanding and support, my wife Tuna, without whom I would never have started this PhD, for her love and support for all these years. And thanks to my son for cheering up my days during the last months of writing up this thesis.

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CHAPTER I

INTRODUCTION

There are millions of natural lakes in the world and the majority, both in number and in total area, are small lakes (smaller than 10 km^2) (Downing et al., 2006). Many of these lakes are shallow and in several regions of the world, shallow lakes are the most abundant (Scheffer, 1998). Shallow lakes and their associated wetlands are rich in biodiversity and have a high ecological value, but many are threatened by human activities (agriculture, sewage discharge, recreation, urbanisation, etc.) and climate change (Jeppesen et al., 2009).

Shallow lakes can exist in either two states: an oligotrophic clear-water state dominated by submerged macrophyte vegetation and a eutrophic turbid-water state dominated by phytoplankton (Scheffer et al., 1993). Several hydrological and morphological parameters can determine the trophic state of a lake, but nutrient loading has been identified as a key factor. Switches between both states are possible and triggered by changes in nutrient loading, but a hysteresis effect, caused by mechanisms in both states that favour stabilizing processes and counteract a shift, has been observed (Mooij et al., 2009). This hysteresis effect means that a shift from the clear-water state to the turbid-water state will occur at a higher nutrient loading threshold than the reverse shift back to the clear-water state. The eutrophic turbid-water state is associated with algae blooms, low water clarity, low dissolved oxygen levels, toxicity and low ecological value (Carpenter et al., 1999). Therefore several studies have examined the relationship between nutrient loading and eutrophication (Jeppesen et al., 2005).

Conducted research so far focussed mostly on the northern temperate lakes in Europe and North America, while lakes in the Mediterranean climate zone have been underrepresented (Alvarez Cobelas et al., 2005; Beklioglu et al., 2007). This is problematic because in the semi-arid conditions of the Mediterranean zone the eutrophication risk even increases, since the water balance has a profound influence on the nutrient dynamics (Coops et al., 2003; Özen et al., 2010). Specific research on Mediterranean lakes is hence required, as the knowledge for northern temperate lakes cannot be just transposed to the Mediterranean (Alvarez Cobelas et al., 2005).

The expected impacts of climate change provide an additional reason for further research on the relationship between hydraulic loading and nutrient concentrations in Mediterranean lake ecosystems. In the cold temperate climate, climate warming is expected to stabilize the eutrophic turbid-water state of lakes and destabilize the macrophyte dominated clear-water state (Mooij et al., 2005; Jeppesen et al., 2009). This is caused by higher temperatures, higher winter precipitation and increased rainfall intensity, which cause higher runoff and higher external nutrient loading (Jeppesen et al., 2009, 2011). Climate change will also affect lake stratification, oxygen levels, the phosphorus (P) release from the sediment, community structure and the occurrence of algae blooms (Gyllström et al., 2005; Mooij et al., 2005; Meerhoff et al., 2007a; Paerl & Huisman, 2008)

Considering the increased eutrophication risk for Mediterranean lakes under the semi-arid conditions and given that the expected effects of climate change will further enhance the problem, a detailed study of the relationship between hydraulic loading and the nutrient status of Mediterranean lakes is required with special focus on the conditions during years with low water levels.

1.1. Lake eutrophication

The relationship between nutrients and primary production was established by *Liebig's Law of the Minimum* which states that the growth of plants is limited by the nutrient that is least available in the environment compared to the demands of the plant (von Liebig, 1855). Given that for terrestrial plants inorganic N and P are commonly limiting the growth in an agricultural setting, these nutrients were also expected to limit the primary production in aquatic ecosystems (Lewis & Wurtsbaugh, 2008).

The term eutrophication or eutrophic was first used by Weber (1907) to describe different German peat bogs and later by Naumann (1919) to describe so-called *high food lakes* with high algae biomass. Attempts were made to classify lakes as either eutrophic or oligotrophic (low food) based on the morphometry of lakes, the oxygen depletion of the hypolimnion and the communities of benthic macroinvertebrates, but it soon proved that the existing continuum of lakes cannot be fit into narrowly defined classes (Schindler, 2006).

Eutrophication took on a different, less descriptive meaning in the second half of the 20th century when the effects of human-induced excessive nutrient loading to lakes became apparent. This anthropogenic eutrophication was defined as the process by which aquatic ecosystems are made more eutrophic through an increase in nutrient supply (Smith et al., 1999). First, eutrophication was considered a natural process wherein lakes evolve from an oligotrophic, deep state to an eutrophic, shallow state as sediment and nutrients build up (Rodhe, 1969), but research on pristine Alaskan lakes in Glacier Bay National Park showed that lakes in a natural setting actually become more oligotrophic over time as the available mineral nutrient sources become depleted (Engstrom et al., 2000). However, pristine isolated lakes are rare currently, as most lakes are influenced by human activities (Smith et al., 2006). Now eutrophication is generally considered a human-induced phenomenon and only a few exceptional lakes are considered to be naturally eutrophic, e.g. the West Midland Meres in England (Moss et al., 1994).

Besides the fact that eutrophication has in most cases an anthropogenic cause and is therefore a deviation from the natural state, lake eutrophication is also related with a number of effects, some of which are considered unwanted for both ecological and economic reasons (Smith, 2003):

- increased productivity and biomass of phytoplankton and suspended algae;
- shifts in phytoplankton composition to bloom-forming species, which can produce toxins or are less effectively consumed by grazers;
- increased productivity and biomass and changes in species composition of periphyton;
- changes in productivity, biomass and species composition of aquatic plants;
- threats to endangered aquatic species;
- decrease in water column transparency;
- taste, odour and filtration problems in drinking water supplies;
- depletion of deep water oxygen;
- decreases in perceived aesthetic value of the water body;
- negative economic impacts, including decreased property values and reduced recreational use.

When the symptoms of eutrophication became widespread in the mid-20th century early measures adopted by water managers involved the use of algaecides to fight the symptoms, rather than treating the cause of the problem (Schindler, 2006). On the Madison symposium on eutrophication held in 1969 the focus rested on vitamins, growth hormones, amino acids and trace elements as the prime suspects of the eutrophication problem and, despite the knowledge of terrestrial ecosystems, N and P were only briefly mentioned (National Academy of Sciences, 1969). The landmark conference held by the American Society of Limnology and Oceanography in 1971 focussed on nutrients as the main cause of eutrophication (Likens, 1972), but the discussion was dominated by carbon limitation versus P limitation, while N was barely considered (Howarth & Marino, 2006).

Lab experiment with bioassays had shown that phytoplankton growth could be enhanced by C addition and was not influenced by N or P, supporting the claim that C might be limiting primary production (Likens, 1972). But whole-lake experiments showed that addition of phosphate led to eutrophic conditions with high phytoplankton biomass, while addition of N and C had no such effect. Schindler (1974) therefor proposed the *Phosphorus limitation paradigm* that P is the primary nutrient limiting phytoplankton production in lakes and that reducing only the P loading to the lake is sufficient to counteract eutrophication. N and C were assumed not to be limiting since uptake from the atmosphere by diffusion and N fixation would compensate for any temporary in-lake shortages (Schindler, 1977).

1.1.1. Phosphorus limitation paradigm

P is a mineral nutrient that is abundant in large quantities on Earth, but also needed in large quantities in the metabolism of living organisms (Moss, 2012). The P cycle is shown in Figure 1.1 (Heathwaite et al., 1996). The main source of P in natural catchments is rock weathering (Reynolds & Davies, 2001). The weathered P binds tightly with clay particles in the soil where it can be assimilated by organisms through special enzymes and fine roots. Only a relatively small fraction will leach out of the soil and will be transported to the water bodies (Heathwaite et al., 1996). The atmospheric part of the P-cycle is very limited compared to the water phase.

The P compounds reaching the lakes can be divided between dissolved forms and particulate forms (Correl, 1998). Dissolved forms are soluble organic P bound to organic compounds or inorganic phosphate-ions (PO_4^{3-}, HPO_4^{2-}, $H_2PO_4^-$) that make up the soluble inorganic or reactive phosphorus (SRP). Only orthophosphate (PO_4^{3-}) can be directly used by algae, bacteria and plants in their metabolism (Correl, 1998).

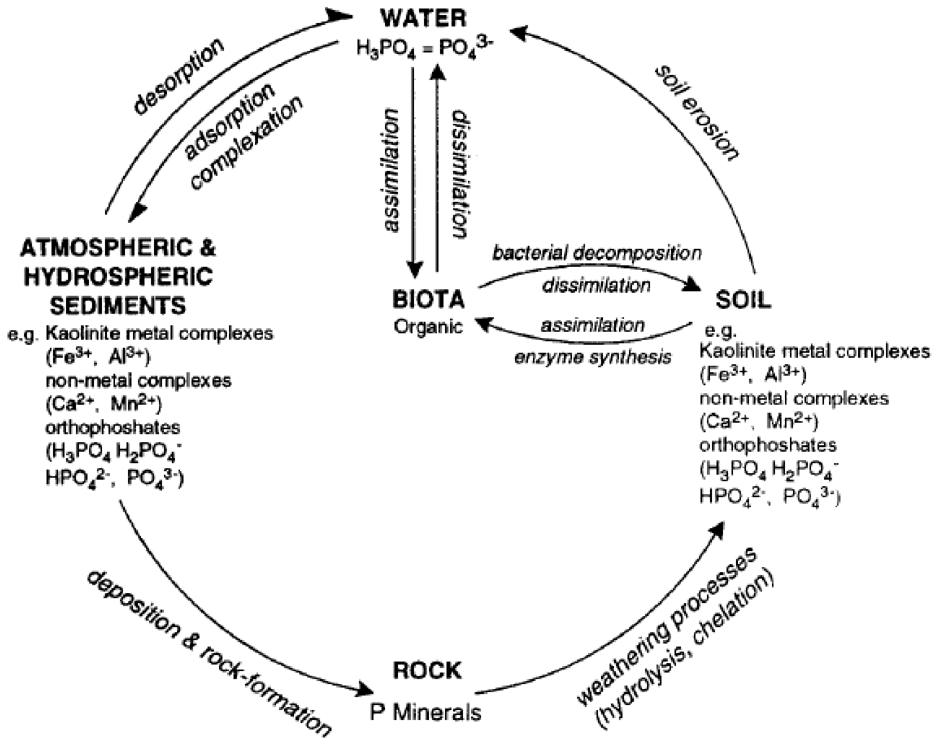


Figure 1.1: Phosphorus cycle (taken from Heathwaite et al., 1996).

Particulate phosphorus (PP) that enters a lake can be transformed to dissolved forms and orthophosphate. PP that does not dissolve in the water column will be deposited in the sediment. P in the sediment also occurs in dissolved form in the pore water. Although these dissolved forms only constitute less than 1% of P in the sediment, they are very important since the exchange with the overlying water layer occurs largely in the dissolved form.

The concentrations of dissolved phosphate in the pore water increases due to the release of organically bound P during the mineralisation of organic matter. This decomposition depends on oxygen as an electron acceptor in the top sediment layer, but deeper in the sediment under anoxic conditions nitrate, iron, manganese or sulphate are used instead (Moss, 2010a). The depletion of oxidised substances during decomposition also lowers the redox potential and weakens the binding of P to inorganic compounds, further enhancing the release of phosphate to the pore water. Phosphate can be released from the pore water to the overlying water layer, but P can also bind again with particulates (binding with oxidised iron and manganese) before reaching the water layer. In oligotrophic lakes with a well-oxygenated water layer, the sediment will act as a P sink throughout the year with the binding intact, but in eutrophic and anoxic conditions the binding is weakened and the sediment becomes a source of P, causing internal P loading (Correl, 1998).

Human activities have severely altered the availability of P in the natural environment and in order to reduce the influx of P in aquatic systems these anthropogenic sources, point and diffuse sources, have to be controlled. The point sources are mainly sewage discharges and runoff from urban areas, while diffuse sources are related with the use of fertilizers in agriculture or the conversion of natural areas to urban or arable land (Smith et al., 1999).

Point sources can be controlled by limiting the use of phosphate in detergents and by treating sewage effluent before discharge in surface water and this has been done with a certain degree of success (Schindler, 2006). Control of diffuse sources is more complicated to enforce and execute. Examples of possible measures are: changes in agricultural practices (restriction of fertilizer use), reforestation, restoration of wetlands and riparian areas and restoration of channelized streams (Jeppesen et al., 2009).

Even when the external P loading is reduced, a delay in the decrease of in-lake P concentrations can be observed, due to release from the sediment (Søndergaard et al., 2003; Jeppesen et al., 2007b). Lakes that were subjected to high nutrient loading for long periods will retain large P stocks in the sediment that are released after external loading is reduced. A new equilibrium between the lake nutrient concentration and the loading forms generally after 10-15 years for P and around 5 years for N, depending on the lake hydraulic retention time and lake depth (Jeppesen et al., 2005).

Although the control of P loading showed results in mitigating the effects of eutrophication (Beklioglu et al., 1999; Jeppesen et al., 2005, 2007a), the paradigm that P was the only nutrient limiting algae biomass was met with criticism (Sterner, 2008). Lewis & Wurtsbaugh (2008) stated 5 objections to the P limitation paradigm: (1) they question the emphasis on the direct correlation between TP and chlorophyll-a since both are components of phytoplankton biomass and therefore always correlated when measured at the same time; (2) any correlation between nutrients and biomass should consider the fractions of the nutrients actually available to the organism instead of total nutrient fractions; (3) the emphasis on whole-lake experiments overlooks the fact that replication in these experiments is low and (4) that not all conducted whole-lake experiments supported P limitation; (5) N fixation by cyanobacteria is not sufficient to offset any limitation of external N loading and therefore N limitation can occur. With the P limitation paradigm under scrutiny, other nutrients returned to the picture.

1.1.2. Nitrogen limitation in lakes

While early research on eutrophication of lake ecosystems was entirely focussed on P, the research in marine biology during the 80’s and 90’s shifted independently to the role of N as a regulating factor of eutrophication in estuaries and coastal areas (Howarth & Marino, 2006). After the need for N control to counteract eutrophication in marine environments was established, lake ecologists had renewed interest in N as a nutrient limiting algae biomass.

A meta-analysis by Elser et al. (2007) showed that addition of N and P together increased algae biomass significantly more than addition of either element separately and that N and P limitation are both equally strong in freshwater ecosystems. Therefore calls have been made to include N reduction measures in eutrophication policy (Gonzalez-Sagrario et al., 2005; Jeppesen et al., 2007b; Conley et al., 2009), although certain authors claim that N limitation has never been supported by whole-lake experiments and is only a temporary side-effect of over enrichment with P (Schindler, 2009).

Contrary to P, N has a large atmospheric stock as N_2-gas (Figure 1.2 from Heathwaite et al., 1996), but this inert gas cannot be used directly by most organisms in their metabolism. Only prokaryotic N fixers are able to process N_2, but only in microaerophilic or anaerobic conditions. Several mechanisms of N fixation exist (Moss, 2010a):

- prokaryotes in symbiosis with fern of vascular plants (Fabaceae and alder);
- free-living prokaryotes in the soil and in water that create microaerophilic conditions close to the enzymes responsible for N-fixation;

- some cyanobacteria fix N in special cells (heterocysts) where the part of the photosynthetic mechanism that emits O_2 is missing;
- prokaryotes living in microaerophilic sediments;
- soil N-fixing bacteria living in soil crumps with oxygen-consuming aerobic bacteria on the outside;
- bacterial N-fixers in special plant root nodules where host cells consume oxygen.

During fixation N_2-gas is converted to N in amino groups in the cell structure of the organism, which becomes available to other organisms through predation or decomposition. The amino groups are converted to ammonium (NH_4^+) by bacteria (ammonification or mineralisation) which can be converted by other bacteria to nitrite (NO_2^-), and finally nitrate NO_3^- (nitrification). Ammonium and nitrate dissolve in water and are available for most living organisms to be metabolised to amino groups. Under anoxic conditions nitrate can be used as an oxidising agent by heterotrophic and autotrophic bacteria and converted to N_2O and eventually N_2-gas (denitrification). Denitrification can contribute significantly to the loss of N in lake systems (Jensen et al., 1990).

The natural sources of bioavailable N are the conversion of N_2-gas by N-fixing prokaryotes, which is estimated as 120-440 TgN per year, and to a lesser extent by lightning (10 TgN a^{-1}) (Vitousek et al., 1997). But in the last decades N-fixation by human-induced processes altered the availability of N to a large extent. In total, an estimated 140 TgN a^{-1} is produced every year through anthropogenic processes such as N-fixation for fertilizer production (80 TgN a^{-1}), the use of N-fixing crops in agriculture replacing natural vegetation (40 TgN a^{-1}) and fossil fuel combustion which frees up fixed-N that had been locked in geological layers for a long time (20 TgN a^{-1}). Additionally an extra 70 TgN a^{-1} of N is mobilized from long-term storage pools through biomass burning, land use changes (deforestation) and drainage of wetlands (Vitousek et al., 1997).

All this extra input to the natural environment causes a build-up of N in terrestrial soils. It then percolates to the groundwater, moves into the surface waters and enters the atmosphere through volatilisation of ammonia (Smith et al., 1999). This added N leads to eutrophication problems in lakes, especially in combination with high P loads, for example in Lake Taihu in China (Paerl et al., 2010), while the atmospheric deposition of nitric acids can lead to acidification (Vitousek et al., 1997).

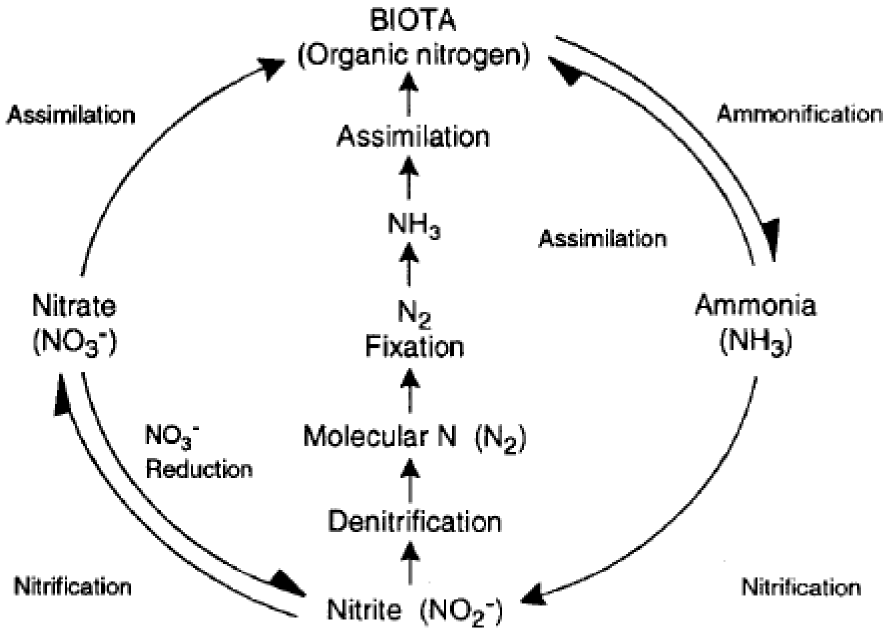


Figure 1.2: Nitrogen cycle (from Heathwaite et al., 1996)

Although several studies showed that both N and P can play a role in limiting primary production and phytoplankton biomass in lakes (Havens et al., 2001; North et al., 2007; Abell et al., 2010; Xu et al., 2010), the discussion over the role of N in eutrophication still continues, as is evident from the exchanges between Schindler et al. (2008), Scott & McCarthy (2010, 2011), Paterson et al. (2011). The discussion mainly focussed on the question if N fixation by cyanobacteria is large enough to offset any reduction in external loading and if such a reduction therefore leads to cyanobacterial dominance (Kosten et al., 2009a; Scott & McCarthy, 2010).

Another approach to the N versus P debate is to consider the ratio of total nitrogen to total phosphorus (TN:TP). Downing & McCauley (1992) examined the relationship between TN:TP and the trophic status of the lake and found that the TN:TP ratio is high in oligotrophic lakes but declines in a curvilinear way when TP concentrations increase, which could be largely explained by the high TN:TP ratio in the loading coming from undisturbed, natural catchments compared to the low TN:TP ratio of sewage or urban runoff (Downing & McCauley, 1992). The TN:TP ratio also determines if a lake is N,P or co-limited (Guildford & Hecky, 2000). Chl-a concentrations are strongly correlated with TP, but weakly with TN and phytoplankton biomass in lakes showed signs of N limitation when TN:TP < 20 (molar) or < 9 (mass), but signs of P-limitation when TN:TP > 50 or 22 (mass), with both nutrients potentially limiting in intermediate conditions (Guildford & Hecky, 2000). Many lakes have been found in the interval where both N and P can be limiting (Sterner, 2008).

1.1.3. Eutrophication and alternative state theory

When measures were put in place to reduce the external loading of initially P, but in some cases also N, it was observed that certain lakes failed to convert back from the eutrophic state to a clear water oligotrophic state. Scheffer et al. (1993) proposed that shallow lakes can exist in two alternative stable states: a clear-water state dominated by aquatic vegetation and a turbid-water state dominated by phytoplankton. The dynamics of the lake system depend on the interaction between aquatic plants and turbidity, because aquatic plants tend to keep the water clear by reducing sediment resuspension, inhibiting phytoplankton through nutrient competition, allelopathy and providing refuge for zooplankton and young piscivores, but turbid water prevents the growth of aquatic plants because of light limitation and competition by phytoplankton.

Several factors can play a role in shifting the lake from a clear-water state to a turbid-water state or vice versa (Schindler, 2006), with nutrient loading playing an important role in the appearance or disappearance of aquatic vegetation (Scheffer et al., 1993). At low nutrient loading only the state with aquatic plants is stable since they can take up nutrients from the sediment while the nutrient concentration in the water body is too low to support high phytoplankton biomass. When the nutrient loading increases, phytoplankton can achieve higher biomass, but the vegetation present will stabilize the clear-water state until at high nutrient loadings a small disturbance (e.g. resuspension of sediment by wind action, grazing by waterfowl or sudden increase in water level) causes the aquatic plants to disappear and the system switches rapidly to a turbid state dominated by phytoplankton. In order to restore the system back to the macrophyte-dominated state a severe reduction in nutrient loading is required (hysteresis effect), since the shading by the phytoplankton biomass prevents the recolonization of the lake by plants and since the turbid-water conditions favour zooplanktivorous fish that reduce zooplankton biomass and thus grazing of phytoplankton (Scheffer et al., 1993). Because of this hysteresis effect the alternative stable states theory can explain why lakes can show a delay in the shift from turbid to clear, even when lake nutrient concentrations are reduced.

Mesocosm experiments and multi-lake dataset analyses have suggested a TN concentration > 2 mg L^{-1} and a TP concentration > 0.10-0.20 mg L^{-1} as a threshold for the shift from the macrophyte dominated state to the turbid state (Gonzalez-Sagrario et al., 2005; Kosten et al., 2009b). Carpenter et al. (1999) on the other hand found that while some lakes showed the hysteresis effect in response to an increase or decrease in external P loading, other lakes were immediately restorable or are irreversible after they shift to the turbid-stable state. Based on an analysis of a dataset of Danish lakes, Jeppesen et al. (2007b) also argued that the hysteresis effect is not always as strong, with benthivorous fish biomass and resuspension quickly dropping and chl-a directly following the TP-concentrations in some lakes.

Scheffer & Van Nes (2007) later expanded the theory in order to incorporate some exceptions that had been observed, including lakes that showed different stable states at different parts due to spatial heterogeneity, lakes that showed cyclical shifts between the different states and lakes where the shifts are very slow and that remain in a ghost stable state for many years. Scheffer & Van Nes (2007) also emphasized lake depth, lake size and climate as important influencing factors. Overall the stable-state theory is viewed as a valuable addition to the eutrophication theoretical framework and certain restoration measures related with manipulating the food web structure are based on it.

The principles of the trophic cascade established by Hairston et al. (1960) and Fretwell (1977) apply also to the freshwater ecosystems (Smith & Schindler, 2009) which means that in systems with 4 trophic levels piscivorous fish control zooplanktivorous fish, so that predation pressure on zooplankton (herbivores) is low and the grazing pressure on phytoplankton (primary producers) is high and their biomass suppressed. If the piscivorous level is not present (3 trophic levels) the grazing pressure on phytoplankton will be low due to regulation of zooplankton numbers by planktivorous fish and algae biomass will be high. This shows that in lakes with the same nutrient loading phytoplankton biomass, and thus the effects of eutrophication, can be different depending on the food web structure of the given lake (Carpenter et al., 1985). In the opposite direction, eutrophication also affects the food web by favouring phytoplankton over macrophytes, which provide refuge against predation for zooplankton and young piscivorous fish (Smith & Schindler, 2009).

Shapiro et al. (1975) proposed to use this knowledge of the impact of the food web structure on phytoplankton biomass to control the effects of eutrophication and to enhance a shift from the turbid-water state to the clear-water state. They argued that lakes are complex ecosystems and simply reducing the nutrient loading that initially caused the eutrophication will not reverse the changes to the food web. Furthermore it was observed that lakes responded slowly to a reduction in the external P loading. Altering the lake food web (biomanipulation) by stocking the lake with piscivorous fish (to establish 4 trophic levels) or removing zooplanktivorous fish (2 trophic levels) can break this resilience and push the lake to a clear-water state (Hansson et al., 1998).

The biomanipulation technique has since been used in a range of countries with varying degrees of success (Jeppesen et al., 2007a). Biomanipulation has also been applied on Lake Eymir in Turkey to aid shifting the lake to a clear-water state after an external loading reduction in 1995 (Beklioglu et al., 2003). Reducing the planktivorous fish stock enabled clear water conditions, although temporarily, as was evident from higher Secchi depth, lower chl-a and suspended solid concentrations and higher macrophyte coverage. A second biomanipulation resulted again in lower chl-a and SS but not in an increase in macrophyte coverage (Beklioglu & Tan, 2008).

Overall biomanipulation can assist in restoring lakes after eutrophication but the focus should remain on reduction of the external loading, since chl-a is strongly correlated with TP concentrations and long-term effects of biomanipulation efforts are rare (Jeppesen et al., 2007a).

1.2. N and P retention in lakes

When nutrients are transported to a lake a portion of the incoming amount will be retained in the lake as can be observed from the difference between the input and output flux (Vollenweider, 1976). These retention processes are important since they reduce the nutrient flux to the downstream surface waters and eventually the nutrient loading to the estuarine and marine ecosystems. The term retention has recently become under scrutiny, since the nutrients are not necessarily retained in the lake ecosystem but can also be lost to the atmosphere. The terms “nutrient loss” or “nutrient change” are therefore also used.

P retention is the sum of sedimentation of particulate P and uptake of dissolved phosphate by aquatic plants. Sedimentation of inorganically and organically bound particulate P will remove it from the water column and retain it in the sediment. This storage can be temporary, since the binding of inorganically bound P can weaken under anoxic conditions and since organically bound particulate P can be converted to dissolved forms during decomposition leading to P being released back to the water column (Correl, 1998). Particulate P can also be resuspended from the sediment by bioturbation or wind action and convert to dissolved forms in the water column. The release of these dissolved forms from the sediment constitutes the internal P loading of the lake, which can be an important source of P when external loading is low. This internal loading process is influenced by bioturbation, water temperature, iron concentrations in the sediment, pH, redox-potential, oxygen concentrations, mineralization and microbial processes (Søndergaard et al., 2003). P can also be lost from the water column through uptake by aquatic plants, but this process is considered small in comparison with the sediment related processes and the retention is only temporary since after the plants die off the P is released back to the water column after decomposition.

The retention of P is also related with the hydrological residence time as is evident from the model proposed by Vollenweider (1976). From $TP_{lake} = TP_{in} / (1 + tw^{0.5})$ it can be deduced that with a water retention time of 1 year the lake TP concentration will be 50% of the inflow concentration with the rest being retained. Koiv et al. (2011) made a meta-analysis of P retention in 54 lakes and found the following trends:

- P retention positively correlated with external P loading;
- proportion of P-loading that is retained is independent from the amount of P-loading;
- hydraulic residence time (HRT) positively correlated with P retention, this relationship is stronger in large lakes (> 25 km^2);
- the bigger the relative depth the higher the P retention, but only for large lakes (>25 km^2) and HRT< 0.3 yr;
- in stratified lakes relative depth inversely correlated with P retention capacity.

N retention is the result of sedimentation of organically bound forms, uptake by aquatic plants and denitrification which produces N_2-gas that is permanently lost to the atmosphere (Wetzel, 2001). Because of the importance of N retention, Harrison et al. (2009) used a spatial model to estimate the global N retention in lakes and reservoirs. They calculated the total N removal in lentic ecosystems (lakes and reservoirs) as 19.7 TgN a^{-1}, about a third of total N loading to freshwater systems (Bouwman et al., 2005). Small lakes (smaller than 50 km^2) remove 9.3 TgN a^{-1}, more than three times as much as larges lakes (3.7 TgN a^{-1}) and almost half of all lentic systems. The removal of N in lakes and reservoirs is comparable to that in streams and rivers and 4 times larger than in estuaries (Harrison et al., 2009).

N retention is influenced by external N loading, hydraulic residence time, mean lake depth and water temperature (Windolf et al., 1996; Kaste & Dillon, 2003). External loading shows the strongest correlation with N retention (Saunders & Kalff, 2001a). Hydraulic residence time is positively correlated with retention since longer hydraulic residence times allow longer time for denitrification. Lake depth on the other hand is negatively correlated with retentions since in shallow lakes the contact between the sediment and the water column is longer. Water temperature has a positive correlation with retention through its influence on denitrification rates.

Denitrification contributes the largest part to N retention in most lakes, followed by sedimentation while uptake by aquatic plants is often considered negligible (Saunders & Kalff, 2001b). Differentiation between the different mechanisms is made by direct measurement of sedimentation rates and N fluxes from sediment cores (Nowlin et al., 2005; Yang et al., 2005) and of uptake by aquatic plants (Kreiling et al., 2011), with denitrification assumed to be the remainder. Kreiling et al. (2011) measured assimilation of N by aquatic plants directly for a backwater lake of the Mississippi river and quantified it at 7-18% of N retention, while 82% of N retention was due to denitrification.

Denitrification is influenced by several factors (Saunders & Kalff, 2001a). Water temperature positively influences the activity of the denitrifying bacteria, while nitrate and organic matter availability can potentially limit denitrification rates. Higher denitrification rates have been observed in more reduced sediments, showing the effect of the redox potential, while lakes with aquatic plants generally show larger denitrification rates. Aquatic plants contribute directly to N retention by assimilating N in their biomass, and indirectly by influencing denitrification and sedimentation rates (Saunders & Kalff, 2001a). Dense plant beds stabilize the water column, favouring sedimentation over resuspension and macrophytes also create an ideal environment for denitrification by increasing the supply of organic carbon and nitrate to denitrifying bacteria. Aquatic plants also release oxygen into the sediment through their roots which increases the sediment redox potential and enhance denitrification (Saunders & Kalff, 2001b).

Several studies quantified N retention as a percentage of N input to the lake and the relative contribution of denitrification, sedimentation and uptake by macrophytes, although the latter is often ignored. (Saunders & Kalff, 2001a) found that lakes retained 34% of N inputs in a selection of European and North-American lakes. Denitrification was found to be responsible for 63% of TN retention. (Kaste & Dillon, 2003) calculated N retention for Norwegian and Canadian lakes as ranging from 6-24% and 50-87% respectively. A study of an Estonian lake showed a N retention of 40-90% (Nõges, 2005). A small British lake showed low N retention of 10% probably explained by the short residence time and lack of submerged macrophytes (Irfanullah & Moss, 2008). Windolf et al. (1996) calculated for 16 Danish lakes that on average 43% of added N is removed based on calculations for 69 lakes and 77% of N lost due to denitrification (Jensen et al., 1990). Rzychon & Worsztynowicz (2008) observed an increase of the N retention in two Polish mountain lakes over ten years from 28-55% to 44-76%, related with an increased plant biomass. The research of the

Lower Lakes in Australia showed a very small N retention on the other hand (6%) owed to high N fixation in the semi-arid lakes (Cook et al., 2010).

1.3. Eutrophication and climate change

Global mean temperature has risen by 0.74 °C between 1906 and 2005 and the rate of climate warming has almost doubled to 0.13 °C per decade over the last 50 year compared to the rate over the last 100 years (Trenberth et al., 2007). Precipitation patterns are less straightforward with precipitation increasing between 1900 and 2005, but downward trends in the tropics since the 1970's. Independent from rainfall amount heavy precipitation events increased, while also droughts have become more common especially in the tropics and subtropics (Trenberth et al., 2007).

The 4th assessment report by the Intergovernmental Panel on Climate Change (IPCC) published in 2007 projected a warming of 0.2°C per decade over a range of emission scenarios, with warming expected to be greatest over land and at most high northern latitudes, and least over the Southern Ocean and parts of the North Atlantic Ocean. Weather extremes like heat waves, droughts but also heavy precipitation events are thought to become more frequent. Increases in the amount of precipitation are deemed very likely in high latitudes, while decreases are likely in most subtropical land regions (Meehl et al., 2007). The Mediterranean region is expected to be affected strongly by a changing climate with a significant decrease in seasonal precipitation, higher precipitation variability and a temperature increase of 0.1-0.4°C per decade (Sanchez et al., 2004).

These projected changes will affect lakes and other freshwater systems in all its aspects: physical, chemical and biological. Physical changes to the lakes can be attributed to the changes in rainfall amount and higher evaporation due to higher mean temperatures. Especially in Mediterranean climates lower precipitation combined with higher evaporation will lead to longer and more frequent drought periods, associated with lower water levels and several secondary effects on the chemical and biological parameters (Beklioglu et al., 2007). Changing climate patterns are also expected to affect the thermal stratification of lakes with potential lower thermoclines, longer stratification periods, lower oxygen levels and higher P release from the sediment and a shorter ice cover period during winter (Blenckner et al., 2007).

Changes in precipitation patterns will influence the nutrient loading coming from the catchment. In high latitudes higher nutrient loading is expected due to increased runoff (Andersen et al., 2006; Jeppesen et al., 2009, 2011), but since the volume of the runoff also increases the concentration of the inflow to the lakes may not change significantly compared to present levels. In lower latitudes decreased precipitation will lead to lower runoff volumes and lower nutrient loading, but this reduction may be offset by lower water levels and consequently higher concentrations in the lake water body (Jeppesen et al., 2009, 2011). Lower lake water levels are also expected to lead to higher salinity (Beklioglu & Tan, 2008).

All these changes in the physical and chemical structure of the lakes will influence the biotic components of the lake ecosystem. Aquatic plants will have higher primary production due to higher temperatures, lower water levels (Coops et al., 2003), and lower nitrate levels as result of higher denitrification rates (Jeppesen et al., 2007a). Also phytoplankton will show higher growth rates because of higher temperatures (French & Petticrew, 2007), higher internal P loading, longer growing seasons and higher external P and N loading (Jeppesen et al., 2007a).

Bloom-forming cyanobacteria are expected to thrive under warmer climate conditions due to their high temperature for optimal growth, the buoyancy capabilities of certain species that allow to benefit from longer stratification periods and due to higher nutrient concentrations (Paerl & Huisman, 2008). Zooplankton communities have been shown to be influenced by the mean temperature during the warmest winter month (Gyllström et al., 2005). Fish communities in northern climates will more resemble the fish communities in subtropical lakes with smaller body sizes, higher densities (but not always higher biomass), higher diversity, higher species richness, continuous reproductive events during the year, more omnivory and less piscivorous fish (Meerhoff et al., 2007a; Teixeira de Mello et al., 2009). Fish kills due to oxygen depletion as a result of warming and high nutrient concentration have also been observed in mesocosm-warming experiments (Moss, 2010b).

All these effects will affect the trophic cascade as the fish communities in a warmer climate are more associated with the aquatic plant beds in the littoral zone, who therefore act less a refuge for zooplankton (Meerhoff et al., 2007b). High predation on zooplankton leads to low zooplankton biomass and dominance of smaller species in a warmer climate. Phytoplankton will likely benefit from these changes in fish community structure and zooplankton densities and climate warming is expected to stabilize the phytoplankton-dominated turbid water state and destabilize the macrophyte-dominated clear water state, although the outcome depends on the interaction with the stabilizing effect of potentially lower water levels on macrophytes (Mooij et al., 2005).

Climate change can have profound effects on water level fluctuations, ranging from a seasonal to a multi-annual scale and especially in Mediterranean lakes hydrology plays an important role (Beklioglu et al., 2007; Jeppesen et al., 2015). Low water levels will enhance development of aquatic plants since larger parts of the lake may fall within the euphotic depth (depending on lake morphology), while rapid water level increases may impact aquatic plants negatively (Coops et al., 2003). On the other hand low water levels can lead to higher lake nutrient concentrations, due to the concentration effect (smaller volume, same nutrient loading) and due to higher internal loading (Özen et al., 2010), which will favour phytoplankton over submerged macrophytes. But at low water levels macrophytes have been shown to be resilient against high nutrient concentrations (Özkan et al., 2010; Bucak et al., 2012). To make the picture more complex, high water levels have also been associated with low phytoplankton biomass, independently from nutrient loading (Nõges et al., 2003), so the ultimate effect of water level fluctuations on the shifts between the turbid and clear water state is difficult to predict .

There is large interaction between the effects of expected climate change on lake ecosystems and the effects of lake eutrophication due to the human activities that have been observed over the past decades (Mooij et al., 2005). This interaction will lead to an exacerbation of the eutrophication problem already observed and current thresholds and mitigation measures will not be strict enough to prevent further eutrophication. Stronger measures will have to be taken, including further reduction of external P and N loading coming from point and diffuse sources through improved sewage treatment, optimized fertilizer use, less intensive farming; restoration of wetland area and buffer zones around the lakes and limiting the use of water resources for irrigation in the southern regions (Jeppesen et al., 2007a).

1.4. Studying eutrophication by analysing nutrient budgets

To study lake eutrophication it is important to have a full understanding of the in- and outputs of nutrients to the lake and how changes in these fluxes will affect the lake nutrient concentrations. Nutrient budgets in which a mass balance is constructed for an element (mainly N and P) deliver valuable insights in the nutrient dynamics of a lake. In mass balance calculations, inputs to the lake from different sources (inflows, precipitation, groundwater, dry atmospheric deposition), outputs out of the lake through outflows and groundwater, changes in lake nutrient concentrations and internal processes are summed. N fixation is difficult to quantify as is often ignored in the budget calculations, leading to an underestimation of N inputs. Nutrient budgets are generally connected with water balances where water gains through inflows, groundwater, and precipitation are balanced with water losses through outflows, groundwater and evaporation. Retention of N and P can then be deducted from the nutrient budgets as the remaining part of N and P fluxes in the lake.

The first nutrient budget for an aquatic ecosystem was constructed by Johnstone (1908) in order to quantify nutrient fluxes in the North Sea, but mass balances of nutrients wouldn't become a common tool to study eutrophication until the 1960's. The research of (Vollenweider, 1968) started the use of nutrient budgets calculated as mass balances to study the eutrophication problem. Vollenweider analysed the existing literature on the eutrophication of lakes at that time in a study for the Organization for Economic Cooperation and Development (OECD) and empirically deduced the link between total P (and in some cases total N) inputs to the lake and lake eutrophication. In the years following Vollenweider's report, nutrient budgets were constructed for lakes in North-America (Johnson & Owen, 1971) and Europe (Jonasson et al., 1974; Jansson, 1979), mainly for northern temperate and artic lakes.

After the role of N in lake eutrophication was established studies also focussed on N and N retention. (Windolf et al., 1996) calculated a 4-year mass balance for 16 Danish lakes to quantify N retention and its seasonal variation, while (James et al., 2011) studied the N dynamics of Lake Okeechobee in Florida (USA).

Romero et al., (2002) created a nutrient budget for N and P for a lake in Greece in the Mediterranean climate zone, and Cook et al. (2010) constructed nutrient budgets for two small connected lakes in the semi-arid climate zone of Australia. These two studies are rare examples of nutrient budgets for lake outside the northern temperate climate zone.

In order to study nutrient processes, nutrient budgets can also be constructed for mesocosm experiments (Olsen et al., 2015). Mesocosm experiments have the advantage that they allow to control certain parameters such as nutrient loading and water level, providing a means to test hypotheses about which factors influence nutrient retention.

1.5. Lake modelling studies

In contrast to statistical models, mechanistic models aim at simulating the different processes that go on in lakes, rather than just the empirical relationship between lake parameters (Reckhow & Chapra, 1999). The first mechanistic model was proposed by Sawyer (1947) with the purpose to determine inorganic nutrient thresholds for algae growth. Progress with mechanistic models was slow due to the complexity of the processes under investigation, but from the 60's and 70's onwards many models concerning lake eutrophication were developed (Reckhow & Chapra, 1999). These models differ widely in functional, hydrodynamic and spatial structure, with differences in included compartments, dimensions and processes (Mooij et al., 2010). Bryhn & Hakanson (2007) compared static and dynamic models in their ability to model lake eutrophication. They concluded that static models can predict in-lake P concentrations and TP retention in a wide range of lakes, but they cannot model internal P fluxes. Dynamic models, on the other hand, showed a lower prediction error, but require more specialized knowledge.

In this thesis PCLake will be used, which is a dynamic complex ecosystem model that includes both the abiotic and biotic components of a lake ecosystem (Janse, 2005). PCLake will be used in combination with SWAT, a catchment based mechanistic model able to simulate the relationship between land use and climate and the hydrological and nutrient dynamics of a lake catchment (Neitsch et al., 2011). The modelling approach to study the interaction between lake eutrophication and climate change has the advantage that a wide variety of scenarios can be tested under controlled conditions, which can enhance the understanding of the functioning of the lake and its catchment. And although modelling studies always represent a simplification of the situation in the field and their results should be interpreted with care, they provide a valuable addition to field measurements and direct experimentation.

1.5.1. Soil and Water Assessment Tool

SWAT was developed by Dr J. Arnolds for the Agricultural Research Service (ARS) of the US Department of Agriculture (USDA), aimed at analysing the impact of different land use forms and management practices on the hydrological, sediment and chemical dynamics of a hydrological catchment. The model simulates physical processes related with the water and nutrient cycle based on input data for the catchment. This allows the model to be used in research about the impact of land use changes and climate change. SWAT is a continuous time model that is aimed at studies of long-term impacts (Neitsch et al., 2011).

SWAT is a catchment-scale hydrological model designed to analyse the impact of different management practices on the hydrological and nutrient dynamics of a catchment. The model is physically based which means that it simulates physical processes associated with the water and nutrient cycle directly, based on inputs about climate, soil properties, topography, vegetation and land use. In this way the model can be used to analyse the impact of alternative input scenarios on the water quantity or quality in the catchment.

A SWAT-model of a catchment starts from the topography of the area under investigation, generally in the form of a Digital Elevation Model (DEM). Based on the topography SWAT will delineate the stream network in the area. SWAT defines the start point of a stream as a point that drains a minimum area equal to a threshold value that can be set by the user. Optionally, known information about the streams and rivers can be added to the model to refine the modelled network. Based on the DEM, SWAT will delineate the catchment as the total area in which all surface water flows to an user selected outflow point.

The catchment is further subdivided in subbasins, where a subbasin of a tributary is defined as the upstream area starting from the point where two tributaries meet. Each subbasin is further subdivided in *Hydrological Response Units* (HRU) based on the soil properties and land use classes. Basically all areas belonging to a certain combination of a soil and land use type are lumped together in an HRU and treated like one patch in the subbasin.

Based on this model of the catchment, SWAT will simulate the hydrological cycle. SWAT simulates hydrological processes like interception, infiltration, surface runoff and subsurface flow for each HRU and all the HRU's are summed up to calculate the outflow of the subbasins and finally the catchment. Also plant growth, erosion processes and nutrient cycles are calculated per HRU. Climatic parameters required by SWAT as input are daily precipitation, maximum and minimum air temperature, solar radiation, wind speed and relative humidity.

The hydrological cycle in SWAT is divided between a land phase, which encompasses the flow of the water and its nutrient and sediment loading overland to the stream network and the water phase, which encompasses the routing of the water through the stream network to the outflow. The hydrological cycle in SWAT is based on the following water balance equation:

$$SW_t = SW_0 + \sum_{i=1}^{t}(R_{day} - Q_{surf} - E_a - w_{seep} - Q_{gw}) \qquad (1)$$

with SW_t = the final soil water content

SW_0 = the initial soil water content

t = time in days

R_{day} = precipitation

Q_{surf} = surface runoff

E_a = evapotranspiration

w_{seep} = water entering the unsaturated zone of the soil

Q_{gw} = groundwater base flow

The driver of the catchment hydrology is the precipitation that is supplied as input to the model. The precipitation that enters the catchment can be intercepted by the vegetation, be lost as evapotranspiration, fall to the soil surface, infiltrate to the soil or flow overland as runoff. Runoff water will rapidly contribute to stream flow while infiltrated water can percolate to the groundwater or contribute to the stream flow as slow subsurface flow. Lakes can be included in SWAT as reservoirs and maximum lake area, maximum lake volume depth and outflow rates should be specified. Lake water levels are simulated based on inflows, precipitation, evaporation and groundwater seepage. Further information on the hydrological cycle in SWAT is given in Appendix A.

Plant growth is modelled in SWAT to account for the loss of water and nutrient from the root zone, transpiration and biomass production. Erosion processes are modelled per HRU with the Modified Universal Soil Loss Equation (MUSLE) that calculates sediment yield based on the amount of runoff. SWAT models the cycles of N and P in the catchment. The N cycle and P cycle as modelled in SWAT are shown in Figure 1.3 and Figure 1.4. N and P uptake by plants is modelled based on supply and demand, nitrate, organic N and soluble P can be transported through the surface runoff, subsurface lateral flow and groundwater percolation based on flow volumes and N-concentrations. P and organic N can also be transported with the sediment flow.

SWAT has been used for many modelling studies, including several in Turkey (Akiner & Akkoyunlu, 2012; Güngör & Göncü, 2013; El-Sadek & Irvem, 2014; Ertürk et al., 2014; Rouholahnejad et al., 2014; Nerantzaki et al., 2015).

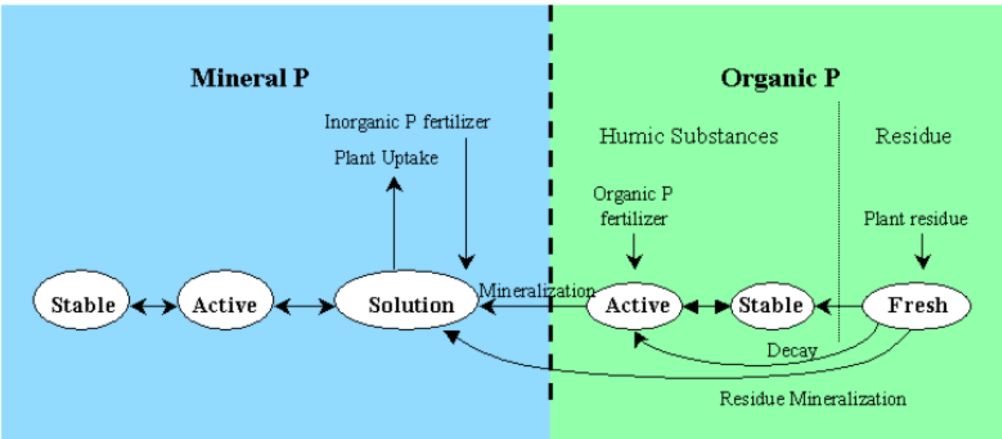


Figure 1.3: P cycle in SWAT (Neitsch et al., 2011).

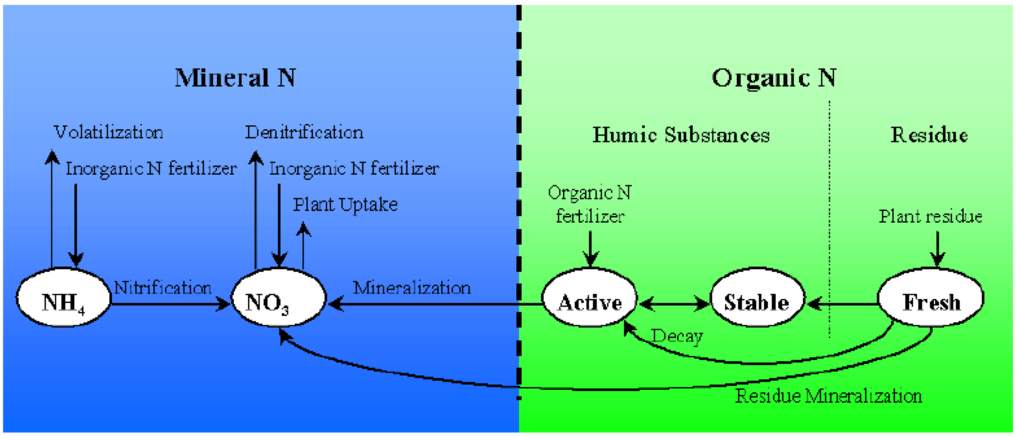


Figure 1.4: Nitrogen cycle in SWAT (Neitsch et al., 2011)

1.5.2. <u>PCLake</u>

PCLake is a dynamic ecosystem model that incorporates both the biotic and abiotic components of the lake ecosystem. In its first conception PCLake was developed to model the P loading and its effects on the hypertrophic Loosdrecht Lakes in the Netherlands (Janse & Aldenberg, 1990). This PCLoos-model was based on a model initially developed by (Kouwenhoven & Aldenberg, 1986) and later expanded by Aldenberg & Peters (1990).

PCLoos was further expanded in consecutive modelling studies of the Loosdrecht Lakes (Janse & Aldenberg, 1991; Janse et al., 1992). Version 2.5 of the model included 20 state variables and added chemical adsorption, improved modelling of sediment processes and minor changes in fish, zooplankton and zoobenthos growth equations. At this stage the model was calibrated using mini-models of the different compartments and using Bayesian statistics combined with a range-check procedure (Janse et al., 1992).

In Janse (2005) PCLake-model version 5.08 was presented. This version consisted of closed and dynamic P, N and Si-cycles and 24 state variables. Most state variables are modelled separately by dry-weight, N and P while detritus is also modelled as Si. CO_2 fluxes are not directly modelled. The model evaluation consisted of a sensitivity analysis, a calibration and an uncertainty analysis using a Bayesian likelihood-measure based on a 43-lake dataset (Janse et al., 2010). This approach deals with the uncertainty connected to the model structure and the identification of many unknown parameters values. The calibration was performed on a set of Dutch, Belgian, Irish and Polish lakes and validated on 9 extra Spanish and Danish lakes.

PCLake is a dynamic ecosystem model that incorporates both the biotic and abiotic components of the lake ecosystem. It also contains a large number of functional groups and can be used for both eutrophication studies as for climate change research. The PCLake model is an integrated lake model that is developed in the Netherlands to simulate the main nutrient and food web dynamics of a non-stratifying lake (Janse, 1997). The model was originally designed to study the effects of eutrophication of shallow lakes and specifically to investigate the shifts that occur between the clear-water state and the turbid-water state as a function of nutrient loading and other parameters.

The basic structure of the model consists of a mixed water column, the top layer of the sediment (10 cm) and the key biotic factors and abiotic factors. The model is meant for shallow non-stratifying lakes so no other horizontal or vertical dimensions are taken into account; only a wetland zone can be added. The structure of the model is presented in Figure 1.5.

The model is mathematically built as a number of coupled differential equations related with a series of state variables. These state variables are: water depth, inorganic matter, humus, detritus, inorganic nutrients (PO_4, adsorbed P, NH_4, NO_3, SiO_2), oxygen, phytoplankton (cyanobacteria, diatoms, small edible algae), submerged vegetation, zooplankton, zoobenthos, juvenile whitefish, adult whitefish, piscivorous fish and marsh vegetation. Most of these state variables are modelled separately as dry weight, P, N and in some cases silica (diatoms, detritus). These nutrient cycles are completely closed, except for in- and outputs and are independent of each other, which allows the C:N and C:P ratios of different trophic levels to be variable. The nutrient cycles are dynamic and the mass balance per element is checked after every time step.

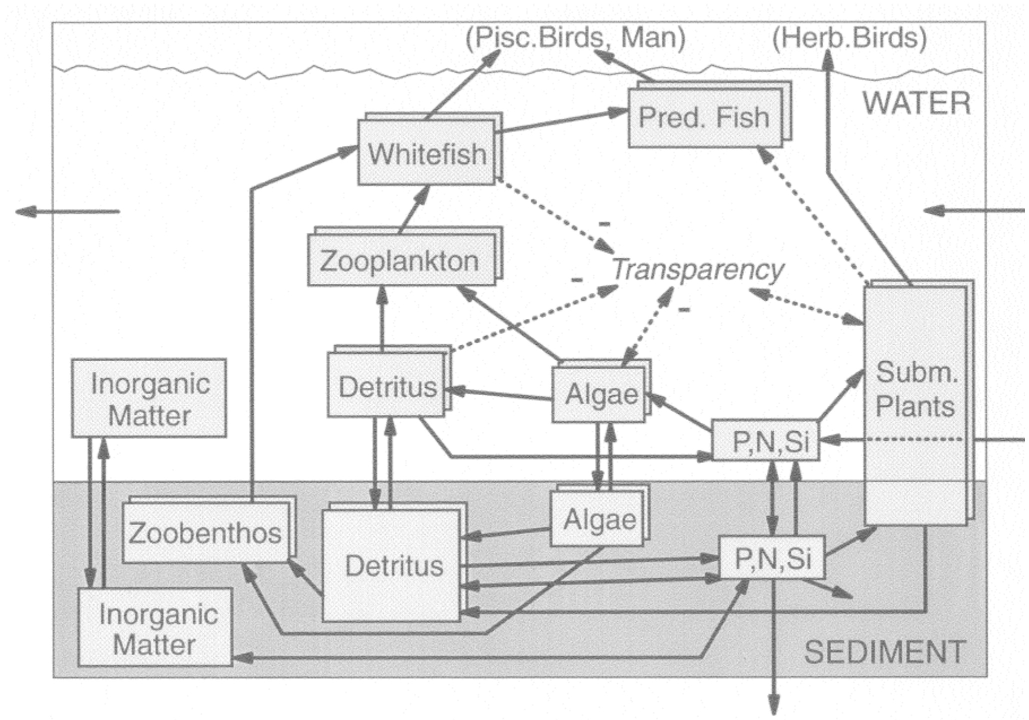


Figure 1.5: PCLake model structure (Janse, 1997): Arrows with solid lines represent mass fluxes; arrows with dotted lines represent empirical relations

PCLake requires the following input: mean depth, lake size (fetch), sediment characteristics, marsh area, water inflow and outflow, groundwater infiltration/seepage, climatic variables (precipitation, evaporation, wind speed, water temperature, radiation), water temperature, external nutrient and suspended matter loading, nutrient concentrations of the sediment and any information about lake management (fisheries, biomanipulation, dredging, mowing). Besides these inputs all state variables must be initialised for the starting conditions.

Based on these inputs PCLake models all the state variables for each time step, based on the mathematical equations that describe the physical, chemical and biological processes in the lake. Water depth is modelled based on the water balance. Sediment consists of inorganic matter, detritus, humus and pore water. Resuspension and settling are modelled based on size of the lake (wind action), sediment porosity and bioturbation, any net gain or loss of sediment is balanced out by burying of sediment to the deeper layers so that the sediment thickness stays constant. Mineralisation of detritus, adsorption of P to inorganic matter, P precipitation, nitrification, denitrification and exchanges of N and P between the pores and the water column are also incorporated.

Phytoplankton is divided in cyanobacteria, diatoms and small edible algae and the biomass is considered to be the result of primary production, respiration, mortality, settling, resuspension, grazing and transport. Aquatic vegetation is modelled as primary production, respiration and mortality, with a possibility to include bird grazing and mowing. The food web module comprises zooplankton, macrozoobenthos, whitefish (juvenile and adult) and predatory fish, with all animal groups basically modelled as the product of feeding, egestion, respiration, mortality and predation. Zooplankton is set to feed on phytoplankton and detritus with grazing pressure depending on seston concentration, filtering rate and food preference parameters. Zoobenthos feeds on sediment detritus and settled algae.

Fish predation is modelled according to the type-III response curve with juvenile whitefish feeding on zooplankton, adult whitefish on zoobenthos and piscivorous fish on all whitefish.

The primary goal of the PCLake model is to simulate the shifts between the alternative stable states of shallow lakes and to determine the critical loading levels that cause these shifts (Janse & Van Liere, 1995). In the early modelling of the Loosdrecht lakes PCLoos proved accurate in simulating the observed delayed response of the in-lake P concentration to a decrease in external loading (Janse et al., 1992). The model was further used to test different management scenarios that could further decrease the in-lake P-concentration, ranging from biomanipulation, deepening of the lake, stronger reduction of external loading and stocking with piscivorous fish (Janse et al., 1992).

After the successful modelling of the Loosdrecht Lakes, PCLake was applied used in the Netherlands in a range of studies (Janse & Van Liere, 1995; Janse et al., 1998, 2008; Van Puijenbroek et al., 2004; Sollie et al., 2008). PCLake was also applied to lakes outside the Netherlands. The study of the Lithuanian Lake Zuvintas focussed on the effects of external nutrient loading reduction and water level fluctuations as management tools to reduce in-lake nutrient concentrations (Stonevicius & Taminskas, 2007). The Russian Lake Shira was parameterized using PCLake to allow future investigation of management options (Prokopkin et al., 2010).

Although PCLake was initially developed to study lake eutrophication focusing on external nutrient loading as the trigger of a transition between stable-states (Janse, 2005), the model also has been used to investigate the effects of projected climate change. For the Netherlands, climate change is expected to negatively affect the macrophyte-dominated clear water state and enhance the phytoplankton-dominated turbid water state (Mooij et al., 2005). PCLake was used to study the effects of a range of temperature and external loading combinations on the alternative stable states of a hypothetic lake (Mooij et al., 2007, 2009). The results of this modelling approach showed a lowered nutrient threshold for the transition from the clear-water state to the turbid-water state with rising temperature in addition to higher summer chlorophyll-a concentrations, more cyanobacteria and less zooplankton (Mooij et al., 2007).

Apart from the general approach PCLake has also been applied to specific lakes to study the effects of climate change. A model of a small peat lake in the Netherlands was calibrated in order to assess the effects of precipitation, evaporation and temperature scenarios on the ecological status of the lake (ter Heerdt et al., 2007). Nielsen et al. (2014) applied PCLake to a Danish lake to test the effect of increases in temperature and nutrient loading.

Further developments for PCLake focus on expanding the model from 0D to include more dimensions so it can be applied to more heterogeneous lakes. Janssen et al. (2014) applied PCLake to the large Lake Taihu to study the occurrence of macrophytes in relation to depth and fetch in different parts of the lake. PCLake was modified and expanded with a 3D hydrodynamics module to apply the model to subtropical lakes (Fragroso Jr. et al., 2009). A new version of PCLake is also under development within the Framework for Aquatic Biogeochemical Models (FABM) which allows the coupling of the lake processes in PCLake with 3D hydrodynamic models (Hu et al., 2016).

1.6. Aim of the research

Shallow lakes are complex ecosystems that react in a non-linear way to many external influences like changes in climate or nutrient loading (Mooij et al., 2005), and predicting the impact on shallow lakes is complex. A combination of different approaches is required to study the relationship between climate, hydrology and nutrient dynamics in shallow Mediterranean lakes.

In this PhD-research a combination of three approaches were used: the use of long-term data series in order to study trends in the past as indicators of future effects; the use of mesocosm experiments in a space-for-time study on a climate gradient across Europe to analyse nutrient budgets in a controlled environment; a modelling approach to model the future effects of climate change on Mediterranean lakes in order to assess which conservation and mitigations measures can be taken to preserve the lake ecosystems.

Considering the possible impact of climate change on the nutrient dynamics of Mediterranean lakes, these three approaches were employed to investigate three compatible hypotheses:

1. Nutrient retention varies on a climate gradient from a northern cold and wet climate to a Mediterranean warm and dry climate.
2. During dry periods, the seasonal changes in hydrology plays a critical role in the increase of the nutrient conditions in lakes, located in a semi-arid climate.
3. As a result of the warmer and dryer conditions in the Mediterranean climate zone, which are expected as a result of climate change, the risk of lake eutrophication will increase, accompanied by higher lake nutrient concentrations.

CHAPTER II

THE INFLUENCE OF NUTRIENT LOADING, CLIMATE AND WATER DEPTH ON NITROGEN AND PHOSPHORUS LOSS IN SHALLOW LAKES: A PAN-EUROPEAN MESOCOSM EXPERIMENT[1]

2.1. Introduction

P and N are considered to play key roles as limiting factors for primary production in lakes (Likens, 1972; Schindler, 1974; Guildford & Hecky, 2000; Lewis & Wurtsbaugh, 2008). The role of N is subject to debate (Schindler et al., 2008; Scott and McCarthy, 2010, 2011; Paterson et al., 2011; Moss et al., 2013), but there is no doubt that it has importance. Human-induced processes have markedly altered the availability of P and N (Vitousek et al., 1997) and have increased the nutrient loading to many lakes, resulting in eutrophication (Hasler, 1947; Correl, 1998; Smith et al., 1999) and focussing attention on nutrient relationships.

Some of the nutrients entering a lake may be retained in the sediments or organisms (N and P) or lost to the atmosphere (N) (Vollenweider, 1976), thereby reducing export from the lake to downstream ecosystems (Wetzel, 2001). Unfortunately, the word 'retention' has been used to include loss to the

[1] This chapter was published as Coppens, J., Hejzlar, J., Šorf, M., Jeppesen, E., Erdoğan, Ş., Scharfenbergerg, U., Mahdy, A., Nõges, P., Tuvikene, A., Baho, D.L., Trigal, C., Papastergiadou, E., Stefanidis, K., Olsen, S., & Beklioğlu, M., Hydrobiologia, in press. The influence of nutrient loading, water level and climate on nitrogen and phosphorus retention in lakes: a pan-European mesocosm experiment. DOI: 10.1007/s10750-015-2505-9. Reprinted with permission of Springer with licence number 3866901046122.

atmosphere and this is potentially confusing so in this study we use “nutrient loss” or “nutrient change” to refer to the removal of any form of N and P from the water column through a combination of sedimentation, assimilation by organisms (e.g. algae, plants, periphyton) or loss to the atmosphere (denitrification) (Correl, 1998; Wetzel, 2001). Such removal may be temporary since N and P, accumulated in the sediment, for example, can be resuspended by sediment disturbance or released back to the water column by diffusion or decomposition of organic matter in the sediment (Søndergaard et al., 2003). The permanent loss of N_2-gas to the atmosphere through denitrification, however, contributes substantially to N loss in lakes (Seitzinger et al., 2006).

External nutrient loading, water temperature, hydraulic residence time, lake depth, phytoplankton and macrophyte abundance, organic matter content in the sediment and oxygen concentration, are all factors that may determine the nutrient concentrations maintained in the water (Saunders & Kalff, 2001a) and also affect nutrient loss (Windolf et al., 1996; Søndergaard et al., 2003; Seitzinger et al., 2006). There is a strong, positive correlation between external loading and nutrient loss but the relative importance of other factors is less well established. Hydraulic residence time and water depth, for example, affect the duration and extent of the contact between the water column and the sediment so that sediment uptake increases with residence time (Brett & Benjamin, 2008) but decreases with water depth (Windolf et al., 1996). Low water depth may also increase plant growth through increased light availability (Bucak et al., 2012). Submerged macrophytes take up nutrients directly from the water column and the sediment and can influence the exchange of nutrients through oxygen release or uptake, reduction of water movement (Madsen et al., 2001) and stabilisation of the sediment (Stephen et al., 1997). Macrophytes also support denitrification by increasing the supply of organic carbon to denitrifying bacteria (Weisner et al., 1994). N removal from lakes is often also higher at high P concentrations

(Finlay et al., 2013, Olsen et al, 2015) because of increased algal production, leading to higher N uptake (Schindler, 2012), increased sedimentation of this algal biomass (Small et al., 2014) and increased denitrification (Saunders and Kalff, 2001a; Seitzinger et al., 2006).

Climate may also have a profound effect on N and P loss in lakes (Jeppesen et al., 2011). Precipitation and evaporation drive the water balance and can influence water depth, hydraulic residence time and external loading, while temperature affects biochemical reactions including denitrification and other microbial processes. N loss is assumed to increase with increasing temperature because water temperature positively influences the activity of the denitrifying bacteria (Herrman et al., 2008; Veraart et al., 2011). Mineralisation of organic matter and release of inorganic P have also been shown to increase with increasing temperatures, potentially leading to lower net nutrient loss (Gomez et al., 1998). Higher temperatures may also enhance primary production and thereby increase the uptake of N and P by macrophytes and algae; though, higher growth rates at higher temperature may alter the nutrient content of organisms and lower their nutrient demand (Woods et al., 2003). The nutrient contents of phytoplankton and periphyton are also significantly dependent on nutrient concentrations in the water and on light availability (Sterner et al., 1997; Danger et al., 2008).

Many studies have quantified N and P loss in lakes and wetlands, showing maximum capacities above 100 g m^{-2} a^{-1} for N (Persson and Wittgren, 2003; Strand and Weisner, 2013) and 5 g m^{-2} a^{-1} for P (Fisher and Acreman, 2004). Most of the study sites have been in the temperate zone (e.g. Jensen et al., 1990; Windolf et al., 1996; Kaste and Dillon, 2003; Nõges, 2005; Han et al., 2011), and few from the Mediterranean and semi-arid regions (Romero et al., 2002; Cook et al., 2010; Özen et al., 2010). Climate change is expected to reinforce the consequences of eutrophication (Jeppesen et al., 2010; Moss et al., 2011), and in Mediterranean climates high temperatures and high evaporation have been shown to affect lake eutrophication and nutrient loss (Özen et al., 2010). Given the projected effects of climate change, more studies of the relationship between temperature and nutrient processing in warmer climate zones are needed. In particular knowledge of the relative importance of climate driven effects on N and P loss is still very general (Olsen et al., 2015) because most studies have been descriptive and calculated nutrient budgets based on lake monitoring data and related nutrient loss to environmental factors. While whole-lake studies can present comprehensive data on specific lakes, comparisons among lakes along a climate gradient are difficult to make owing to confounding local effects, lack of controlled nutrient loading and lack of replicability.

In this study, we used a controlled experimental environment using mesocosms with a space-for-time approach, using latitude as a surrogate for time, to compare the impact of nutrient loading, nutrient concentrations and water depth on N and P loss under different climatic conditions. We performed concurrent, standardised mesocosm experiments in six countries along a latitudinal climate gradient from Sweden to Greece. This approach allowed us to study the relative importance of climate compared with nutrient loading and concentration, in different water depths and complements the study by Olsen et al. (2015), which focused on N loss in a collection of independent mesocosm experiments.

Our specific hypotheses were firstly that increased temperature and lower water volume would reduce loss of P significantly (because of greater release from the sediments) and therefore that with global warming we might expect increase in the symptoms of eutrophication within a P-limited lake, and greater export downstream of P. Our second hypothesis was that greater temperatures and reduced water level would promote denitrification and therefore increased loss of

N. The concomitant of these hypotheses, if supported, would be a trend towards N as the key limiting nutrient in shallow lakes.

2.2. Methods

2.2.1. Experimental design

The experiments were conducted between May and November 2011 in six European countries covering a wide climate gradient and following a standardised construction and sampling protocol (Landkildehus et al., 2014). Mesocosms were established in Lake Erken in Sweden, Lake Võrtsjärv in Estonia, Lake Müggelsee in Germany, Experimental Pond FROV JU Vodňany in the Czech Republic, ODTÜ DSI 50-Yıl dam lake in Turkey and Lake Lysimachia in Greece (Table 2.1).

The set-up included a pontoon bridge with sidewalks, holding 16 closed fibreglass mesocosms each with a diameter of 1.2 m suspended in the lakes. The experiment was run at two contrasting depths, with a 2×4 factorial design with two nutrient levels and four replicates at each depth. Since the mesocosms were made of fiberglass, no water was exchanged with the surrounding lake or the lake sediment.

The mesocosms were initially filled with 10 cm sediment composed of 90% washed sand and 10% natural sediment from a nearby oligotrophic lake in each country. Largely artificial sediment was chosen to obtain similar conditions in all countries in order to minimise the effect of local sediment characteristics. The sediment was sieved through a 10 mm mesh and large particles were removed (e.g. plant fragments, mussels, stones, debris). In the autumn and winter prior to the experiment, the sediment was stored in two tanks below a water layer 20–50 cm. In one tank the concentration of the overlying water layer was 25 µg P L^{-1} to prepare the sediment for the low nutrient mesocosms, in the other tank the overlying water had a concentration of 200 µg P L^{-1}. The water was replaced once a month. To optimise equilibration the sediment was stirred and resuspended into the water column by a rake. The equilibration process was continued until the P concentrations of the overlying water layer remained stable at 25 µg L^{-1} and 200 µg L^{-1}. No equilibration was performed for N because total nitrogen (TN), in contrast to TP, responds quickly to changes in external loading, suggesting low importance of internal loading and fast equilibration of TN levels (Jeppesen et al., 2005).

All mesocosms were initially filled with nutrient-poor lake or tap water with a total phosphorus (TP) concentration of < 25 µg L^{-1} by which the low TP level was achieved. Soluble reactive phosphate (as Na_2HPO_4) and nitrate (as $Ca(NO_3)_2$) were added to eight of the mesocosms to establish the high nutrient mesocosms with 200 µg L^{-1} TP and 2.0 mg L^{-1} TN levels. The dosing levels and ratios followed those used in previous experiments (Gonzáles Sagrario et al., 2005; Jeppesen et al., 2007a).

The mesocosms for the shallow experiment were established with an initial water level of 1 m and the deeper mesocosms with a 2 m water depth. Water level fluctuations were considered an integral part of the influence of the local climate at the experimental sites and, after establishment the water levels in the mesocosm were left to fluctuate following local precipitation and evaporation. In the Czech Republic, however, excess water due to high precipitation had to be removed on two occasions to prevent the mesocosms from overflowing. Fifty litre of water were removed at both times by filling a bucket with a mixed water sample. These water removals were taken into account as output in the nutrient budget calculations.

Table 2.1: Location, altitude, location of the meteorological stations and rainfall chemistry monitoring stations (with distance to the experimental sites), average daily mean temperature and total precipitation (May–November) during the mesocosm experiments carried out across a latitudinal gradient.

Country	**Lake**	**Altitude (m.a.s.l.)**	**Meteo-rological station**	**Average daily air T (ºC)**	**Total precipi-tation (mm)**	**Rainfall chemistry station**
Sweden	Erken	11	Lake Erken (on site)	15.1	271	Aspvreten (75 km)
Estonia	Võrtsjärv	35	Lake Võrtsjärv (on site)	15.6	252	Alam-Pedja, Loodi and Otepää (30 km)
Germany	Müggelsee	32.4	Müggelsee (on site)	16.4	424	Müggelsee (on site)
Czech Republic	Vodňany	395	České Budějovic e (30 km)	15.5	401	Vodňany (on site)
Turkey	METU DSI 50-Yıl dam lake	998	Ankara (15 km)	19.8	168	Çubuk and Çamkoru (100 km)
Greece	Lysimachia	16	Agrinio (15 km)	23.4	252	-

To establish the mesocosm food web, mixed phytoplankton and zooplankton inoculations from five local lakes in each country, then eight *Myriophyllum spicatum* L. plants and finally six fish with a 1:1 male:female ratio were introduced. In all countries three-spined sticklebacks (*Gasterosteus aculeatus* L.) were stocked, except in Sweden, where roach (*Rutilus rutilus* L.) and Greece, where mosquito fish (*Gambusia affinis* Baird and Girard) had to be used. Dead fish were replaced during the experiment to maintain constant fish populations. The water in the mesocosms was circulated by a submersed pump (300 l h^{-1}) for the entire duration of the experiment.

Monthly additions of N and P were made to counteract nutrient loss and to reset the nutrient concentrations to approximately the initial levels. Different doses were used for the different treatments to account for the differences in volume and nutrient level: 5.1 mg P and 102 mg N were added monthly to the four shallow/low nutrient mesocosms (SL); 40.8 mg P and 816 mg N to the four shallow/high nutrient mesocosms (SH); 10.8 mg P and 216 mg N to the four deep/low nutrient mesocosms (DL) and 86.0 mg P and 1720 mg N to the four deep/high nutrient mesocosms (DH). In order to ensure a similar nutrient loading in all countries, these quantities were kept constant during the experiment, even when the water volume changed in some of the countries.

The first sampling for chemical analysis was conducted three days after the establishment of the different nutrient treatments. Thereafter monthly water samples were taken and analysed for soluble reactive phosphorus (SRP), TP, ammonium (NH_4-N), nitrite and nitrate (NO_2+NO_3-N) and TN. The water samples were analysed in the laboratories of the respective countries where the experiment was performed. Concentrations below the detection limits of the equipment were equated with those limits. Dissolved oxygen, chlorophyll-a and percent volume inhabited by plants (PVI) were measured at monthly intervals in all countries. Macrophyte dry weight biomass was determined at the end of the experiment.

2.2.2. Nutrient loss calculations

Nutrient loss from the water column was calculated for TN, dissolved inorganic nitrogen (DIN), TP and SRP. While DIN and SRP represent the inorganic nutrient pools, TN and TP encompass both inorganic and organic nutrient components. An operationally defined "organic" portion of N and P loss was calculated as the difference between total and inorganic nutrient loss. This portion includes N and P mainly in planktonic organisms and detritus but also in dissolved forms other than DIN and SRP. Nutrient loss between two sampling dates was calculated for N and P using the mass balance model of Messer & Brezonik (1978):

$$\text{Nutrient loss} = \text{Input} - \text{Output} - \Delta \text{ nutrient content} \quad (2)$$

where "Input" is the input of nutrients to the mesocosms in the relevant time interval through nutrient additions and atmospheric deposition (N fixation is neglected); "Output" is loss of nutrients through water removal, and "Δ nutrient content" (= nutrient content at time t + 1 minus nutrient content at time t) is the change in nutrient content in the water column between the two sampling dates.

Nutrient loss (TN_{loss}, DIN_{loss}, TP_{loss}, SRP_{loss}) was calculated in milligrams (mg) and converted to mg m^{-2} d^{-1} by dividing the nutrient loss value by the surface area of the mesocosms and the number of days between the two sampling dates. The difference of TN_{loss} - DIN_{loss} = $OrgN_{loss}$ and TP_{loss} - SRP_{loss} = $OrgP_{loss}$ was also calculated for all mesocosms. All calculations were first performed for each time interval between the monthly sampling dates, after which a weighted average was calculated for the entire experimental period. Weighted averages were calculated by multiplying the values measured on the sampling date by its week number and dividing the sum of these by the sum of the week numbers. Weighted averages were used because they reduce the influence of the early transitional phase after the start of the experiment (Stephen et al., 2004).

The weighted average for shallow mesocosms was calculated based on the data until mid-September because the shallow mesocosms in Greece had significantly decreased water levels owing to evaporation, followed by complete drying out (after 11 September). The average temperature in Greece was markedly higher than in the northern countries, making the Greek data highly relevant for analysing the relationship between nutrient loss and climate. Therefore, we chose to exclude the data for October and November from all countries rather than exclude all the shallow mesocosms in Greece. For the deep mesocosms, the weighted average was calculated for the entire duration of the experiment (May–November).

Additionally, nutrient loss was calculated relative to the pool of available TN, DIN, TP and SRP and expressed as $TN_{\%\text{-loss}}$, $DIN_{\%\text{-loss}}$, $TP_{\%\text{-loss}}$ and $SRP_{\%\text{-loss}}$ in order to examine the actual nutrient loss compared with maximum potential nutrient loss:

$$\text{Relative nutrient loss} = \text{total nutrient loss} / \text{total available nutrient pool} \quad (3)$$

The total available nutrient pool for the entire experiment duration was calculated by adding the monthly nutrient additions, monthly nutrient loading through precipitation and the nutrient content present in the water column at the start of the experiment.

Meteorological data were taken from the meteorological stations listed in Table 2.1. N and P concentrations of precipitation were measured near the experimental site in Germany (TN, TP) by the Department of Chemical Analytics of the Leibnitz Institute of Freshwater Ecology and Inland Fisheries (IGB) and in the Czech Republic (DN, TP) at Pond Vodňany. For the other countries data from nearby monitoring stations were used. Data from Nõges et al. (1998) were used for TP concentrations of precipitation in Estonia, as well as for Sweden due to lack of data. TP concentrations for Turkey were taken from Koçak et al. (2010). Monitoring data on nutrients in precipitation were not available from Greece, so DIN and TP concentrations from Turkey were used as substitute. SRP loading through precipitation was ignored for the calculation of SRP loss owing to lack of data.

2.2.3. Statistical analysis

SAS 9.2 Statistical Software (SAS Institute Inc., Cary, NC) was used for statistical analysis. One-way ANOVA and non-parametric Kruskal-Wallis tests were used on mean values of TN and TP to test if significantly different effects of nutrient treatments emerged.

ANCOVA analyses using mixed linear models (Proc Mixed in SAS) were performed separately for the shallow and deep mesocosms on weighted average nutrient loss and relative nutrient loss, with average air temperature at the experimental sites as a covariate and nutrient treatment as a fixed factor. The shallow and deep mesocosms were treated as two distinct experiments because of the differences in nutrient loading, which means that no direct test for the water depth effect could be made. ANCOVA analysis was performed using weighted averages because it produces consistent results with repeated measures analysis when the data do not fit normality and sphericity assumptions (Stephen et al., 2004). Transformations (natural logarithmic and square root) were used to meet the normality and homoscedasticity assumptions of ANCOVA.

Air temperature was used to represent the climatic gradient in the ANCOVA analysis. Air temperature was measured daily at each site and water temperature on the sampling date was significantly correlated with average air temperature between that sampling date and the previous sampling ($p<0.0001$, $r^2 = 0.85$).

Although measured temperatures differed between countries, no significant differences in water temperature among mesocosms were found within the individual countries (tested with ANOVA). Air temperature at the experimental sites could therefore be used for all mesocosms in the statistical analysis.

In Germany, one DL mesocosm was lost during the experiment and thus excluded from analysis. In Germany and the Czech Republic, there were issues with temporarily submerged mesocosms (two DH and one SH), unaccounted water loss (one DL) and overflowing of mesocosms (two SH, one SL), but as the nutrient loss values of these latter mesocosms were not affected, they were included in the statistical analysis.

Stepwise multiple-linear regression (Proc Reg) was performed for the shallow and deep experiments using weighted average nutrient loss and relative nutrient loss as dependent variables. For every dependent variable, one linear regression analysis was performed with a set of multiple independent variables, in order to examine the relationship between nutrient loss and a set of relevant physical, chemical and biological variables measured during the experiment. Mean temperature, water depth, mean nutrient concentrations (TN, DIN, TP or SRP), TN:TP, nutrient loading quantities (nutrient additions and atmospheric deposition), macrophyte dry weight, chlorophyll-a (chl-a) and dissolved oxygen (DO) were initially selected as independent variables. For each dependent variable the coefficient of determination (R^2) for the overall fit of the regression model, and the semi-partial correlation coefficients (pr^2) for each independent variable were determined to show their individual contribution.

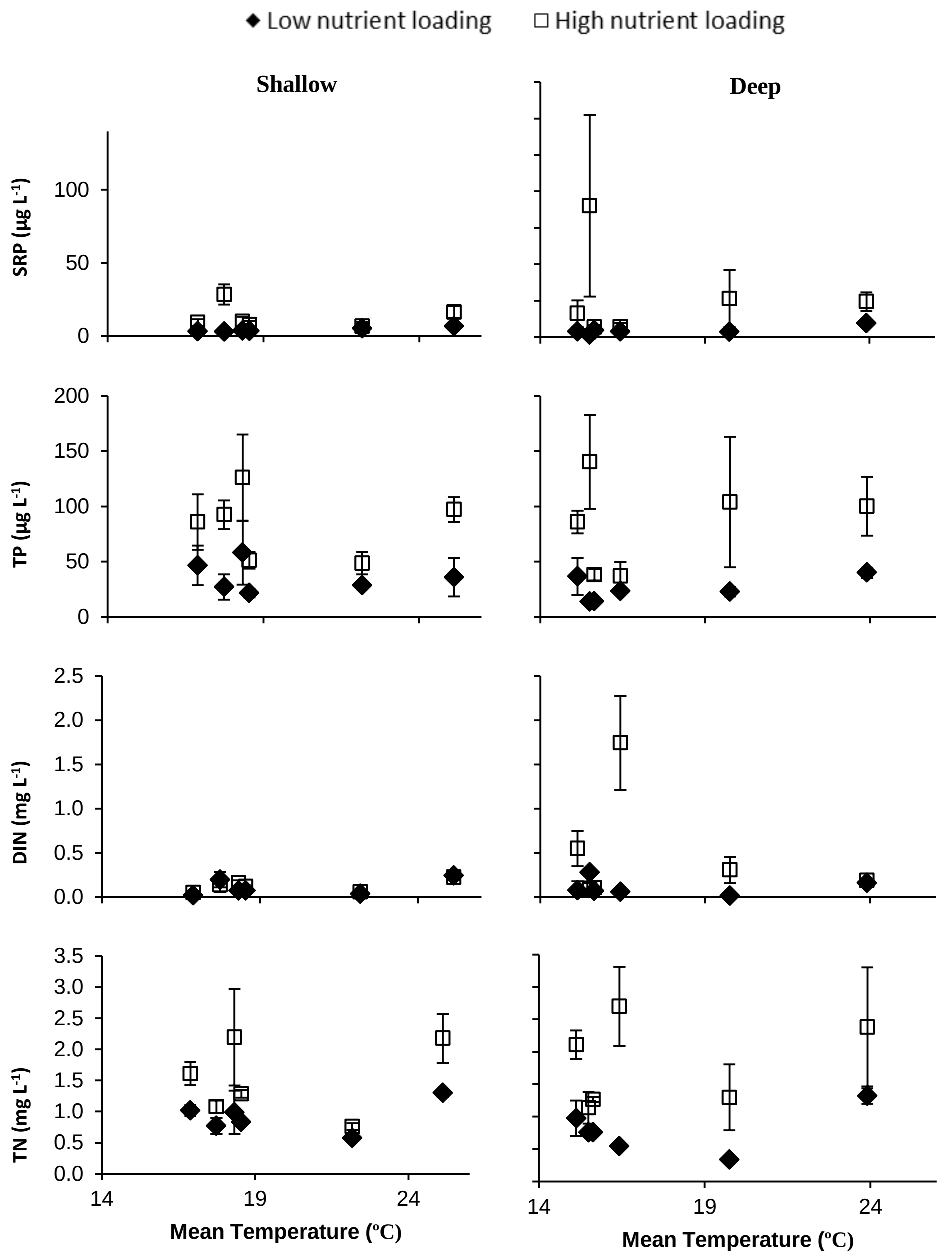


Figure 2.1: Weighted averages with standard error for SRP, TP, DIN and TN concentrations during an experiment with shallow and deep mesocosms set up across a latitudinal gradient and mean air temperature.

2.3. Results

Mean TN, TP, DIN and SRP concentrations in the mesocosms are shown in Figure 2.1 for all countries. ANOVA and Kruskal-Wallis tests showed that low and high nutrient mesocosms had significantly different TN and TP concentrations in all countries ($p<0.01$) except for TN in the Czech mesocosms and the deep Greek mesocosms.

Precipitation constituted a large source of N for the low nutrient mesocosms. In five of the countries the atmospheric deposition of N ranged from 0.84 mg DIN m^{-2} d^{-1} in Sweden to 2.69 mg DIN m^{-2} d^{-1} in Estonia. In Germany, however, TN deposition was as high as 10.2 mg m^{-2} d^{-1}. Addition of TP through precipitation ranged between 0.02 mg m^{-2} d^{-1} for Turkey and 0.72 mg m^{-2} d^{-1} for Germany and constituted a large source of P in the low nutrient mesocosms.

2.3.1. Total nitrogen loss

TN_{loss} was highest in the deep and high nutrient mesocosms (DH>SH>DL>SL) in all countries except Greece (Figure 2.2). The $TN_{\%\text{-}loss}$ was highest in the SH (85%) and DH (79%) treatments (Figure 2.2). In the low nutrient mesocosms, the $TN_{\%\text{-}loss}$ was 66% for DL and 59% for SL (SH>DH>DL>SL).

Nutrient treatment had a significant effect on the TN_{loss} and $TN_{\%\text{-}loss}$ in both the shallow and the deep mesocosms (Table 2.2). In the deep mesocosms, mean temperature at the experimental sites had a significant negative effect on TN_{loss} (Figure 2.3), while there was a significant positive effect of temperature on $TN_{\%\text{-}loss}$ in the shallow mesocosms (Figure 2.3).

Table 2.2: Effects of nutrient loading and temperature on losses of nitrogen from the water columns of a mesocosm experiment carried out across a latitudinal gradient. F-values, significance levels (ns: not significant; *: 0.05>p>0.01; ** 0.01>p>0.001; ***p<0.001) and nature of the relationship (+ or -) of ANCOVA are shown.

Depth	Effect	TN_{loss}	$TN_{\%\text{-}loss}$	DIN_{loss}	$DIN_{\%\text{-}loss}$
Shallow	Nutrient	180*** (+)	33***(+)	903***(+)	56***(+)
	Temperature	ns	20***(+)	ns	ns
	Temperature × nutrient	ns	ns	ns	ns
Deep	Nutrient	298***(+)	9**(+)	480***(+)	ns
	Temperature	5*(-)	ns	ns	ns
	Temperature × nutrient	ns	ns	ns	ns

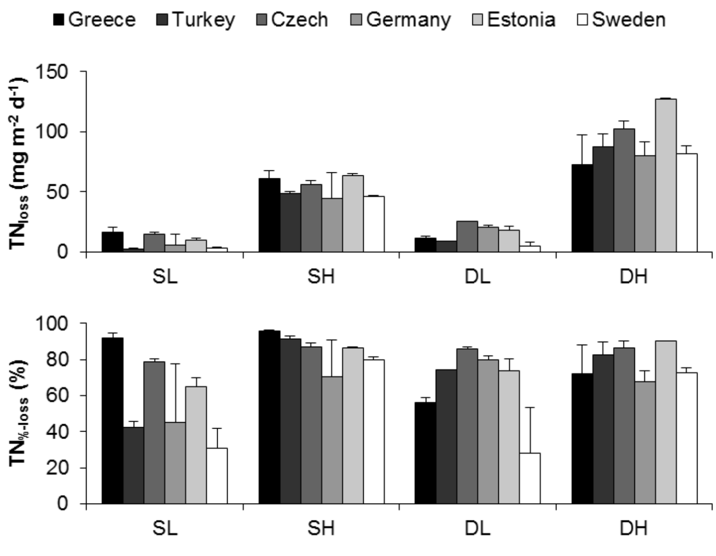


Figure 2.2: Mean absolute and mean relative losses of TN from the water column.

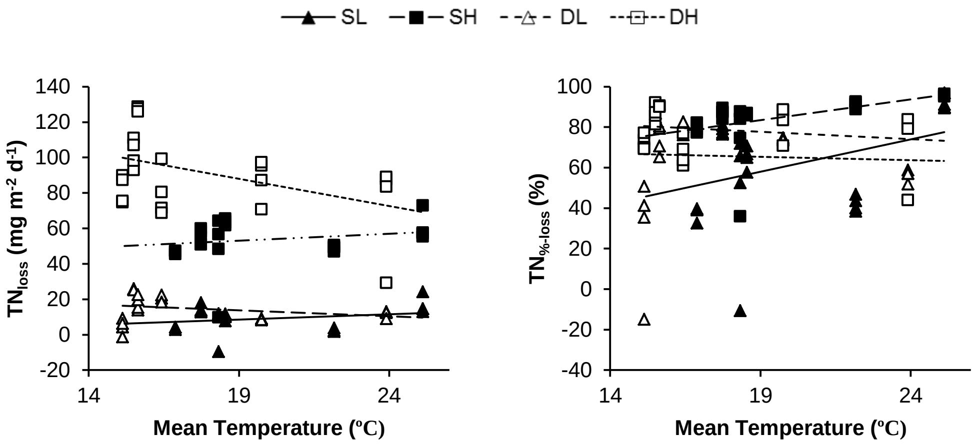


Figure 2.3: Relationships between mean absolute and relative loss of TN from the water column and mean air temperature.

2.3.2. Dissolved inorganic nitrogen loss

In all countries, DIN concentrations declined during the experiment and reached values below 0.06 mg L^{-1} in the SL, SH and DL treatments by the end of the experiment. Only in the DH mesocosms, average DIN concentrations were as high as 0.84 mg L^{-1} in November, despite high nutrient loss.

Similar to TN_{loss}, DIN_{loss} was highest in the high nutrient mesocosms and in the deep mesocosms (DH>SH>DL>SL), and the pattern was the same in all countries (Figure 2.4). The overall DIN loss was higher than the TN loss in the high nutrient treatments but lower in the low nutrient mesocosms. In all treatments, the $DIN_{\%-loss}$ (Figure 2.4) was higher than 90%, except in the deep mesocosms in Sweden and Germany, with SH>SL≈DL>DH.

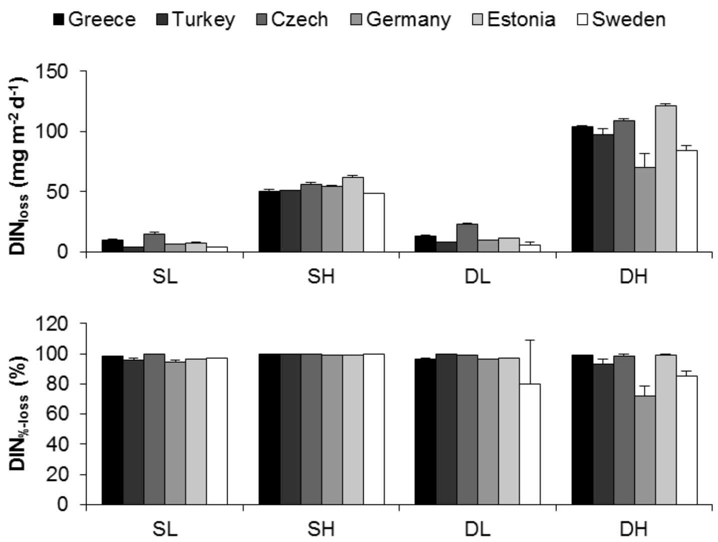


Figure 2.4: Mean absolute and mean relative losses of DIN from the water column.

Nutrient level had a significant positive effect on the DIN_{loss} and $DIN_{\%-loss}$ in the shallow mesocosms and on the DIN_{loss} in the deep mesocosms (Table 2.2). No significant temperature effect was detected at the experimental sites for either DIN_{loss} or $DIN_{\%-loss}$ (Figure 2.5).

The effects of temperature and nutrient level were also determined for organic nitrogen loss ($orgN_{loss}$) taken as the difference between TN_{loss} and DIN_{loss} (not in table). Temperature had a significant and slightly positive effect in the shallow mesocosms ($p<0.001$), while the interaction between nutrient level and temperature was significant for the deep mesocosms ($p<0.01$). Figure 2.6 shows the negative relationship between $orgN_{loss}$ and temperature in the DH mesocosms.

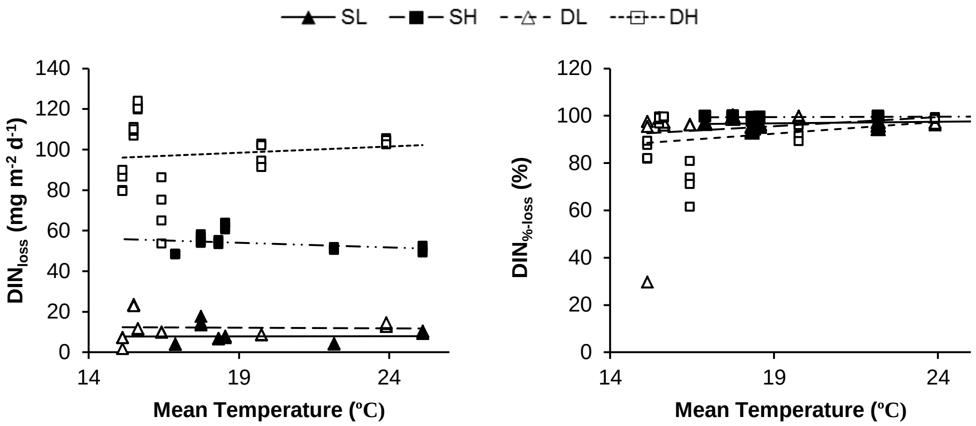


Figure 2.5: Relationships between mean absolute and relative DIN loss from the water column.

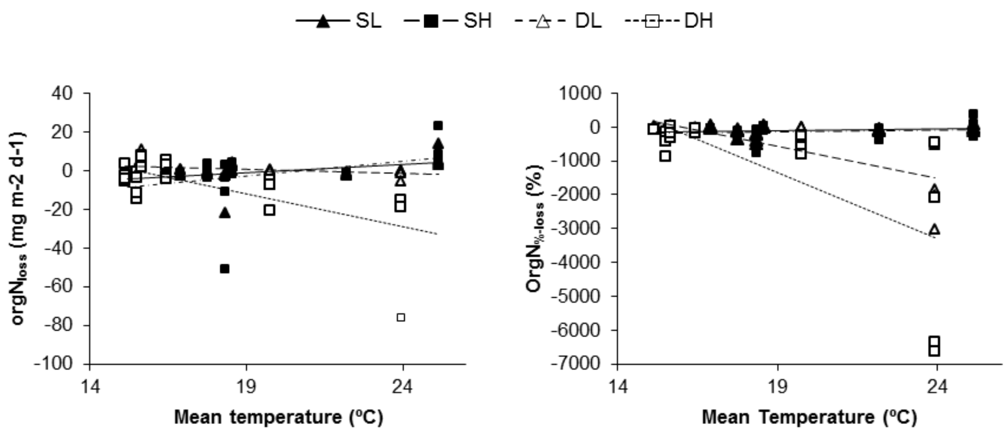


Figure 2.6: Relationships between mean absolute and relative organic N loss from the water column and mean air temperature.

2.3.3. Total phosphorus loss

TP_{loss} was highest in the high nutrient and deep mesocosms (DH>SH>DL>SL) (Figure 2.7). $TP_{\%\text{-}loss}$ was highest in the DH and SH mesocosms (86%), followed by DL (66%) and SL (62%) (SH≈DH>DL≈SL) (Figure 2.7).

TP_{loss} was significantly affected by nutrient level in both the shallow and the deep mesocosms (Table 2.3). The effect of temperature was significant only for the deep mesocosms, with TP_{loss} decreasing with increasing temperature (Figure 2.8). The relationship between temperature and $TP_{\%\text{-}loss}$ depended on depth, with a positive relationship in the shallow mesocosms but a negative relationship in the deep mesocosms (Figure 2.8).

Table 2.3: Effects of nutrient loading and temperature on losses of phosphorus from the water columns of a mesocosm experiment carried out across a latitudinal gradient. F-values, significance levels (ns: not significant; *: 0.05>p>0.01; ** 0.01>p>0.001; ***p<0.001) and nature of the relationship (+ or -) of ANCOVA are shown.

Depth	Effect	TP_{loss}	$TP_{\%\text{-}loss}$	SRP_{loss}	$SRP_{\%\text{-}loss}$
Shallow	Nutrient	272***(+)	32***(+)	336***	92***(+)
	Temperature	ns	12**(+)	ns	6*(+)
	Temperature × nutrient	ns	ns	19***(-)	ns
Deep	Nutrient	437***(+)	20***(+)	1170***(+)	25***(+)
	Temperature	16***(-)	17***(-)	ns	12**(-)
	Temperature × nutrient	ns	ns	ns	ns

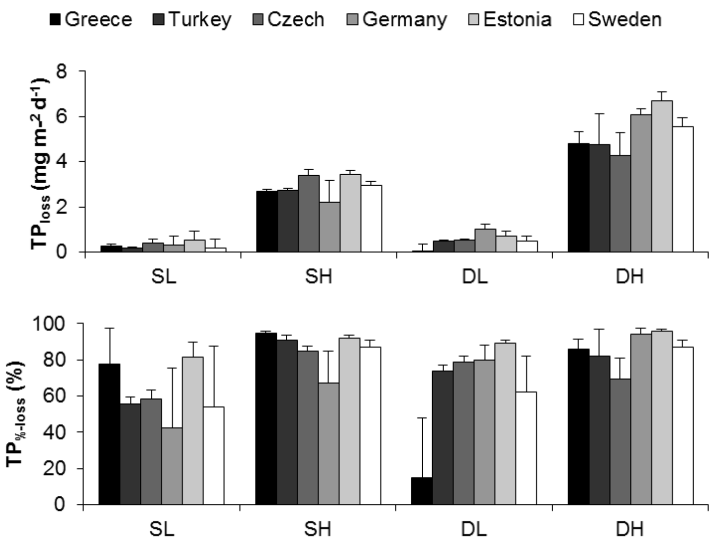


Figure 2.7: Mean absolute and mean relative losses of TP from the water column.

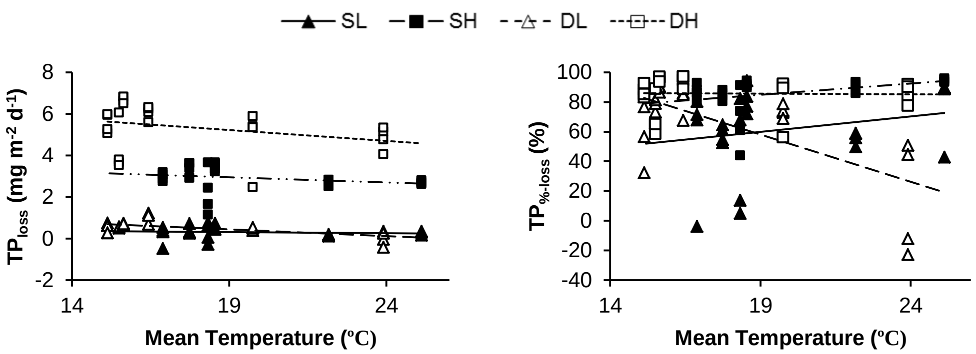


Figure 2.8: Relationships between mean absolute and relative losses of TP from the water column and mean air temperature.

The initial average TN:TP ratio by weight among treatments was 43 (SL), 47 (DL), 15 (SH) and 13 (DH). However, differences among the countries were marked, with ratios ranging from 10 to 111 in the low and from 7 to 25 in the high nutrient treatments. Manual monthly nutrient additions to all treatments were made according to a TN:TP ratio of 20, partially causing the ratios in the different treatments to converge during the course of the experiment. The mean TN:TP ratios were 31 and 36 in the SL and DL treatments, and 17 and 23 in the SH and DH treatments, respectively.

2.3.4. Soluble reactive phosphorus

SRP decreased during the first month of the experiment and the average concentrations at the end of the experiment were below 10 µg L^{-1} in all treatments except DH (34 µg L^{-1}). SRP_{loss} followed the same patterns as TP and TN with DH>SH>DL>SL. Differences among the countries were small (Figure 2.9). Similarly to $DIN_{\%-loss}$, $SRP_{\%-loss}$ was higher than 85% in all treatments, with the high nutrient mesocosms showing higher relative nutrient loss than the low nutrient mesocosms (SH>DH>DL>SL) (Figure 2.9).

Nutrient level had a significant effect on SRP_{loss} in the deep mesocosms and on $SRP_{\%-loss}$ in both the shallow and the deep mesocosms (Table 2.3). For SRP_{loss} in the shallow mesocosms, a significant interaction between temperature and nutrient treatment was found (Figure 2.10). A significant positive effect of mean air temperature was recorded on $SRP_{\%-loss}$ in the shallow mesocosms in contrast to a significant negative effect in the deep mesocosms (Figure 2.10).

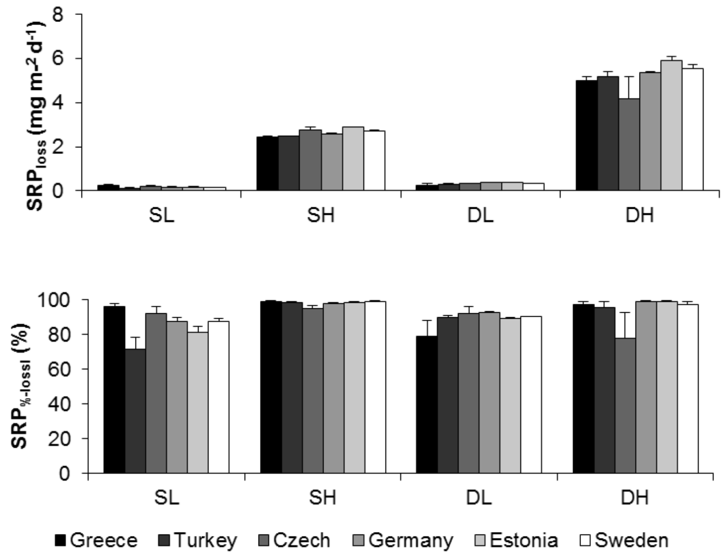


Figure 2.9: Mean absolute and mean relative losses of SRP from the water column.

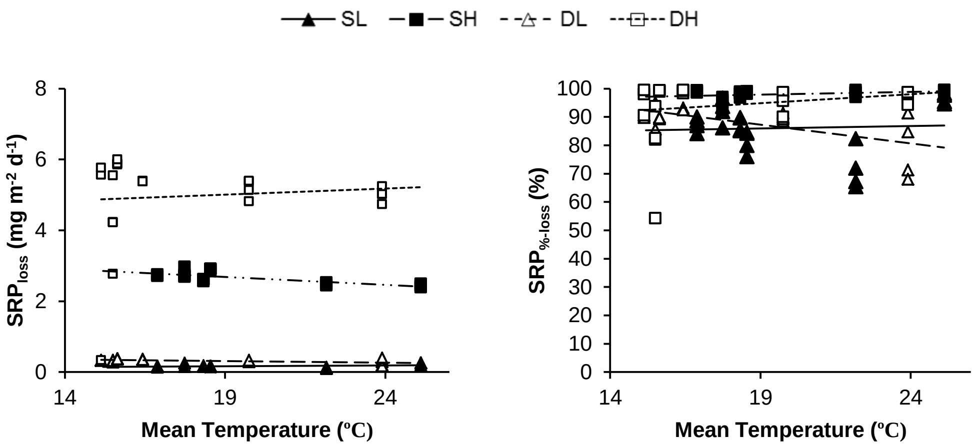


Figure 2.10: Relationships between losses of SRP from the water columns of sets of mesocosms set up along a latitudinal gradient and mean air temperature.

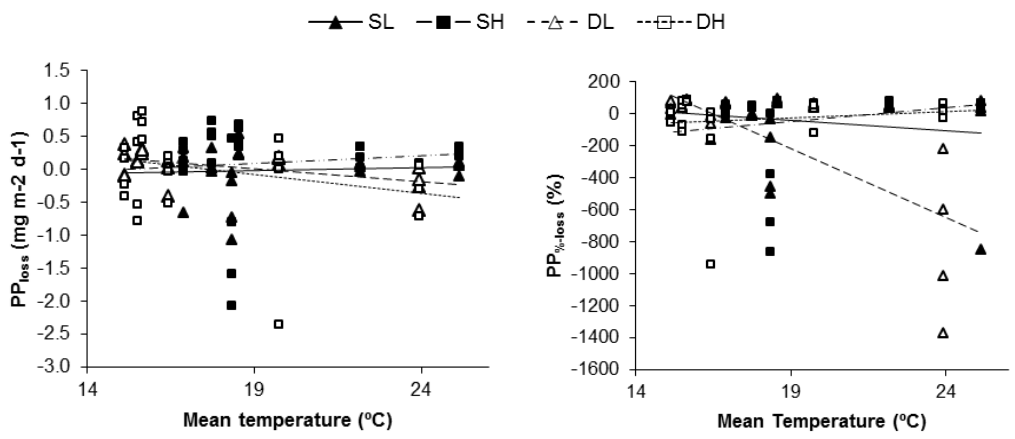


Figure 2.11: Relationships between mean absolute and relative losses of organic phosphorus from the water column and mean air temperature.

Tests for the effects of temperature and nutrient level on OrgP$_{loss}$ showed no significant results for the shallow mesocosms and a significant positive effect ($p<0.001$) of temperature in the deep mesocosms (Figure 2.11). No significant effect of nutrient level was found.

2.3.5. Other environmental variables

Average values for depth, DO, PVI, macrophyte biomass and chl-a are shown in Figure 2.12. Average depth clearly decreased towards the warmer countries. PVI and macrophyte biomass was highest in the warmer countries with PVI highest in shallow tanks, while chl-a was highest in deep tanks. Highest PVI in the shallow mesocosms was generally measured near the end of the experiment in September. In the deep mesocosms, while PVI decreased slightly in November in some mesocosms, macrophytes mostly remained until the end of the experiment. DO in the mesocosms was markedly lower in the warmer countries.

Depth and DO concentrations showed significant negative correlations with temperature ($r^2 < -0.60$; $p<0.001$), PVI had a significant positive correlation with temperature ($r^2 > 0.60$; $p<0.001$) in the shallow mesocosms, while mean TP and mean TN were significantly correlated with P loading and N loading respectively ($r^2 >0.60$, $p<0.001$). Therefore these variables had to be excluded from the regression analysis. Multiple linear regression analysis (Table 2.4) showed a significant positive influence of nutrient loading on $TN_{loss,}$ $DIN_{loss,}$ TP_{loss} and SRP_{loss} while other factors had only minor influence. For relative nutrient loss, nutrient loading was still significant, but contributed less to the overall fit of the regression models. Mean temperature, TN:TP and mean nutrient concentration were the other significant factors for the relative nutrient loss.

Chl-a had a significant, mostly negative, relationship with weighted average and relative N and P loss in the shallow mesocosms, but was only significant for TP loss in the deep mesocosms. Macrophyte biomass at the end of the experiment was negatively related to TN loss in shallow mesocosms and DIN_{loss} in the deep and was not related to relative nutrient loss. TN:TP demonstrated a significant positive relationship with TN and SRP loss in the shallow mesocosms, but a negative relationship in the deep mesocosms.

Table 2.4: Regression relationships relating environmental conditions to loss of nitrogen and phosphorus pools in the water columns of a set of mesocosm experiments carried out across a latitudinal gradient. Regression coefficients (β), semipartial correlation coefficients (pr^2) and coefficients of determination (R^2) for multiple linear regressions are shown for (a) shallow mesocosms and (b) deep mesocosms (-, not included in the model for that dependent variable (DV); ns, not significant;*, 0.05>p>0.01; **, 0.01>p>0.001; ***p<0.001).

(a)

DV		Mean T°	Mean NO_3	Mean SRP	TN:TP	Nutrient loading	Macro-phyte biomass	Chl-a	R^2
TN_{loss}	β	ns	-	-	0.36**	0.04***	-0.11**	-0.20***	
	pr^2				0.02	0.82	0.00	0.06	0.91
TP_{loss}	β	ns	-	-	ns	0.04***	ns	-0.01***	
	pr^2					0.83		0.09	0.92
DIN_{loss}	β	ns	29.34***	-	ns	0.04***	ns	-0.04*	
	pr^2		0.01			0.96		0.00	0.97
$\ln(SRP_{loss})$	β	ns	-	13.72*	0.01**	0.04***	ns	-0.00**	
	pr^2			0.00	0.00	0.95		0.01	0.97
$TN_{\%-loss}$	β	2.42**	-	-	0.87***	0.03***	ns	-0.22**	
	pr^2	0.14			0.12	0.27		0.09	0.62
$\ln(TP_{\%-loss})$	β	-0.11***	-	-	-0.04***	-0.03***	ns	0.01***	
	pr^2	0.16			0.15	0.28		0.14	0.72
$DIN_{\%-loss}$	β	ns	8.67***	-	0.04*	0.00***	ns	-0.02**	
	pr^2		0.14		0.04	0.43		0.07	0.68
$\ln(SRP_{\%-loss})$	β	-0.07*	-	ns	-0.02**	-0.03***	ns	ns	
	pr^2	0.06			0.06	0.61			0.74

Table 2.4 (continued)

(b)

DV		Mean T°	Mean SRP	TN:TP	Nutrient loading	Macrophyte biomass	Chl-a	R^2
TN_{loss}	β	-2.34**	-	-0.24*	0.02***	ns	ns	
	pr^2	0.02		0.02	0.83			0.87
TP_{loss}	β	ns	-	ns	0.04***	ns	-0.01***	
	pr^2				0.92		0.02	0.94
DIN_{loss}	β	ns	-	ns	0.03***	-0.20*	ns	
	pr^2				0.92	0.01		0.93
SRP_{loss}	β	ns	-18.18***	-0.01**	0.03***	ns	ns	
	pr^2		0.02	0.00	0.96			0.99
$\ln(TN_{\%\text{-loss}})$	β	0.09*	-	0.01*	-0.00*	ns	ns	
	pr^2	0.07		0.07	0.14			0.28
$\ln(TP_{\%\text{-loss}})$	β	0.08*	-	-0.02*	-0.00***	ns	ns	
	pr^2	0.11		0.20	0.29			0.60
$\ln(DIN_{\%\text{-loss}})$	β	ns	-	0.01*	ns	ns	ns	
	pr^2			0.09				0.09
$\ln(SRP_{\%\text{-loss}})$	β	ns	24.82***		-0.01***	ns	ns	
	pr^2		0.40		0.39			0.79

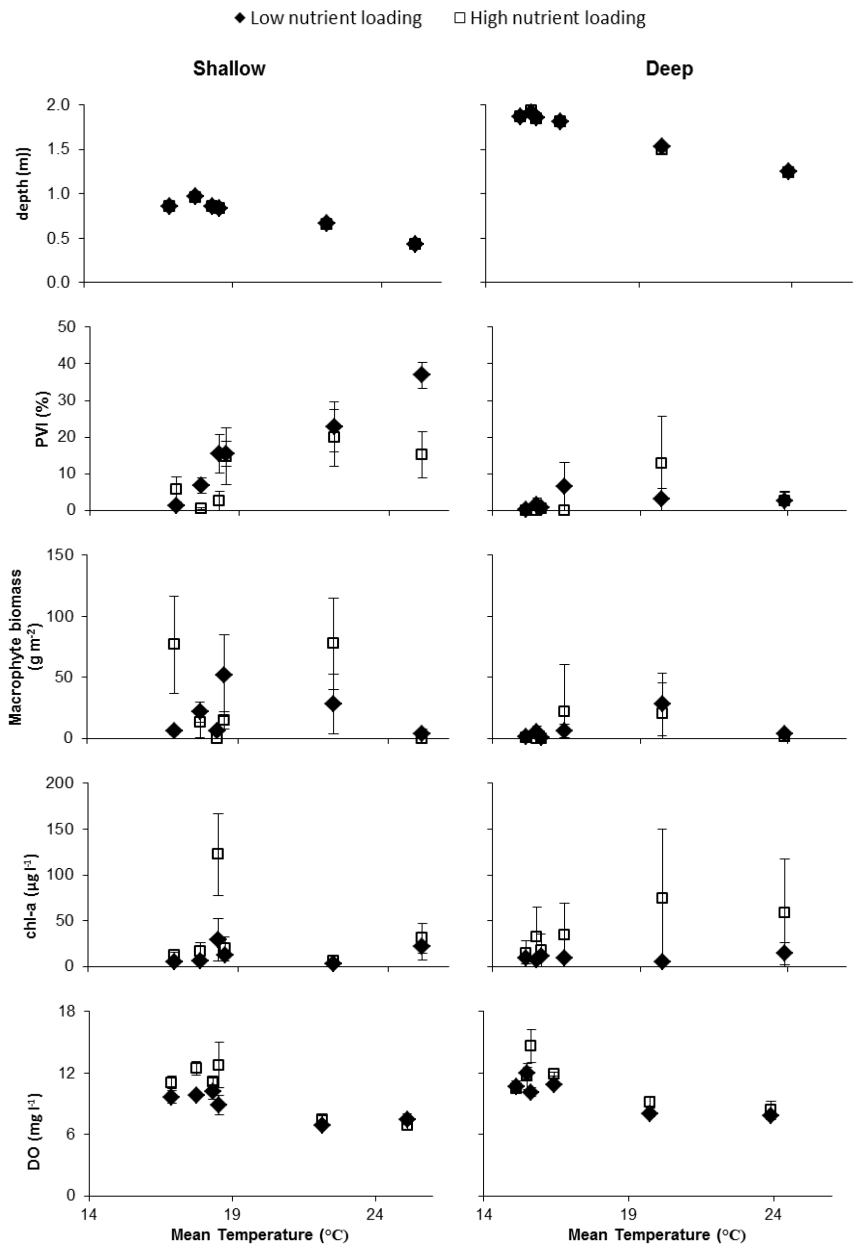


Figure 2.12: Differences in environmental factors in the water columns of mesocosms set up along a latitudinal gradient and mean air temperature. Weighted averages with standard error are shown for water depth, percent volume inhabited by plants (PVI), chlorophyll-a (chl-a), dissolved oxygen (DO) and macrophyte dry weight biomass (measured at the end of the experiment) for shallow and deep mesocosms.

2.4. Discussion

Our results show that external loading had a large influence on nutrient loss along the climate gradient together with a comparatively modest influence of temperature, depth, and algal and macrophyte biomass. In all six countries, TN_{loss}, DIN_{loss}, TP_{loss} and SRP_{loss} were determined by the availability of nutrients through nutrient loading, with the highest nutrient loss (weighted mean) occurring in high nutrient treatments in both the shallow and the deep mesocosms. Higher nutrient loading means more nutrients are available in the system for plant and algal uptake, and thus creates a larger potential for nutrient loss from the water column through uptake. The results of the multiple linear regression clearly demonstrated the importance of external loading for nutrient loss, with nutrient loading contributing the largest component of the coefficients of determination. This is in agreement with the findings of previous studies (Windolf et al., 1996; Saunders and Kalff, 2001; Søndergaard et al., 2003; Brett and Benjamin, 2008).

The relative nutrient loss as a proportion of the available nutrient pool ($TN_{\%\text{-}loss}$, $DIN_{\%\text{-}loss}$, $TP_{\%\text{-}loss}$ and $SRP_{\%\text{-}loss}$) was also significantly higher in the high nutrient mesocosms. The relative loss of DIN and SRP was close to 100% in all treatments and countries, showing that the uptake capacity of the mesocosm communities was not saturated. Within the nutrient loading range used in our experiments, nearly all available inorganic nutrients were lost in our mesocosms, either through biomass uptake or loss to the sediment or atmosphere. The mesocosms acted as a near complete sink for inorganic nutrients in all countries on the climate gradient.

We also found a significant, though weak, decrease in TN and TP loss in the deep mesocosms when moving from the colder northern to the warmer southern countries, but no effect on DIN and SRP loss, implying a lower organic N and P loss (or higher organic N and P accumulation in the water column), in the warmer south. In the shallow mesocosms, however, we found a tendency for TP and TN loss to increase with temperature, though only significant for relative nutrient loss. Our hypothesis that warming would decrease P loss whilst increasing N loss was partly supported, though only for P in the water columns of the deeper mesocosms and for N in the shallower mesocosms. The decreasing TN_{loss} and TP_{loss} with increasing temperature in the deep mesocosms can mainly be attributed to lower nutrient loss in Turkey and Greece. This may reflect the markedly higher average temperature in Turkey (20 ºC) and Greece (24 ºC) during the experiment, compared with 15–16 ºC in Sweden, Estonia, Germany and the Czech Republic.

Previous studies in microcosms and field measurements have shown increased denitrification rates with increasing temperature (Pinay et al., 2007; Herrman et al., 2008, Veraart et al., 2011), in part caused by a decrease in DO concentrations with rising temperature due to lower solubility of oxygen in water and higher respiration rates compared with photosynthesis (Veraart et al., 2011; Scharfenberger et al., unpublished data). We also found a strong negative correlation between DO and temperature, with DO concentrations being lowest in Greece and Turkey, potentially enhancing denitrification (Pearson et al., 2012; Small et al., 2014). However, DIN_{loss} was generally higher in Estonia and the Czech Republic, indicating that DO was not of key importance for the increase in N loss in these fully mixed mesocosms.

Moreover, DIN_{loss} showed no relationship with average temperature in either the shallow or the deep mesocosms. This finding concurs with Kosten et al. (2009) who found no indication of lower nitrate concentrations in the warm lakes along a latitudinal gradient (5º–55º S), suggesting that other factors limited denitrification and offset the potential increasing effects of temperature. A comparative study by Olsen et al. (2015) of various mesocosm experiments with considerably higher average target nitrate concentrations (10 mg L^{-1} in high N treatments) than in our study (2 mg L^{-1} in high N treatments) revealed accumulation of nitrate in the water column in warmer climates, while nitrate was rapidly removed in experiments conducted in the temperate zone (Feuchtmayr et al., 2009; González Sagrario et al., 2005; Jeppesen et al., 2007b). Accumulation of nitrate in the warm mesocosms was ascribed to limitation of denitrification due to low organic matter availability as no correlation with DO was found (Olsen et al., 2015).

In our experiment, both nitrate and organic matter availability may be potential candidates limiting N loss (Davidsson & Leonardson, 1998; Piña-Ochoa & Álvarez-Cobelas, 2006; Taylor & Townsend, 2010). Nitrate declined throughout the experiment in the low nutrient mesocosms to average values of 30–40 µg L^{-1} for all countries. Both the decline in nitrate concentrations in the mesocosms and the high $DIN_{\%\text{-loss}}$ values (>95%) indicate that almost all available DIN was taken up or lost to the sediment from the mesocosms in all countries. Further N loss through denitrification under warm climate conditions may therefore have been limited by nitrate availability, preventing higher DIN loss in the southern countries.

Analysis of the metabolic processes during our mesocosm experiment by U. Scharfenberger et al. (unpublished data) showed an increase in both gross primary production (GPP) and ecosystem respiration (ER) with increasing temperature. Because ER increased more steeply than GPP, a shift from autotrophy to heterotrophy occurred with increasing temperatures. Therefore net primary production can be expected to be on average lowest in the warmer countries. In the warmer mesocosms more carbon fixed by primary production was released again as CO_2, leaving less organic carbon available for denitrification. Organic matter availability was therefore likely another factor that limited DIN_{loss} in the warmer south.

In the deep mesocosms, the decreasing trend in TN_{loss} and TP_{loss} but not in DIN_{loss} or SRP_{loss} with increasing temperature also indicates a key role of processes related to organic N and P. TN and TP loss, besides denitrification, includes settling of sestonic particles, and removal from the water column through uptake by non-sestonic organisms (such as macrophytes, fish, large invertebrates and periphyton). But inorganic N and P loss also encompasses conversion of inorganic nutrients to organic forms assimilated in the phytoplankton biomass or released into the water column. Average inorganic N and P loss was higher than total N and P loss in the deep, high nutrient mesocosms, indicating organic N and P production and accumulation, as also evidenced by the negative relationship between temperature and both $orgN_{loss}$ and $orgP_{loss}$ in the DH mesocosms. This finding is in accordance with the higher biomass of phytoplankton recorded in the deep, high nutrient mesocosms in these two countries and shown in Figure 2.12.

In the shallow mesocosms, phytoplankton biomass was not higher in the Greek and Turkish mesocosms, but instead macrophyte development was high. In these countries water levels in the shallow mesocosms fell from 100 cm to 20–50 cm over the course of the experiment, which likely enhanced macrophyte development through more favourable light conditions (Bucak et al., 2012). Macrophytes can play an important role in nutrient loss from the water column, either through direct nutrient uptake or by creating conditions favourable for denitrification (Desmet et al., 2011; Kankaala et al., 2002). Higher macrophyte development and higher nutrient uptake by plants can therefore explain why TP and TN loss tended to be higher in the warmer countries and why a positive relationship with temperature was found for $TN_{\%\text{-}loss}$ and $TP_{\%\text{-}loss}$. However, we found a negative relationship between macrophyte biomass and TN_{loss} and DIN_{loss} in the multiple linear regression, but the contribution of macrophyte biomass to the fit of the regression model was low and macrophyte biomass was only measured at the end of the experiment and may thus not reflect the macrophyte nutrient uptake throughout the experiment.

The effect of macrophyte nutrient uptake was further quantified by multiplying the macrophyte biomass by estimates of macrophyte N and P content taken from Duarte (1992) who compiled data on C, P and N content of a series of freshwater angiosperms based on literature data. Lower and upper estimates from this reference show that in countries with high macrophyte growth in the shallow mesocosms (Turkey, Greece, Estonia), between 30 and 100% of the available N-pool and between 10 and 80% of the available P-pool were potentially locked up in the macrophyte biomass. Although this analysis only provides a crude estimate, it suggests that macrophyte uptake potentially played an important role for the nutrient loss in the shallow mesocosms.

The observed differences in nutrient loss among countries could potentially be due to inaccurate estimations of the atmospheric deposition in certain countries for which deposition measurement data were missing. However, reducing atmospheric deposition to zero or doubling the estimate used for atmospheric deposition in Sweden and Greece, where deposition was estimated based on data from Estonia and Turkey respectively, would lead only to a 20–30% change in calculated TN and DIN loss in the low nutrient mesocosms, and would be negligible in the high nutrient mesocosms. For TP loss, the differences would be only about 10%, except in the DL mesocosms in Greece where atmospheric deposition is an important determinant for the nutrient input (90% change in retention estimate). However, such differences do not affect the overall results or the climate pattern observed, because the differences in nutrient loss between the treatments and countries are sufficiently large. The main conclusions remain the same even if reduced or increased values of atmospheric deposition are used.

Nutrient loss pathways through sedimentation, uptake by organisms (including fish) or denitrification were not quantified separately in our experiment. The relation between the trends in macrophyte and algal biomass and the trends in N and P loss, however, suggest that algal and plant uptake played an important role in nutrient loss. All available pools of SRP and DIN were then used by macrophyte or phytoplankton and locked up in the biomass.

2.5. Conclusion

Our study confirmed the importance of nutrient loading in determining N and P loss in the water column of shallow lake ecosystems. Further, we found consistently high levels of DIN and SRP loss in all countries along the climate gradient, demonstrating the high nutrient uptake capacity of shallow lake ecosystems. In deep mesocosms we found a decrease in TN and TP loss with increasing temperature but no decrease in DIN or SRP loss, indicating higher algal production of organic N and P in warmer systems. In shallow mesocosms we found an increasing trend of TN and TP loss with increasing temperature, related to high macrophyte growth in the warmer countries. Increasing N loss through denitrification under warmer conditions due to lower availability of DO seemed to have played a minor role. Other factors, such as nitrate (at low N loading) and organic matter availability, therefore likely determined DIN_{loss}.

While uptake by plant and algal biomass seemed to play an important role, the ultimate fate of the nutrients cannot be ascertained. Further studies involving detailed analysis of sediment uptake, algal and plant uptake and denitrification under different climate conditions would be valuable. Our results, based on a space-for-time mesocosm experiment, with space used as a proxy for climate, should be interpreted with care given the small size of the mesocosms. Mixing conditions in mesocosms are different from natural mixing in lakes and sediment characteristics can play an important role in nutrient loss processes. However, since our experiment was repeated under similar conditions in each country and given the control of nutrients and water depth, we believe that our results provide a valuable contribution to further our understanding of nutrient processing in lakes under different climate conditions.

CHAPTER III

IMPACT OF ALTERNATING WET AND DRY PERIODS ON LONG-TERM SEASONAL PHOSPHORUS AND NITROGEN BUDGETS OF TWO SHALLOW MEDITERRANEAN LAKES[2]

3.1. Introduction

Shallow lakes in semi-arid and arid climate regions, including the Mediterranean, are subjected to large water level fluctuations (WLF), both on a seasonal and an annual scale (Beklioğlu et al., 2007). On a seasonal scale, precipitation is limited to winter and spring, while high evaporation in summer causes substantial water loss from freshwater bodies (Naselli-Flores and Barone, 2005; Önol and Unal, 2014). Year-to-year variation in precipitation is typically high in the Mediterranean climate zone, causing alternations between dry periods with low hydraulic loading, high residence times and low water levels and wet periods with high inflows, increased flushing and high water levels (Mariotti and Dell'Aquila, 2012).

WLF have major effects on the ecological functioning of shallow lakes, including changes in nutrient processing and nutrient retention (Jeppesen et al., 2015). Nutrient concentrations in lakes are predominantly determined by the external nutrient loading (Saunders and Kalff, 2001), but in years with reduced

[2] This chapter was published as Coppens, J., Özen, A., Tavşanoğlu, Ü.N., Erdoğan, Ş., Levi, E.E.,Yozgatlıgil, C., Jeppesen, E. & Beklioğlu, M., 2016. Impact of alternating wet and dry periods on the long-term seasonal phosphorus and nitrogen budgets of two shallow Mediterranean lakes. Science of the Total Environment 563–564: 456–467. Reprinted with permission of Elsevier with licence number 3866900696879.

external loading, concentrations can remain high due to internal loading by resuspension or release of nutrients stored in the top layer of the sediment (Søndergaard et al., 2003). Lake nutrient concentrations are also influenced by retention or loss of nutrients present in the water column through a combination of sedimentation processes, assimilation by organisms and loss to the atmosphere (denitrification) (Correl 1998; Wetzel, 2001).

Retention and internal loading are influenced by water depth and hydraulic residence time. The empirical model of Vollenweider (1976) proposed lower lake nutrient concentrations with increasing hydraulic retention time at otherwise similar external loading. By contrast, Özen et al., (2010) found higher nutrient concentrations during periods with long residence times and low water levels and in two Mediterranean lakes, which they attributed to higher internal loading and higher concentrations caused by evaporative water loss.

Climate change will have major implications for the water and nutrient balances of lakes (Jeppesen et al., 2011). The Mediterranean climate zone is predicted to be particularly severely affected by climate change in the near future (Giorgi, 2006). Precipitation is expected to decrease and evaporation to increase over this century (Erol and Randhir, 2012). Önol and Unal (2014) have predicted a 3-4 ºC increase in temperature for the Central Anatolian region in Turkey by 2071-2100, while the annual precipitation is projected to decrease (-10%) despite enhanced precipitation in autumn. Fewer days with rainfall will occur and the frequency of extreme events will increase. All these predicted changes in climate have the potential to significantly affect the water and nutrient balances of shallow lakes. Therefore, it is important to study the relationship between the water balance and the nutrient dynamics of shallow lakes.

Nutrient budgets have been calculated for a series of temperate lakes (e.g. Garnier et al., 1999; Schernewski, 2003; Nõges, 2005; French and Petticrew, 2007) and for some lakes in other climate zones (Romero et al., 2002; Romo et al., 2005; Cook et al., 2010; Özen et al., 2010; James et al., 2011). Özen et al., (2010) studied the nutrient budgets of Lakes Mogan and Eymir between 1993 and 2007. We built on this study and added 6 more years of data to further investigate the effect of high hydraulic loading after prolonged drought conditions on the shallow lakes in the semi-arid climate zone. In addition, we used a seasonal approach and constructed seasonal water and nutrient budgets for both lakes and all years to investigate the impact of periods with low and high water levels on seasonality. We further included both inorganic and total P and N in the nutrient budgets and used time series regression analysis to study the influence of water level and external and internal loading on the nutrient concentrations in the lakes. This allowed us to study in more detail how changes in hydraulic loading, water level and hydraulic residence time influenced the nutrient concentrations under low and high water level conditions. Such knowledge might contribute to the understanding of how climate change affects lake eutrophication and the consequences hereof for lake restoration and management. The aim of our research was (1) to further analyse the impacts of low and high water level periods on the nutrient concentrations of shallow, semi-arid lakes based on an extended set of data from the two lakes studied by Özen et al., (2010) and (2) to explore how the seasonality of lake nutrient concentrations changes between low and high water level periods.

3.2. Material and methods

3.2.1. <u>Study area</u>

The study area comprised two shallow interconnected lakes, Lakes Mogan and Eymir, situated 20 km south of Ankara at an altitude of 970 m on the Anatolian plateau in Central Turkey (Figure 3.1). The local climate is characterised by arid cold steppe conditions (Peel et al., 2007) with hot summers, cold winters and most rainfall (including snow) in winter and spring. The average annual temperature is 12.0 °C and average temperatures have been slowly increasing in recent decades, from 11.2 °C in 1980 to 13.2 °C in 2013. Average yearly total rainfall is 408 mm, alternating between dry periods with precipitation below 350 mm and wet years with precipitation above 500 mm (Turkish Meteorological Office). Average yearly pan evaporation is 1200 mm. Pan evaporation was positively correlated with mean temperatures (0.88***) and due to increases in temperature, yearly pan evaporation has been increasing from 1125 mm in 1985 to 1275 mm in 2012 (Turkish Meteorology Office).

Upstream Lake Mogan (39º 47'N 32º 47'E) has a surface area of 5.6-8 km^2, depending on water level, and an average depth of 2.4 m. The catchment area covers 940 km^2. The lake has several, mainly seasonal, inflows, but only four major inflows (Sukesen, Gölcük, Yavrucak and the Çölovası wetland area) account for 90% of the total hydraulic loading to the lake. The outflow of the lake is regulated and is situated at the north end of the lake. It connects Lake Mogan with Lake Eymir through a canal and a wetland area. Inflow volume to Lake Mogan is influenced by precipitation, but the structure of the catchment means that the relationship is not straightforward. In the Çölovası catchment, precipitation is first collected in two small reservoirs before flowing in the wetland area that feeds Lake Mogan, causing a delayed response to changes in precipitation. In the Sukesen and Yavrucak catchments precipitation recharges the groundwater table and surface flow is relatively limited, especially in summer, leading to a suppressed response. But modelling of the catchment shows that changes in precipitation directly influence inflow volumes to the lakes on an annual basis (see chapter IV). Chlorophyll-a (chl-a) concentrations and percent volume inhabited by submerged plants (PVI) are shown in Table 3.1. PVI during 2001, 2004 and 2008 years with low water levels was higher than 40%, whereas few plants were observed during the high water level period from 2009 to 2013.

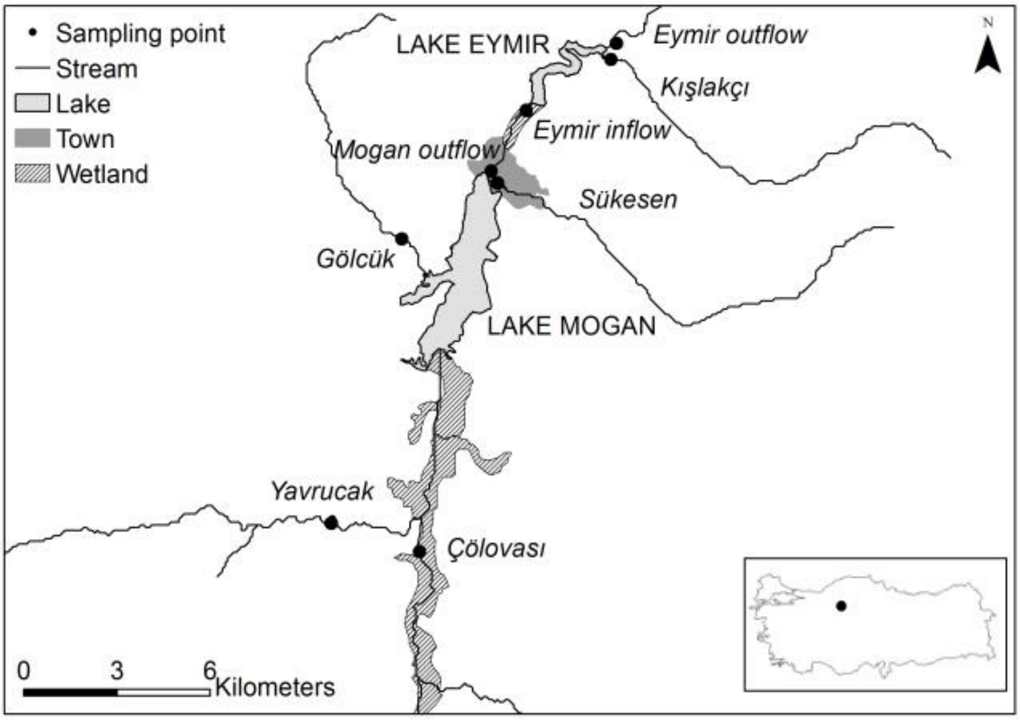


Figure 3.1: The study area comprising the Lakes Mogan and Eymir catchment in the Central Anatolia region of Turkey, showing both lakes, the sampling stations on the major in-and outflows and the location of Gölbaşı town and the upstream and downstream wetland areas.

Downstream Lake Eymir (39º 57’N 32º 53’E) is smaller than Lake Mogan and has a surface area of 1-1.3 km^2 and an average depth of 3.2 m. Apart from the main southern inflow, coming from Lake Mogan, a small second inflow (Kışlakçı) is situated near the north end of the lake, contributing to the flow to the lake during the spring snow melt. The outflow of Lake Eymir is regulated as well and flows to the low-lying areas of the city of Ankara. Including the Lake Mogan catchment, the total drainage area of Lake Eymir is 1010 km^2. The predominant land use is agriculture (68% of the catchment) with non-irrigated wheat farming as the most common crop. The north of the catchment borders the city of Ankara and the suburban town of Gölbaşı (100,000 inhabitants) is located between the two lakes.

Table 3.1: Average soluble reactive phosphorus (SRP), total phosphorus (TP), dissolved inorganic nitrogen (DIN) and total nitrogen (TN) and chlorophyll-a (Chl-a) concentrations together with percent volume inhabited by submerged plants (PVI %) for the entire study period, during low water level (LWL) years and high water level (HWL) years for Lake Mogan.

	Lake Mogan		
	1993-2013	**LWL**	**HWL**
SRP ($\mu g\ L^{-1}$)	30 ± 10	30 ± 10	20 ± 10
TP ($\mu g\ L^{-1}$)	90 ± 30	110 ± 40	80 ± 20
DIN ($mg\ L^{-1}$)	0.32 ± 0.24	0.38 ± 0.34	0.28 ± 0.10
TN ($mg\ L^{-1}$)*	1.26 ± 0.67	1.56 ± 0.39	1.12 ± 0.78
Chl-a ($\mu g\ L^{-1}$)	13.3 ± 4.6	14.2 ± 5.4	12.6 ± 4.2
PVI (%)	20 ± 20	29 ± 20	13± 17

* data for TN only between 2008-2013

Table 3.2: Average soluble reactive phosphorus (SRP), total phosphorus (TP), dissolved inorganic nitrogen (DIN) and total nitrogen (TN) and chlorophyll-a (Chl-a) concentrations together with percent volume inhabited by submerged plants (PVI %) for the entire study period, during low water level (LWL) years and high water level (HWL) years for Lake Eymir.

	Lake Eymir		
	1993-2013	**LWL**	**HWL**
SRP (µg L^{-1})	160 ± 100	180 ± 70	150 ± 120
TP (µg L^{-1})	270 ± 130	290 ± 80	260 ± 160
DIN (mg L^{-1})	0.50 ± 0.27	0.54 ± 0.27	0.48 ± 0.28
TN (mg L^{-1})*	1.72 ± 1.25	2.70 ± 1.24	1.24 ± 0.80
Chl-a (µg L^{-1})	27.5 ± 5.4	29.7 ± 17.4	26.0 ± 10.9
PVI (%)	5 ± 8	5 ± 9	5 ± 9

* data for TN only between 2008-2013

Lake Eymir received sewage effluent from Gölbaşı town for up to 25 years until diversion was established in 1995 (Altınbilek et al., 1995). After sewage effluent diversion, biomanipulation was carried out in the lake in 1998-1999 and again in 2006-2014 to aid the recovery of the lake. In 2000, after removing 50% of the lake's stock of common carp (*Cyprinus carpio*) and tench (*Tinca tinca)*, chl-a and suspended solid concentrations dropped 2-fold and 4-fold, respectively, while Secchi depth increased 2.5-fold. This caused macrophyte coverage to expand from around 2.5% to 40-90% post biomanipulation (Beklioğlu and Tan 2008). Five years later, the lake had shifted back to a turbid water state, with low macrophyte coverage and recovering carp and tench populations at the onset of the drought period from 2003 until 2009 (Beklioğlu and Tan 2008). A second biomanipulation resulted in a 2-fold and 1.5-fold decrease in chl-a and suspended solids, respectively, but macrophytes did not recover, likely because water levels were high, especially after the start of a new wet period in 2009. Lake Eymir had higher chl-a concentrations than Lake Mogan, averaging 28 μg L^{-1}. In Lake Eymir, PVI never exceeded 10% from 2002 onwards, not even after further biomanipulation. Due to the higher water depth in Lake Eymir (max. depth > 6 m). summer stratification was regularly observed, leading to deoxygenation of the bottom water layer in summer (Figure 3.2).

3.2.2. Data collection

The lakes and the five major inflows and both outflows, were sampled from March 1997 to December 2013. Sampling was carried out fortnightly during spring, summer and autumn and monthly during winter. Water temperature and dissolved oxygen (DO) were measured every half metre at the deepest point of the lake. Mixed water samples from the entire water column were collected using a Ruttner water sampler and analysed in the laboratory for soluble reactive phosphorus (SRP) and total phosphorus (TP) using the molybdate reaction methodology (Mackereth et al. 1978) and analysed for NH_4-N, NO_2-N+NO_3-N and total nitrogen (TN) using the Scalar Autoanalyzer Method (San++ Automated Wet Chemistry Analyzer, Skalar Analytical, B.V., Breda, The Netherlands). Monitoring data on the lakes for the period 1993-1995 were taken from the study of Altınbilek et al. (1995). Water samples were also analysed for chl-a for both lakes. Macrophyte PVI has been measured in both lakes once a year in August or September since 2001 using plant surface coverage, height and water depth (Canfield et al. 1984).

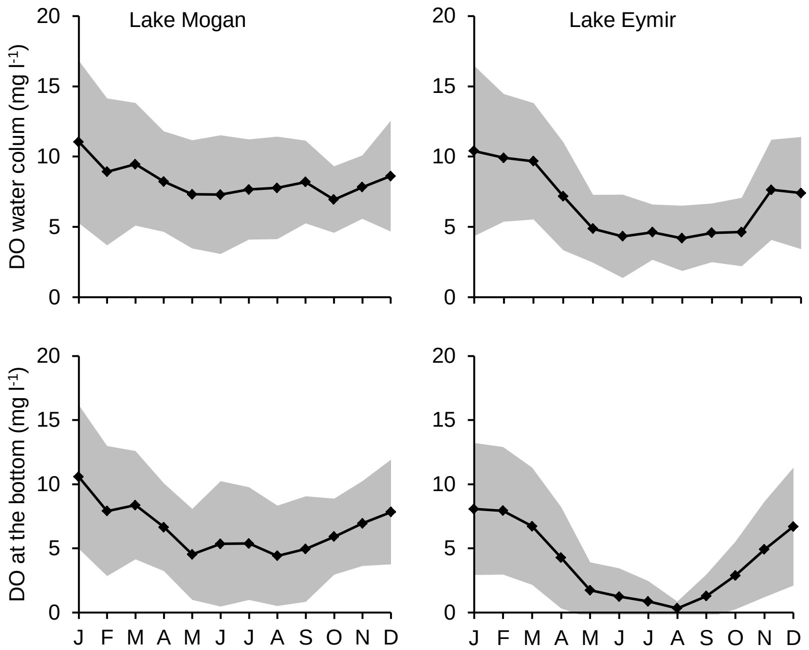


Figure 3.2: Average seasonal dissolved oxygen (DO) concentrations (mg L^{-1}) with standard deviation (grey shade) of the whole water column and at the lowest 50 cm of lake water for Lakes Mogan and Eymir.

Meteorological data were taken from the Turkish State Meteorology Service (DMI). Precipitation was measured on a daily basis at the central Ankara station. Daily evaporation was calculated based on pan evaporation measurements and seasonal pan coefficients. During the winter months, when pan evaporation was not measured, evaporation was calculated based on radiation, temperature, relative humidity and with speed with the Penman equation of an open water surface (Penman 1948). Data on the atmospheric deposition of N were taken from measurements at the Çubuk and Çamkoru stations located 100 km north of the lakes (Turkish State Meteorology Service). Estimates of TP concentrations of precipitation were taken from Koçak et al. (2010).

Daily monitoring of the lake water levels and in- and outflow flow estimates were made by the Turkish General Directorate of Electrical Power, Resources Survey and Development Administration (EIE), until 2010. The lake volume was calculated from bathymetric maps made available by the General Directorate of State Hydraulic Works (DSI). After 2011, the hydrological monitoring was transferred to the authority of DSI, but data collection was not resumed until 2013 and then only for lake water levels. Lake water levels during 2011-2013 were measured biweekly at the deepest point of the lake.

To complete the water budgets for 2011-2013, a hydrological model of the catchment for 1995-2013 was set up (Chapter IV) using the Soil and Water Assessment Tool (ArcSWAT2012) (Neitsch et al. 2011). Meteorological data, CORINE 2006 land-use data, a digital terrain model, soil data from the Harmonized World Soil Database and information provided by the Turkish Food, Agriculture and Livestock Ministry regarding the common agricultural practices in the catchment were used to initialise the model. The model was warmed up with data from 1995-1999, calibrated on monthly flow rate data from 2000-2005 (bR^2 = 0.77) and validated on data from 2006-2010 (bR^2 = 0.50). This validated model was used to estimate the flow rates in the five inflows and two outflows for both lakes between 2011 and 2013.

3.2.3. <u>Budget calculations</u>

The water budgets for both lakes were calculated on a monthly basis using the following formula:

$$V_t - V_{t-1} = I_t + P_t - O_t - E_t \pm G_t \qquad (4)$$

with V_t being the monthly average lake volume (m^3), V_{t-1} the average lake volume of the previous month, I_t the inflow volume, P_t precipitation, O_t outflow volume, E_t evaporation and G_t the groundwater flow. G_t was calculated as the residual part of the budget after taking all other factors into account and can be assumed to be groundwater seepage to and from the lake or residual surface flow. The monthly water budget was then summed up for the annual water budget. Seasonal water budgets were constructed by calculating for every month (January-December) the average over all the years of the study period (1993-1995, 1997-2013) for all the factors of the water budget.

Hydraulic residence time (HRT) was calculated on a yearly basis as:

$$HRT = V_{lake} / Outflow \qquad (5)$$

with V_{lake} being the average lake volume and "outflow" the total outflow rate in m^3 per year.

To study the effect of WLF, a distinction was made between low water level years (LWL) and high water level years (HWL). Although both lakes are naturally formed, the outflows are controlled by a gate mechanism, implying that the hydrological regimes of the lakes are artificial. For this reason, years were categorised based on annual average water levels rather than total annual precipitation. For Lake Eymir, the years 2001-2002 and 2004-2009 had an average annual water level below 5.1 m (the long-term average water level) and were classified as LWL years. The HWL years were 1993-1995, 1997-2000, 2003 and 2010-2013 with an average annual water level above 5.1 m. For Lake Mogan, the LWL years were 1993-1995, 2000-2001 and 2005-2009 with water levels below 3.8 m and the HWL years were 1997-1999, 2002-2004 and 2010-2013 with water levels above 3.8 m.

For both lakes, the nutrient budgets for SRP, TP, dissolved inorganic nitrogen (DIN) and TN were calculated on a monthly basis using the following formula:

$$N_t - N_{t-1} = I_t + P_t - O_t \pm G_t \pm R_t \qquad (6)$$

with N_t being the monthly average nutrient content of the lake (tonnes), N_{t-1} the average nutrient content of the previous month, I_t the nutrient loading through the major inflows, P_t the atmospheric deposition, O_t the nutrient efflux through the outflow, G_t the nutrient loading or efflux through the groundwater/residual flow and R_t the nutrient retention or internal loading.

The small seasonal inflows contributed 20% to the inflow volume to the lakes but were not monitored for nutrient concentrations. The nutrient loading from these inflows was estimated using average inflow concentrations of the nearby monitored streams.

The groundwater nutrient flow is not regularly monitored in the catchment. Groundwater nutrient concentrations were estimated to be equal to lake concentrations for groundwater flow out of the lake. For the groundwater flow into the lake, we used groundwater concentrations data from Altınbilek et al. (1995), who measured average TP and DIN of the groundwater in the Eymir-Mogan catchment to be 24 µg L^{-1} and 3.4 mg L^{-1}, respectively.

R_t was calculated as the remaining part of the budget after taking all other factors into account and included the removal of dissolved, organic and particulate N and P from the water column through a combination of sedimentation processes, assimilation by organisms and loss to the atmosphere (denitrification).

All factors in the nutrient balance were then divided by lake area to convert them to g m^{-2} and summed to annual nutrient budgets. Seasonal nutrient budgets were constructed by calculating for every month (January-December) the average over all the years of the study period (1993-1995, 1997-2013) for all the factors of the nutrient budget.

The changes in lake nutrient concentration (in mg L^{-1}) were compared with the changes in lake nutrient quantity (in tonnes). If lake volume does not change over time, any change in nutrient content will be accompanied by a similar change in nutrient concentration. Therefore, any discrepancy between the changes in nutrient content and nutrient concentration can be attributed to changes in water volume and thus water level. The water level effect was quantified by dividing the lake nutrient quantity by the average lake volume of the previous year and by comparing this hypothetical nutrient concentration with the observed nutrient concentration.

3.2.4. Statistical analysis

We analysed the relationships between the monthly average TP, SRP and DIN concentrations and nutrient loading and water level for the two lakes for 1997-2010. We performed an autoregressive distributed lag time series regression (ARX) analysis that takes the autocorrelation structure of the time series into account by using autoregressive integrated moving average (ARIMA) time series analysis (proc Arima in SAS). External loading, nutrient retention, water level, precipitation and evaporation were used as exogenous independent variables in the regression models after checking for multi-collinearity though precipitation and evaporation were not significant for any of the models and thus not included in the results.

3.3. Results

3.3.1. Water budgets

Lake Mogan

The average annual water depth of Lake Mogan ranged from 3 m in 2008 to 4.5 m in 2011 (Figure 3.3), corresponding to lake volumes between 12.6 hm^3 (= 10^6 m^3) and 23.8 hm^3, respectively. Seasonal WLF averaged ± 0.60 m owing to evaporative water loss combined with absence of inflows during summer (Figure 3.4).

Evaporation was responsible for the largest proportion of water loss equivalent to more than half of the average lake volume (Table 3.3). Outflow volumes were low and almost zero during 2004-2009, causing theoretical HRT to be larger than 100 years during the period of drought. HRT was approximately 1-2 years during HWL years in 1999-2000 and 2010-2012.

Table 3.3: Average water level and water budget with standard deviation for Lake Mogan for the entire study period (1993-2013), during low water level years (LWL) and high water level years (HWL).

	Lake Mogan		
	1993-2013	**LWL**	**HWL**
Water level (m)	3.8 ± 0.4	3.4 ± 0.2	4.0 ± 0.2
Inflows (hm^3 a^{-1})	9.6 ± 6.6	5.3 ± 6.4	13.0 ± 5.2
Precipitation (hm^3 a^{-1})	3.0 ± 0.8	2.5 ± 0.5	3.4 ± 0.9
Outflows (hm^3 a^{-1})	5.4 ± 8.1	2.8 ± 7.9	7.5 ± 8.3
Evaporation (hm^3 a^{-1})	9.1 ± 1.3	8.1 ± 0.6	9.9 ± 1.1
Groundwater (hm^3 a^{-1})	2.0 ± 3.7	2.9 ± 2.2	1.3 ± 4.6

Table 3.4: Average water level and water budget with standard deviation for Lake Eymir for the entire study period (1993-2013), during low water level years (LWL) and high water level years (HWL).

	Lake Eymir		
	1993-2013	**LWL**	**HWL**
Water level (m)	5.2 ± 0.7	4.5 ± 0.4	5.7 ± 0.3
Inflows ($hm^3\ a^{-1}$)	5.8 ± 7.7	0.6 ± 0.8	10.0 ± 8.6
Precipitation ($hm^3\ a^{-1}$)	0.5 ± 0.1	0.4 ± 0.1	0.5 ± 0.1
Outflows ($hm^3\ a^{-1}$)	4.5 ± 7.4	0.1 ± 0.1	8.0 ± 8.8
Evaporation ($hm^3\ a^{-1}$)	1.5 ± 0.2	1.4 ±0.1	1.6 ± 0.2
Groundwater ($hm^3\ a^{-1}$)	0.2 ± 2.0	0.3 ± 0.6	0.6 ± 2.7

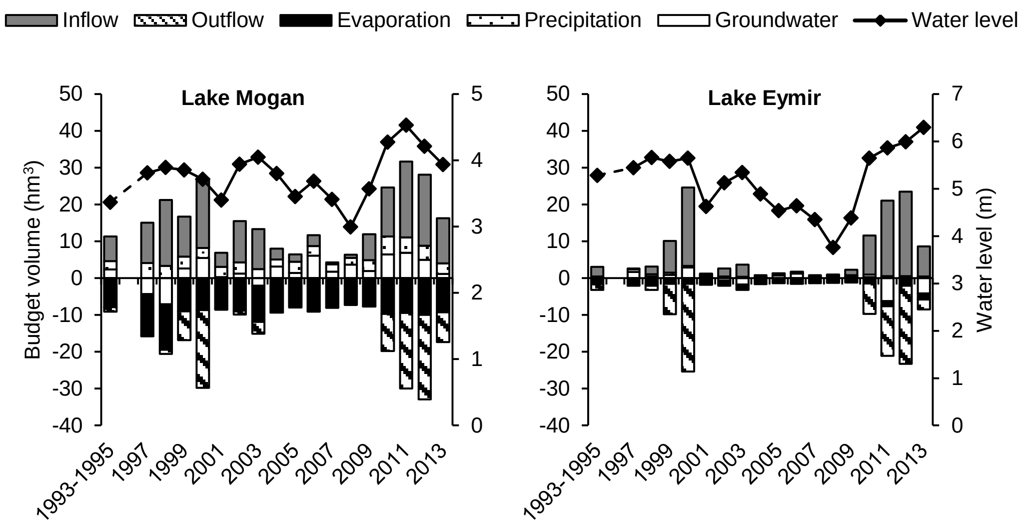


Figure 3.3: Annual water budgets (hm^3) and lake water level (m) for Lakes Mogan and Eymir (1993-1995, 1997-2013).

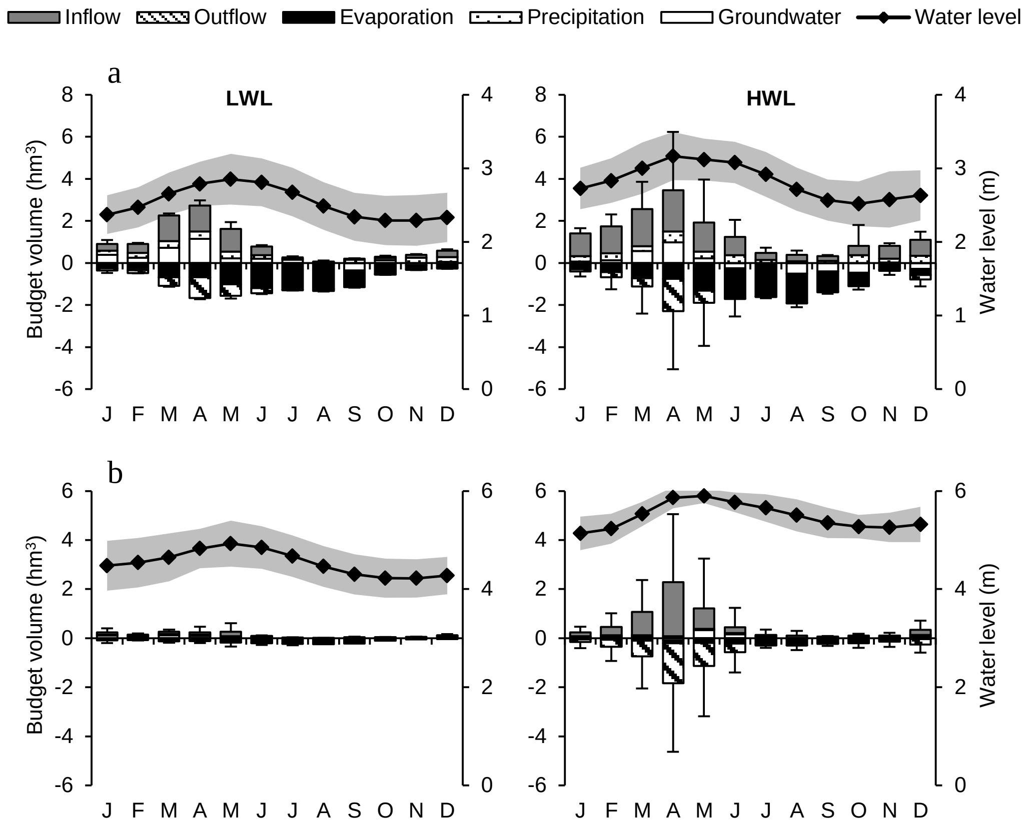


Figure 3.4: Seasonal water budgets (hm^3) and seasonal water level (m) with standard deviation (grey shade) in low water level years (LWL) and high water level years (HWL) for a) Lake Mogan and b) Lake Eymir (1993-2013).

Lake Eymir

The average annual water level of Lake Eymir fluctuated between 3.8 m (2.3 hm^3) in 2008 and 6 m (5 hm^3) in 2013 (Figure 3.3). Seasonal WLF averaged ± 0.65 m (Figure 3.4). Since the outflow of Lake Mogan is the major inflow of Lake Eymir, inflows were very limited in LWL years (Table 3.4). Most water exited Lake Eymir through the outflow, but high outflow volumes were only recorded in 1999, 2000 and 2010, causing HRT to be lower than 1 year in these years, compared with theoretical HRT >100 years in LWL years.

3.3.2. Nutrient budgets

Lake Mogan

Average nutrient concentrations for Lake Mogan are shown in Table 3.1. P budgets are shown in Figure 3.5 with marked peaks in P concentrations observed in 2004 and 2008. On average, 90% of the annual external TP loading to the lake was retained and net annual TP retention was observed in Lake Mogan except in 1999 and 2007. The time series regression analysis showed that both external loading and retention were significant factors influencing nutrient concentrations (Table 3.5).

TP concentrations tended to be higher during LWL years than in HWL years (Table 3.1) despite a two-fold higher retention in LWL years. The higher P concentrations in LWL years were caused by a peak in April driven by higher inflow concentrations, while higher retention in late spring and early summer resulted in a decrease in TP and SRP concentrations (Figure 3.7). During HWL years, the seasonal changes were smaller and, despite higher inflow volumes, external loading had limited effect due to lower inflow concentrations, though it was still around 3 times higher than average lake concentrations. Furthermore, retention was limited under HWL conditions.

Table 3.5: Regression coefficient estimates and R^2-values of the autoregressive time series analysis with exogenous variables (ARX) analysis for monthly SRP, TP and DIN concentrations in Lakes Mogan and Eymir (1997-2010), with external nutrient loading, nutrient retention and water level as independent variables (n=168). Y_{t-1} represents the autocorrelation with the previous month, ε_{t-1} represents the autocorrelation with the error term of the previous month (moving average term) (*: 0.05>p>0.01; ** 0.01>p>0.001; ***0.001>p).

	Lake Mogan			*Lake Eymir*		
Parameter	**SRP**	**TP**	**DIN**	**SRP**	**TP**	**DIN**
Y_{t-1}	0.59***	0.65***	0.66***	0.85***	0.82***	0.82***
ε_{t-1}	-0.82***	-0.82***	-0.95***	-0.60***	-0.61***	-0.56***
External loading	0.13*	0.19***	0.17***	0.09*	0.04*	0.14***
Retention	-0.19***	-0.20***	-0.20***	-0.18***	-0.13***	-0.14***
Water level	0.007***	0.02***	0.07***	0.04***	0.05***	0.10***
R^2	**0.78**	**0.79**	**0.85**	**0.88**	**0.84**	**0.84**

TN concentrations in Lake Mogan showed peaks in 2008 and 2010, while DIN had only minor peaks in 2003, 2006 and 2011. On average, 90% of the average annual external DIN loading to the lake was retained or lost by denitrification (Figure 3.6). TN retention accounted for, on average, 42% of the external TN loading (only data from 2008-2013). DIN concentrations tended to be higher during LWL years than HWL years (Table 3.1), with both higher external loading and higher retention in HWL years.

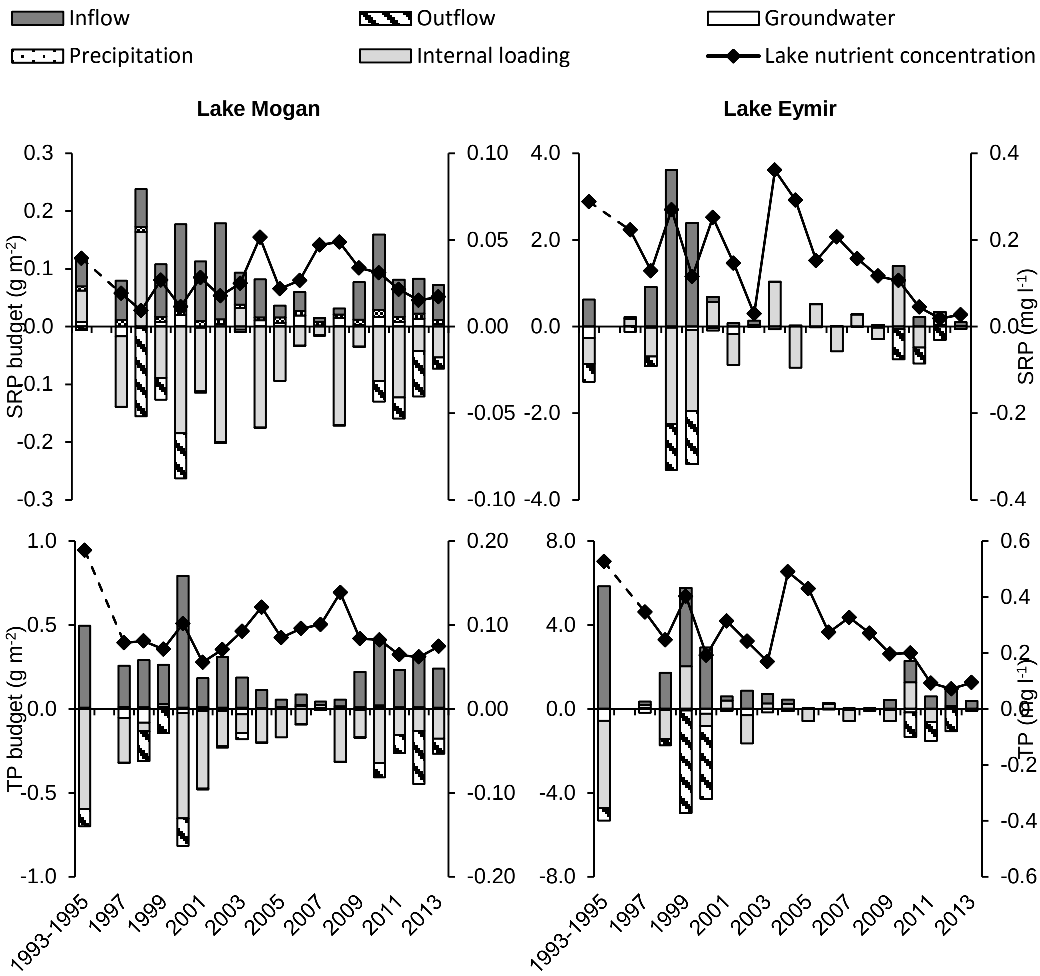


Figure 3.5: Annual nutrient budgets (g m^{-2}) and average annual nutrient concentrations (mg L^{-1}) of SRP and TP for Lakes Mogan and Eymir (1993-1995, 1997-2013).

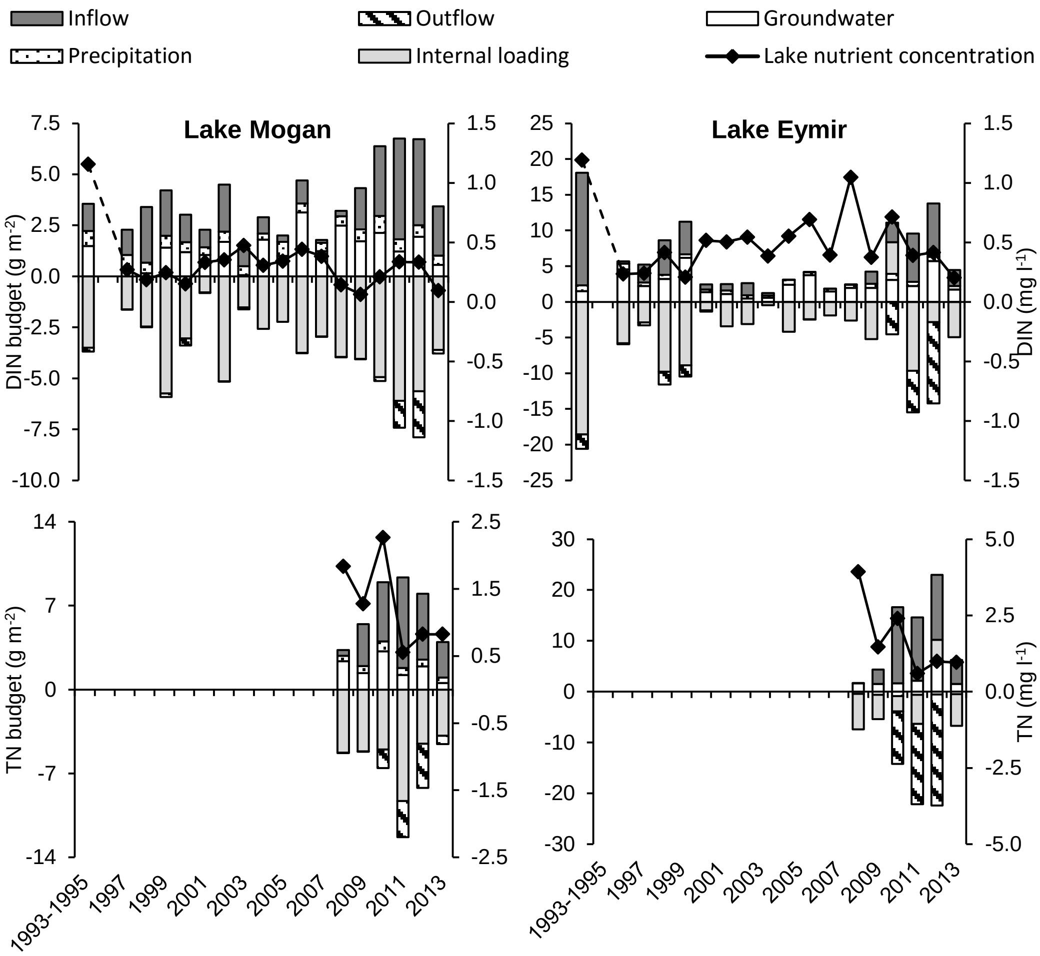


Figure 3.6: Annual nutrient budgets (g m^{-2}) and average annual nutrient concentrations (mg L^{-1}) of DIN and TN for Lakes Mogan and Eymir (1993-1995, 1997-2013).

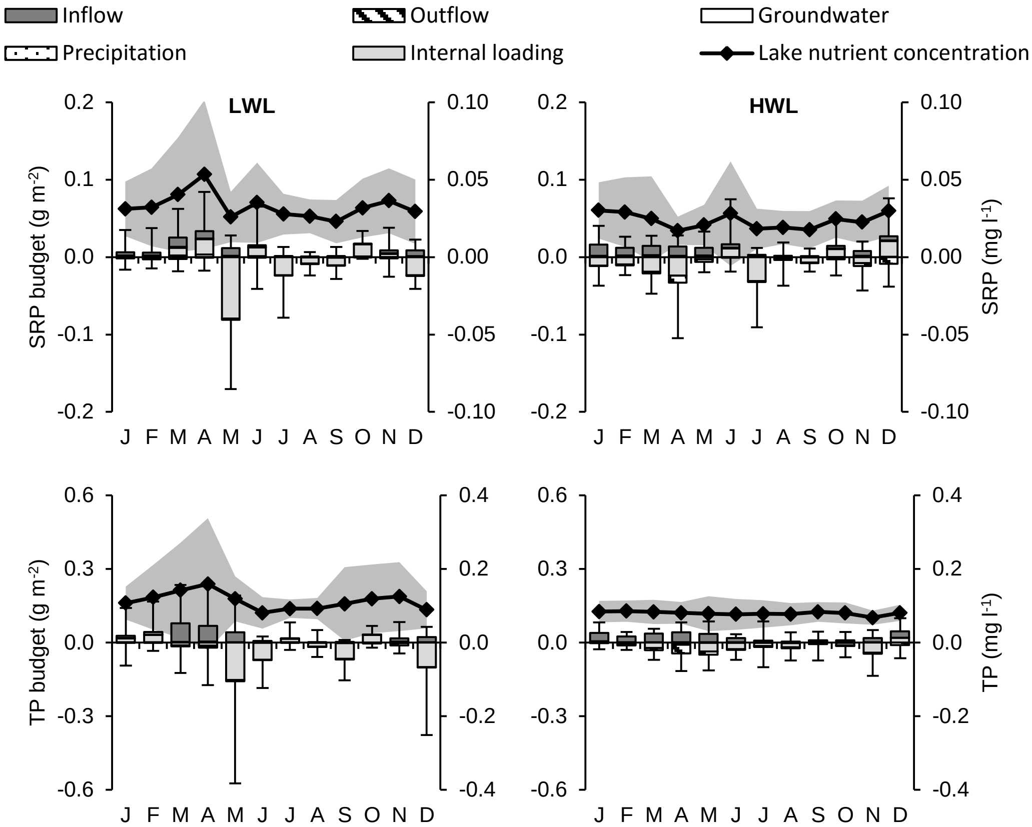


Figure 3.7: Seasonal nutrient budgets (g m^{-2}) and nutrient concentrations (mg L^{-1}) with standard deviation (grey shade) of SRP and TP in low water level years (LWL) and high water level years (HWL) for Lake Mogan.

The higher N concentrations in LWL years resulted from a peak in spring (Figure 3.8) due to the external loading with high nutrient concentrations (3 to 9 times higher than lake concentrations). Lower concentrations in summer were caused by retention. During HWL years, N concentrations were more stable throughout the year and external loading was balanced with retention during most months.

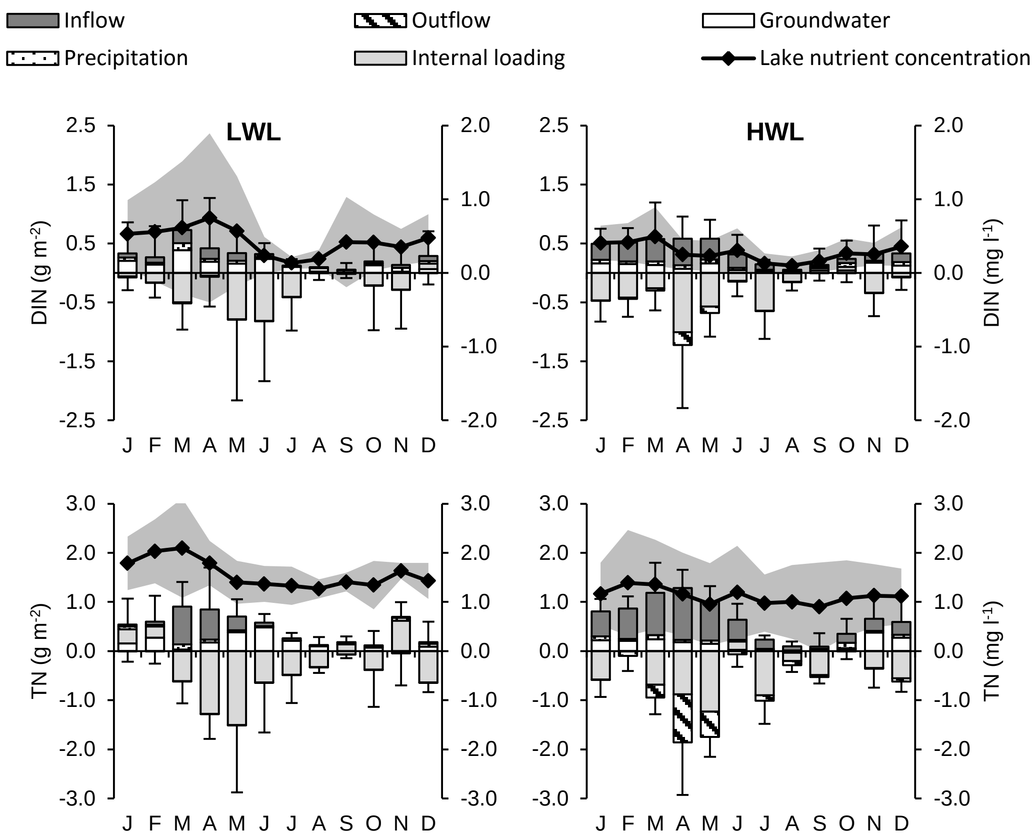


Figure 3.8: Seasonal nutrient budgets (g m-2) and nutrient concentrations (mg L^{-1}) with standard deviation (grey shade) of DIN and TN in low water level years (LWL) and high water level years (HWL) for Lake Mogan.

Comparison of nutrient quantity (in tonnes) and nutrient concentrations (in mg L^{-1}) revealed that decreasing water levels led to 5-25% higher concentrations in 2004 and 2008 than would be expected from the increase in nutrient quantity alone. In other years with increasing water levels (e.g. 2002, 2009), nutrient concentrations were 20% lower than explainable by a reduced nutrient content alone. On a seasonal scale, the evaporative water loss during summer resulted in, on average, 10% higher nutrient concentrations.

Lake Eymir

P concentrations in Lake Eymir decreased over time from high concentrations in 1993 to less than 100 µg L^{-1} in 2011, with peaks in 2004 and 2007 (Figure 3.5). Differences in lake P concentrations between LWL and HWL years were low, with slightly higher concentrations in HWL years (Table 3.1).

The external nutrient loading (tonnes per year) to Lake Eymir was lower compared with Lake Mogan, but Lake Eymir received higher areal loading (g m^{-2}) due to its smaller size. The direct nutrient loading from the town of Gölbaşı, located between the two lakes, could be quantified to 6.5 t TP and 18 t DIN in 1993-1995. After the diversion of sewage discharge in 1995, the N and P loading from Gölbaşı dropped to less than 1 t a^{-1}. An alternation was observed between net annual retention and net annual internal loading; and on average, 49% of the total TP loading was retained.

In LWL years, high internal loading in May-June resulted in high summer concentrations, while retention in late summer-autumn led to decreased P concentrations (Figure 3.9). In HWL years, low P concentrations during summer were caused by low external loading and high retention, though during spring and autumn, high external loading produced a slight increase in the P concentration (Figure 3.9).

Average inflow TP concentrations to Lake Eymir were 1.6-1.8 times higher than average lake concentrations and increased from HWL (410 ± 340 µg L^{-1}) to LWL years (690 ± 730 µg L^{-1}).

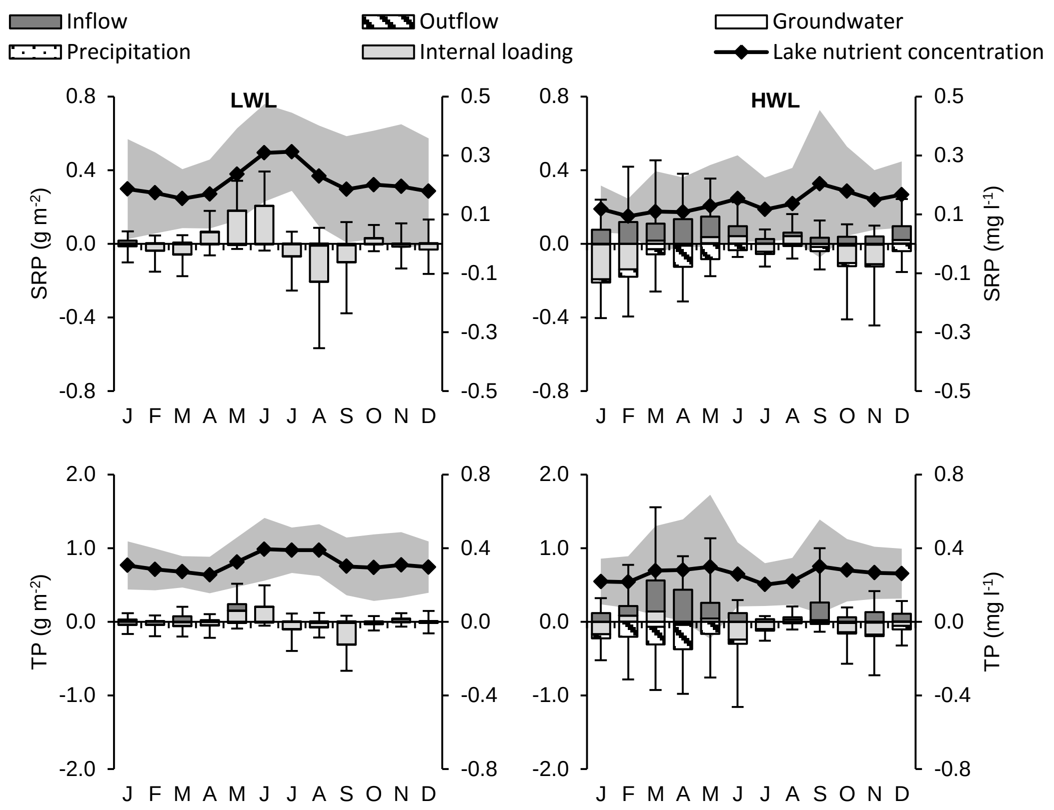


Figure 3.9: Seasonal nutrient budgets (g m^{-2}) and nutrient concentrations (mg L^{-1}) with standard deviation (grey shade) of SRP and TP in low water level years (LWL) and high water level years (HWL) for Lake Eymir.

Both TN and DIN concentrations in Lake Eymir showed a peak in 2008 followed by a slow decline (Figure 3.6). Retention encompassed 82% of the average annual external DIN loading. Regression analysis for Lake Eymir showed that external and internal loading were significant factor determining in-lake SRP, TP and DIN concentrations.

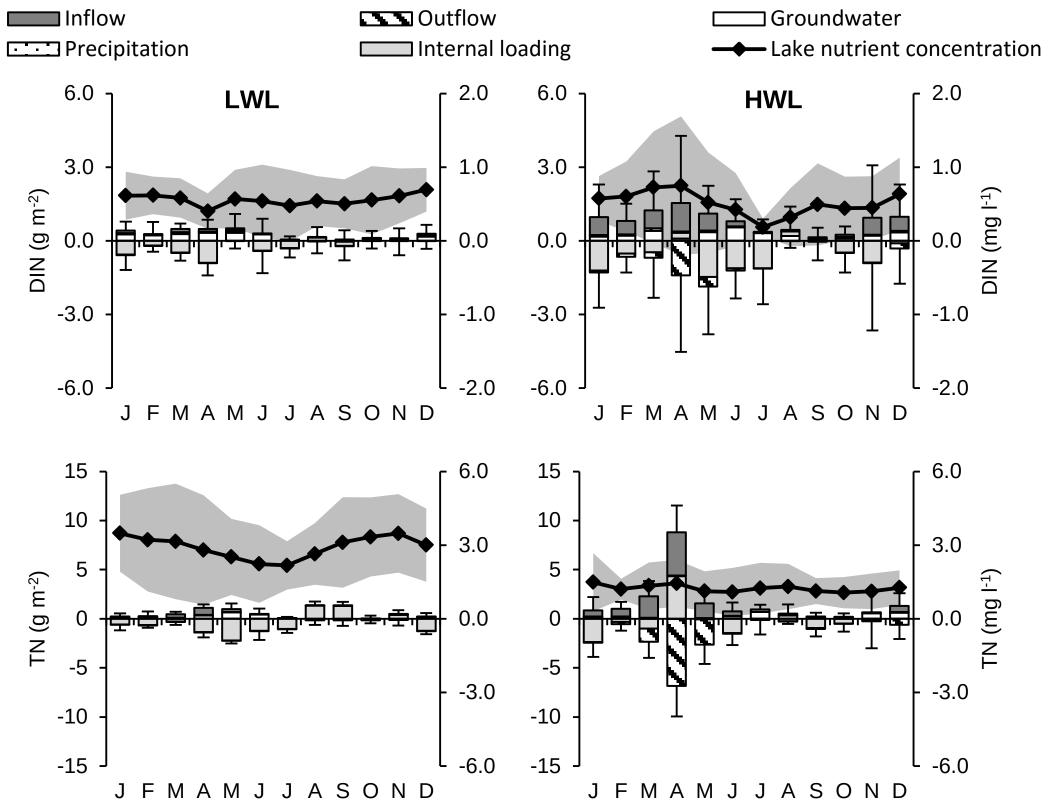


Figure 3.10: Seasonal nutrient budgets (g m^{-2}) and nutrient concentrations (mg L^{-1}) with standard deviation (grey shade) of DIN and TN in low water level years (LWL) and high water level years (HWL) for Lake Eymir.

DIN concentrations were similar in HWL and LWL years, but TN was higher in LWL than in HWL years (Table 3.1). DIN and TN concentrations in LWL years were low in summer due to retention and increased again during autumn and winter (Figure 3.10). During HWL years, this pattern was more pronounced, with external loading causing higher concentrations in autumn and winter. Lower average inflow concentrations during HWL years contributed to overall lower in-lake TN concentrations.

Decreasing water levels produced 5 and 15% higher nutrient concentrations in 2004 and 2007 than what can be expected from an enhanced nutrient quantity alone. During the lowest water level year, 2008, the concentrations were 30% higher than the nutrient quantity. External loading and retention were the main factors influencing nutrient concentrations in Lake Eymir, but the water level effect was also significant for SRP (Table 3.5).

3.4. Discussion

The main findings of our 20-year mass balance study of two shallow, semi-arid lakes are: (1) the water budgets of both lakes showed high variability between wet and dry years with a 50% reduction in lake volume from the wettest to the driest year, demonstrating the threat of drying out of the lakes and having major implications for lake water quality, (2) the six years of additional data from a period with high water levels following the prolonged drought showed that the P and N concentrations decreased with high water levels and short HRT opposite to the observations during LWL years with low external loading and prolonged HRT, (3) a shift occurred in the seasonal patterns of nutrients between HWL and LWL years. During HWL years, despite high external loading, flushing of the lakes during spring, combined with retention during summer, resulted in decreased concentrations of nutrients. During LWL years, however, nutrients accumulated in spring and increased during summer due to internal loading and evaporative water loss.

The water budgets showed large annual variation in the hydrology of Lakes Mogan and Eymir. During LWL years, inflow volumes to Lake Mogan were 30% lower than during HWL years due to lower precipitation (-25%) and higher evaporation (+ 5%). This pattern is similar to that in other semi-arid lakes in, for instance, Greece (Myronidis et al., 2012), Italy (Naselli-Flores and Barone, 2005) and Israel (Zohary and Ostrovsky, 2011). For Lake Eymir, the differences between LWL and HWL years were even more pronounced given its more artificial water regime, with major inflows occurring only during a few years, leading to 85% lower inflow volumes in LWL years.

The two lakes had a similar seasonal hydrological pattern, with an average seasonal water level change of ± 0.6 m, mainly caused by evaporation in summer. Despite a slow increasing trend, evaporation did not vary much between years and the extent of the annual water level fluctuations can be ascribed to changes in net external hydraulic loading during winter and spring (Pham et al., 2009). In LWL years, hydraulic loading (from inflows and groundwater) was unable to restore the water volume after the evaporative water loss during the previous summer. On average, during winter and spring, at least 9 hm^3 was needed to flow into Lake Mogan and 1.5 hm^3 into Lake Eymir to maintain stable annual water levels, equalling about half of the average water volume. The outflow gates were closed during the low water level years, but even so water levels steadily declined. This illustrates the severity of the threat of drying out of the lakes in the semi-arid climate zone if longer periods of low inflows will occur as evaporative water loss is high and not compensated by low volume of inflows. Given the direct relation between precipitation and inflow volumes and between temperature and evaporation, with climate change the inflow volumes are expected to decrease and evaporation losses are expected to increase, due to increasing temperatures and decreasing precipitation as anticipated from the global climate change predictions for this region (Önol and Unal, 2014).

Changes in nutrient concentrations in the lakes were influenced to a large extent by changes in external loading. In Lake Eymir, after diversion of sewage effluent, external loading decreased, causing nutrient concentrations to decrease by 50%. Except this sewage diversion, many of the changes in external loading were related to hydrological factors such as precipitation and inflow volumes. Besides the impact on external loading, hydrology influenced nutrient dynamics in the lakes due to the changes in water level and the consequent effects on internal loading and retention.

We found that the water balance had an important influence on the nutrient concentrations in the lakes. During HWL years, external loading was high due to higher inflow volumes and nutrient concentrations in both lakes were mainly determined by the nutrient concentrations of the inflow and by retention, while during LWL years external loading was low due to lower precipitation and lower inflows so evaporation and internal loading from the sediment played an important role. During LWL years, although both lakes were characterised by low inflow volumes, large peaks in N and P occurred, probably due to low water levels and high HRT. Our results showed a peak in N and P in 2008, the driest year, which is in accordance with the previous results until 2007 reported by Özen et al., (2010), while nutrient concentrations decreased during the following HWL period from 2009 onward. These results show that eutrophication of the lakes can occur during years with low water levels even if external loading is low. This is similar to results obtained on Canadian semi-arid prairie lakes, which showed higher nutrient concentrations under dry conditions, while high inflows with lower nutrient concentrations flushed the lakes in wet years (Starks et al., 2014).

Differences between the LWL and HWL years caused changes in the seasonality of the nutrient budgets. In Lake Mogan, external loading was observed during winter and spring in both LWL and HWL years, but inflow nutrient concentrations tended to be higher in LWL years, causing a spring nutrient peak of N and P during LWL years. During summer, low water level conditions with high HRT generally enhanced retention (Koiv et al., 2011). Retention of SRP and TP increased from HWL to LWL years, and retention of TP almost doubled under LWL conditions to 100% of the average external input. In HWL years the spring nutrient peak and summer retention were lower, leading to less seasonal variation.

In Lake Eymir, the seasonal pattern for N was more pronounced in HWL years with external loading in spring and retention in summer. During LWL years, external loading was almost absent, but retention in summer still caused lower nutrient concentrations. In the years with low water levels, DIN concentrations also showed peaks in some years, but unlike the case of P, internal loading during summer did not play a role, likely because N did not accumulate to the same extent in the sediment. Higher DIN concentrations were rather caused by evaporative water loss and lower retention.

For P, there was a substantial difference in seasonality between LWL and HWL years in Lake Eymir. Inflows and external loading were absent in LWL years and internal loading during summer played an important role driving the increases in nutrient concentrations. A similar increase in nutrient concentrations due to higher internal loading in summer has also been observed in other lakes with warm summers (Mitraki et al., 2004; Mäemets et al., 2006) and is in line with observations made in northern temperate lakes where internal P loading during summer results in higher P concentrations (Graneli 1999; Søndergaard et al., 2003). The internal loading in summer between the years 2003 and 2009 may reflect overall low water levels, high hydraulic water retention time (Brett and Benjamin, 2008) and increased resuspension (Nõges et al., 1998). The difference in the response to low water level conditions between the lakes can be explained by the large accumulation of P in the sediment of Lake Eymir due to its 25-year history of untreated sewage effluent discharge. After diversion of the sewage effluent this large stock of nutrients in the sediment likely caused the shift from retention to internal loading. Furthermore, the higher depth of Lake Eymir, with frequent occurrence of deoxygenation in the hypolimnion in summer (Fig. 3), creates higher P release from the sediment under anoxic conditions in both HWL and LWL years (Romero et al., 2002; Schernewski, 2003; Nowlin et al., 2005).

N retention was highest during the HWL years with high external loading and lowest between 2003 and 2008 with low water levels, demonstrating the relationship between external loading and N retention (Søndergaard et al., 2003), while the relative proportion retained of external N input was higher in LWL years.

The regression analysis showed that for both lakes, low water levels mostly had an indirect effect on nutrient concentrations through decreased external loading and increased internal loading. Nonetheless, the increase in nutrient concentrations during summer in LWL years was further augmented in both lakes by evaporative water loss during summer, causing the nutrient content to be concentrated in a smaller water volume (Pham et al., 2008; Özen et al., 2010). This effect could be quantified as a 5-20% increase in nutrient concentrations on an annual basis and 10% on a seasonal basis, showing that a reduction in lake volume may have a significant impact on nutrient levels and can generate a shift to the eutrophic state even under low external loading conditions. This pattern was observed in Lake Eymir when a water level drop in 2002-2004 weakened the recovery obtained through biomanipulation after sewage diversion and shifted the lake back to the turbid state from the macrophyte dominated clear water state (Beklioğlu and Tan, 2008).

Macrophytes can play an important role in N retention, either through direct nutrient uptake or by creating conditions favourable for denitrification (Kankaala et al., 2002; Desmet et al., 2011). However, similar to Özen et al., (2010), our data did not reveal a relationship between macrophyte development in the lake and DIN retention. Macrophyte PVI was highest in LWL years, likely because low water levels enhanced underwater light availability and supported plant growth (Bucak et al., 2012), but DIN retention was lower than in the HWL years due to the higher external loading.

During biomanipulation in Lake Eymir (1998-1999 and 2006-2013) on average around 11 t of fish was removed annually. Based on estimates of average nutrient content of fish (Tanner et al., 2000) this corresponds to a direct removal of 50 kg P and 220 kg N annually, which is small compared to average nutrient stocks in the water column of the lake (on average 1 t TP and 6 t TN) and the surface sediment actively in contact with the water (average internal loading from the sediment 0.3 t TP and 4.4 t TN per year).

The biomanipulation, however, indirectly affects the nutrient processing, which is to be expected as clear water conditions can encourage macrophyte and benthic algal development, leading to higher uptake and higher oxygenation of the sediment (Woodruff et al., 1999). Following both fish removals, TP and DIN concentrations declined, possibly reflecting reduced sediment resuspension caused by bottom feeding fish (Gulati and Van Donk, 2002), while the increased macrophyte development after the first biomanipulation increased the nutrient uptake and thus lowered the nutrient concentrations in the water column.

The groundwater contribution to the lakes, as calculated from the water budgets, was estimated to ± 25% of the input to the lake and ± 15% of the output from the lake. These estimates are higher than the ± 5% groundwater contribution found in a groundwater modelling study of both lakes by Yağbasan and Yazicigil (2009), possibly reflecting the fact that the groundwater term in our water budgets includes residual surface flow. The nutrient concentrations of the groundwater in the Lakes Mogan and Eymir joint catchment are not monitored and had to be estimated based on literature data. These estimates directly affect the calculation of retention as the remaining part of the nutrient budgets. If estimates of groundwater concentrations are changed by +/- 100% (i.e. doubled or reduced to zero), our retention estimates show a change of +/- 5-15% for P and +/- 40% for N for Lake Mogan. For Lake Eymir, the changes would be +/- 8% for P and +/- 60% for N. While the retention quantities may depend on which estimates for groundwater loading are used, the overall annual and seasonal trends in retention/internal loading remain the same.

In our study, the relationship between extreme drought or wet conditions and the lake nutrient concentrations was not entirely straightforward. The lowest water level was recorded in 2008 in both lakes, but the highest increases in P concentrations were observed in Lake Eymir in 2004 and 2007 where water levels were only moderately lower than in the adjacent years. In fact, P concentrations in Lake Eymir decreased in 2008 due to high retention in the previous autumn. In Lake Mogan, P concentrations increased in 2008 but decreased again to the previous levels in 2009 and the same was observed for DIN concentrations in Lake Eymir. During the wet period 2010-2013, both P and N concentrations declined slowly, but nutrient reduction started in 2009, a year with limited inflow and limited flushing. Therefore, the extreme years did not necessarily lead to a dramatic shift in nutrient dynamics; rather periods of low and high water levels created the conditions that could lead to either high nutrient concentrations due to internal loading or low nutrient concentrations due to flushing of the lake, similar to the delayed response of lakes to drought and wet conditions observed by Starks et al., (2014) in semi-arid prairie lakes.

3.5. Conclusion

The climate change scenarios for Turkey predict higher evaporation and lower precipitation in the coming decades. Based on the water and nutrient balance, this will pose a big threat to shallow lakes, with implications for water quality.

We observed changes in the seasonal nutrient budgets between high water level and low water level conditions. In years with high water levels, external loading was high in winter and spring, but inflow concentrations were lower and in-lake nutrients were also lost through retention. In dry years, external loading with high nutrient concentrations of the inflows increased nutrient levels and when no external loading occurred in the driest years, retention and internal loading drove the nutrient balance and led to high nutrient concentrations. While retention occurred in spring, high nutrient concentrations were found in summer due to internal loading if large amounts of nutrients were present in the sediment.

Our results confirm the previous conclusions drawn by Özen et al., (2010) that nutrient concentrations in shallow lakes under semi-arid climate conditions can increase under low water level conditions despite low external loading, these peaks being related to increased internal loading and accumulation of nutrients due to evaporative water loss. Furthermore, long drought periods with declining water levels will lead to higher nutrient concentrations, which may enhance eutrophication problems even if measures to reduce external loading are implemented.

CHAPTER IV

IMPACT OF CLIMATE CHANGE ON A SHALLOW MEDITERRANEAN LAKE USING CATCHMENT AND LAKE MODELLING APPROACHES

4.1. Introduction

The Mediterranean climate zone is considered a hot-spot to be markedly affected by climate change (Giorgi & Lionello, 2008) emphasizing projected major increases in air temperature and reductions in precipitation in this century (Erol & Randhir, 2012). For the semi-arid region in central Turkey; a 3-4 ºC increase in temperature and a 10% decrease in precipitation is projected between 2070 and 2100 compared with 1960-1970 (Önol & Unal, 2014). Such changes will have profound effects on the hydrology of the shallow lakes in the region. Higher temperatures will cause increased water loss through evaporation, while lower precipitation can lead to decreased inflows and lower runoff, potentially leading to lower nutrient loading to the lakes. Rainfall will be concentrated in fewer but more extreme events, however, which may have the opposite effect resulting in a potentially higher nutrient loading (Jeppesen et al., 2009, 2011).

The effects of the global climate change are expected to enhance eutrophication of shallow lakes (Jeppesen et al., 2009; Moss, 2012). Direct effects on the lakes can be expected from higher water temperature, creating higher primary production and favouring certain cyanobacteria species (Paerl & Huisman, 2008; Wagner & Adrian, 2009). Higher temperatures can also affect the stratification of lakes producing lower oxygen levels and higher internal loading (Blenckner et al., 2007). Changes in zooplankton and fish communities have been observed when moving from a temperate to a warmer climate (Gyllström et al., 2005; Meerhoff et al., 2007a; Brucet et al., 2012). Higher temperatures cause higher water loss through evaporation which is already a major water loss from shallow lakes under semi-arid conditions. Combined with the lower inflows from the catchment, increased evaporation will lead to lower water levels and increased salinity with many effects on the ecological processes in lakes (Jeppesen et al., 2015). Water loss through evaporation increases nutrient concentrations in the lake, leading to eutrophication problems even when external loading is low (Özen et al., 2010).

In this study we aimed to estimate the effect of the projected climate change on shallow Lake Mogan located on the Anatolian plateau in Central Turkey using a modelling approach. In order to predict the impact of climate change on the water and nutrient balance of the lake a combined approach using catchment modelling and lake modelling was implemented. We used the Soil and Water Assessment Tool (SWAT) to assess the impact of changes in temperature and precipitation on the hydraulic and nutrient loading to the lake. We then used the modelled changes in flow rate and nutrient loading as input to the PCLake model to evaluate the effects of changes in water temperature, evaporation, inflow volume and nutrient loading on nutrient concentration, macrophyte development and phytoplankton biomass in the lake.

4.2. Methods

4.2.1. Study area

The study area encompasses the catchment of Lake Mogan (39º 47’N 32º 47’E) located south of Ankara on the Central-Anatolian plateau in Turkey. The local climate is characterised as arid cold steppe (Peel et al. 2007) with an average daily temperature of 12 °C and an average yearly total rainfall of 408 mm (Turkish Meteorological Office), but there are large seasonal differences with hot and dry summers (23 °C, 20 mm rainfall per month) and cold winters (2 °C), with most rainfall in winter and spring (43 mm rainfall per month).

Lake Mogan has a surface area of 5.6-8 km^2, depending on the water level. The average mean depth was 2.4 m and ranged from 1.6 m to 3.5 m. Between 1997 and 2010, the average TP concentration was 90 µg L^{-1} and the average TN concentration 1.69 mg L^{-1}. The average chl-a concentration was 14 µg L^{-1}. Percentage volume inhabited by aquatic macrophyte (PVI) during 2001, 2004 and 2008, all years with low water levels, was higher than 40%, whereas few plants were observed during the wet period from 2009 to 2013. The catchment has an area of 940 km^2 and contains three major subbasins (Figure 4.1): the Çölovası stream with the Dikilitaş and İkizce reservoirs to the south of the lake, the Yavrucak stream to the southwest and the Sukesen stream to the east. These major inflows contribute 90% of the measured flow to Lake Mogan, while several smaller and mostly seasonal inflows accounting for the remaining flow Table 4.1.

The elevation of the basin ranges from 973 m at the outflow of the lake to 1700 m in the Sukesen subbasin in the eastern mountain range. The basin has mostly gentile slopes, 83% of the slopes being smaller than 10%. Only in the Sukesen, subbasin slopes are predominantly higher than 10% and exceed 40%.

The major land use in the basin is agriculture with non-irrigated wheat farming as the dominant crop (71%). Grasslands cover 16% of the basin, with barren lands and wetlands accounting for 2% each. The remainder of the basin is build-up areas (8%), mostly in the town of Gölbaşı and the surrounding industrial zones.

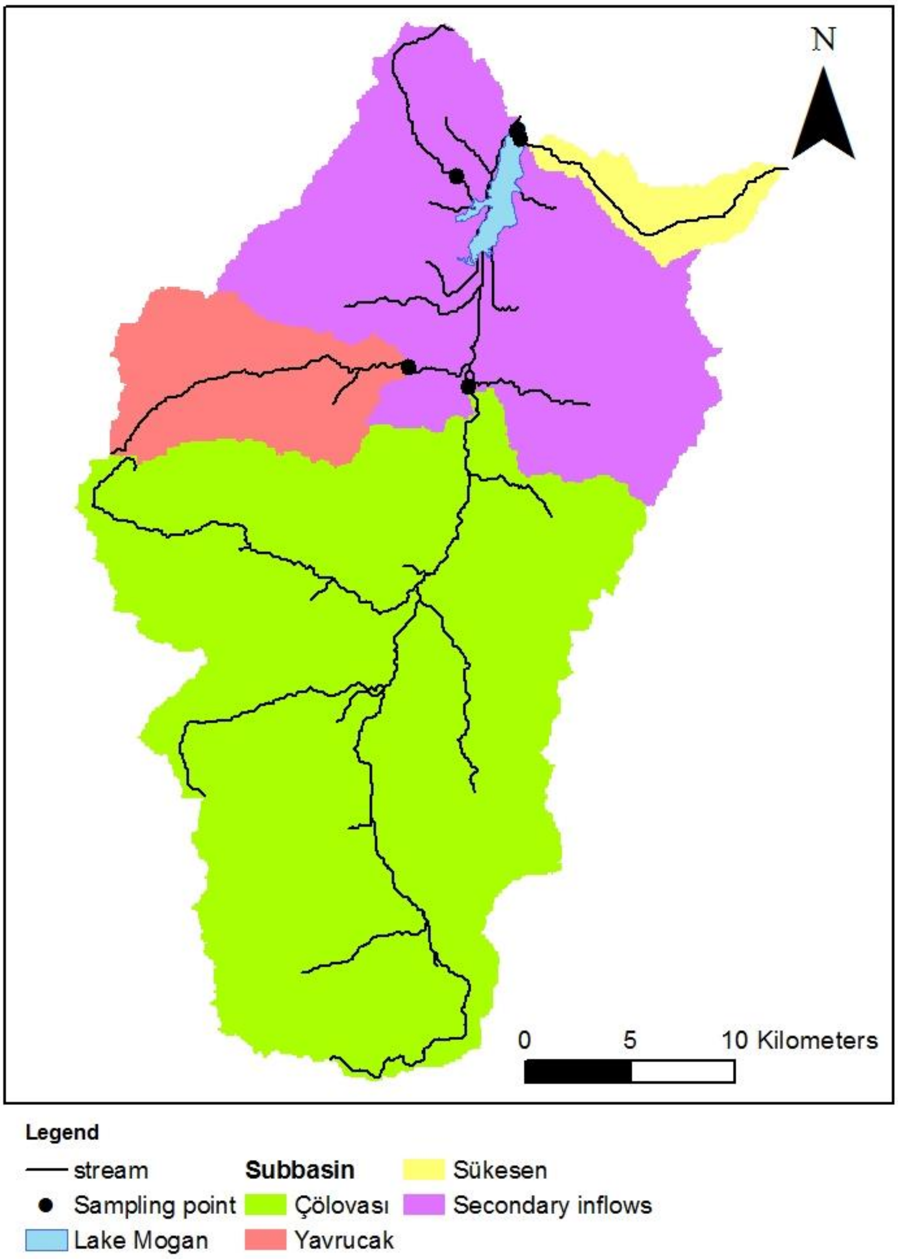


Figure 4.1: Overview of the Lake Mogan basin showing the subbasins and sampling points.

Table 4.1: General characteristics of the Lake Mogan basin and its subbasins

	Basin	**Sukesen**	**Yavrucak**	**Çölovası**	**secondary inflows**
Area (km^2)	940	31	83	549	278
Average altitude (m.a.s.l.)	1126	1271	1168	1124	1099
Major land use	agriculture (71%)	grassland (58%)	agriculture (67%)	agriculture (77%)	agriculture (68%)
Average flow (m^3/s)	0.22	0.03	0.08	0.08	0.02

4.2.2. Model set-up and calibration

To assess the impact of climate change scenarios on Lake Mogan, the catchment model SWAT (Arnold et al., 1998) and the lake model PCLake (Janse & Van Liere, 1995) were used in a combined approach. The effects of climate change scenarios on the hydraulic and nutrient loading coming from the catchment to the lake were modelled with SWAT. The output of SWAT related to inflows to the lake was then used as input in PCLake to model the effects of changes in water temperature, hydraulic loading and nutrient loading on the internal chemical, physical and biological processes in the lake.

SWAT is a physically based, semi-distributed model aimed at analysing the impact of different land use forms, management practices and climate effects on the hydrological, sediment and chemical dynamics of a hydrological catchment (Neitsch et al., 2011). The model simulates physical processes related to the water and nutrient cycle based on input data from the catchment. The ArcSWAT version 2012.10.13 was used to model the Lake Mogan basin.

The lake catchment was delineated using a digital elevation model **(**DEM) with a resolution of 30 m. Streams were generated based on the DEM by assigning 600 ha as the minimum drainage area required for a new stream. Subbasins outlets were designated at the outflows of the three reservoirs (Mogan, Dikilitaş, Ikizce), at the Yavrucak, Çölovası, Sukesen and all the remaining inflows to Lake Mogan to create a total of 18 subbasins. Based on the DEM, five slope classes were assigned **(**< 5%, 5-10%, 10-20%, 20-40% and > 40%).

Soil data was taken from the Harmonized World Soil Database and matched with the closest soil type in the SWAT database based on texture and soil properties. Regarding land cover, data from the CORINE 2006 land-use data set was used. Based on field observations the area of irrigation agriculture was reduced manually in ArcGIS and the extent of build-up area was increased to reflect recent developments. The agriculture land use class was further divided into wheat (91%) and barley (9%), the dominant crops in the basin. Information provided by the Turkish Food, Agriculture and Livestock Ministry was used to model the farming practices in the basin (Table 4.2). The overlay and intersection of the different land use, soil and slop classes resulted in 585 hydrological response units (HRU), which are the basic simulation unit of the SWAT model (Neitsch et al., 2011).

Daily data on precipitation, wind speed, maximum and minimum temperature, solar radiation and relative humidity was taken from the central Ankara station of the Turkish State Meteorology Service (DMI), which is located 30 km north of the catchment.

Table 4.2: Management operations as entered in SWAT for winter wheat and winter barley in the Lake Mogan basin.

	winter wheat	winter barley
31 March	urea 10 kg/ha	N fertilisation 33-0-0 90kg/ha
30 April	N fertilisation 33-0-0 45kg/ha	N fertilisation 33-0-0 90kg/ha
15 July		harvest
31 July	harvest	
15 October	tillage mold plow 2 way 4-6b	tillage mold plow 2 way 4-6b
30 October	planting WWHT	planting WBAR
1 November	P fertilisation 00-15-00 400 kg	P fertilisation 00-15-00 400 kg

The model was set up for the 1995-2010 period. The first five years were used as warmup for the model, 2000-2005 was the calibration period and 2006-2010 the validation period. The SWAT model was calibrated and validated based on daily flow measurements for the three major inflows provided by the Turkish General Directorate of Electrical Power, Resources Survey and Development Administration (EIE). Calibration for nutrient loading was performed for NO_3 and SRP, since data on organic N and P was limited. Calibration was done based on total monthly NO_3 and SRP loading, which was calculated from monitoring data on the major streams obtained between 2000 and 2010 (see paragraph 3.2.2. for more details).

Calibration of the SWAT-model was done using the Sequential Uncertainty Fitting **(**SUFI-2) procedure in the Soil and Water Assessment Tool Calibration and Uncertainty Analysis Programs **(**SWAT-CUP) (Abbaspour et al., 2007; Abbaspour, 2012). Before calibration, a global sensitivity analysis was performed and 32 parameters sensitive to flow rate or nutrient loading were identified in at least one subbasin of the three major inflows (Table 4.3).

During the calibration, weights were assigned to the inflows based on their respective discharge and nutrient loading rates in the objective function to determine the best fit. Best fit was determined based on the bR^2 (slope of the regression relation x R^2 coefficient) and the Nash-Sutcliffe (Nash & Sutcliffe, 1970) model efficiency coefficients.

Table 4.3: Calibration parameter set of the calibrated SWAT-model with fitted values for the three subbasin of the Lake Mogan catchment and compared to the default value automatically set by the model for each parameter

Parameter	**description**	**default**	**flow 11**	**flow 17**	**flow 19**
SFTMP.bsn	Snowfall temperature	1	1.14	1.14	1.14
SMTMP.bsn	Snow melt base temperature	0.5	1.9	1.9	1.9
SNO50COV.bsn	Fraction of snow volume represented by SNOCOVMX that corresponds to 50% snow cover	0.5	0.3	0.3	0.3
SNOCOVMX.bsn	Minimum snow water content that corresponds to 100% snow cover [mm]	1	247	247	247
TRNSRCH.bsn	Reach transmission loss partitioning to deep aquifer	0	0.23	0.23	0.23
SURLAG.bsn	Surface runoff lag time [days]	4	0.56	0.56	0.56
CNOP{[],1}.mgt	SCS runoff curve number for moisture condition II: specific for planting		58	55	55

Table 4.3 (continued)

CNOP{[],5}.mgt	SCS runoff curve number for moisture condition II: specific for harvest		75	75	74
SOL_AWC(1).sol	Available water capacity of the soil layer 1	0.16			0.19
SOL_AWC(2).sol	Available water capacity of the soil layer 2	0.09			0.14
CH_N2.rte	Manning's n value for main channel	0.014	0.07	0.045	0.065
ALPHA_BNK.rte	Base flow alpha factor for bank storage [days]	0	0.025	0.03	0.15
CH_K2.rte	Effective hydraulic conductivity [mm/hr]	0	120	115	113
CH_K1.sub	Effective hydraulic conductivity in tributary channel [mm/hr]	0	108	100	116
GW_DELAY.gw	Groundwater delay [days]	31		79	222
GWQMN.gw	Threshold depth of water in the shallow aquifer required for return flow to occur [mm]	1000		760	1097
GW_REVAP.gw	Groundwater "revap" coefficient	0.02	0.09	0.08	0.095
RES_K.res	Hydraulic conductivity of the reservoir bottom [mm/hr]	0			0.235
GWSOLP.gw	Concentration of soluble phosphorus in groundwater contribution to streamflow from subbasin (mg P/l)	0	0.28	0.14	0.001
NPERCO.bsn	Nitrogen percolation coefficient	0.2	0.07	0.07	0.07
MANURE_KG{[],9}.mgt	Dry weight of manure deposited daily ((kg/ha)	0	7.5	31	2.25

Table 4.3 (continued)

ERORGP.hru	Organic P enrichment ratio	0	0.03	2.8	0.3
ERORGN.hru	Organic N enrichment ratio	0	1.1	2.7	
N_UPDIS.bsn	Nitrogen uptake distribution parameter	20	58	58	58
FRT_SURFACE{[],3}.mgt	Fraction of fertilizer applied to top 10mm of the soil	0.2	0.07		
FRT_KG{[],3}.mgt	Amount of fertilizer applied to HRU (kg/ha)	depends on land use		1.1 x default	
LAT_ORGP.gw	Organic P in base flow (mg/l)	0	4.75	4	
LAT_ORGN.gw	Organic N in base flow (mg/l)	0	2.47	5.5	
SHALLST_N.gw	Concentration of nitrate in groundwater contribution to streamflow from subbasin (mg N/l)	0	9	6	
SDNCO.bsn	Denitrification threshold water content	1.1	1	1	1
HLIFE_NGW.gw	Half-life of nitrogen in groundwater (days)	0		150	
RCN.bsn	Concentration of nitrate in precipitation (ppm)	0	1	1	1

PCLake is a dynamic ecosystem model that incorporates both the biotic and abiotic components of the lake ecosystem. The primary purpose of the PCLake model was to simulate the shifts between the alternative stable states of shallow lakes and to determine the critical loading levels that cause these shifts (Janse & Van Liere, 1995). The basic structure of the model consists of a mixed water column, the top layer of the sediment (10 cm) and the key biotic factors and abiotic factors. The model is developed for shallow non-stratifying lakes so no other horizontal or vertical dimensions are taken into account. The DATM (Database Approach To Modelling) version of PCLake was used (Mooij et al., 2014).

The hydrological inputs to the model consist of inflow data, outflow data and evaporation. Daily evaporation was calculated based on pan evaporation measurements and seasonal pan coefficients. During the winter months, when pan evaporation was not measured, evaporation was calculated from radiation, temperature, relative humidity and wind speed using the Penman equation for an open water surface (Penman, 1948). Inflow data consisted of the combined hydraulic loading of all the inflows, direct precipitation and groundwater flow into the lake. Outflow data included the discharge from the lake through the regulated outflow and groundwater seepage from the lake. Groundwater inflow and seepage was estimated as the remaining factor of the water budget of the lake, taking into account the changes in the lake water level provided by EIE.

In addition to evaporation, PCLake required water temperature, wind speed and solar radiation as meteorological inputs. Daily water temperature was calculated based on the relationship between daily average air temperature and water temperature measured on sampling days twice a month during 1997-2010. Nutrient loading to the lake was entered into the model for NH_4, NO_3, orgN, PO_4

and orgP based on linear extrapolation of measurements taken at the major inflows between 1997 and 2010.

The model was calibrated and validated for the 2000-2010 period, with the years from 1997 to 1999 serving as warmup period. The state variables of the model were initialised based on sampling data for January 1997 for N, P, silicate, dissolved oxygen, lake fetch and water level. All other state variables, including the sediment parameters were initialised by running the model for 25 years with average hydraulic and nutrient loading.

Sampling data for Lake Mogan between 2000 and 2010 were used to calibrate and validate the model. Water samples were analysed for N and P, as well as chl-a. Macrophyte coverage has been measured in both lakes once a year in August or September since 2001 using plant surface coverage, height and water depth (Canfield et al. 1984). More details on sampling and data analysis can be found in paragraph 3.2.2.

Before calibration, a one-at-a-time sensitivity analysis was performed. In every iteration of the model, one parameter was changed by adding or subtracting 20% of the default value, and for every iteration the change in the response variables (SRP, TP, DIN, chl-a and plant coverage) was determined. All parameters were then ranked according to sensitivity for each response variable and the 16 most sensitive parameters were used in the calibration.

For model calibration, the approach described in Nielsen et al. (2014) was adopted. In this approach, the aim of the calibration is not to find the single calibration parameter set that gives the best replication of the observed data, but rather a range for each calibration parameter that takes into account the uncertainty related to determining the parameter values. Complex models like

PCLake may have several parameter combinations that reproduce observations to a certain extent, but they may diverge in model predictions when scenarios are applied. For this reason, 250 runs of PCLake were combined, with a random value within a certain range applied to each parameter in the calibration set for each run.

Calibration aimed to include > 75% of observed chl-a and plant coverage measurements and >50% of the observed values for N and P in the simulation band, while keeping the parameter ranges as small as possible within a -20 to +20% range. The simulation band was determined by selecting the highest and lowest value simulated at each time step after exclusion of the 2.5% most extreme values at the top and bottom. The range for each parameter in the calibration parameter set and the default value of the original model calibrated for a set of Dutch and Danish lakes (Janse, 2005) is presented in Table 4.4.

Table 4.4: Calibration parameter set with fitted ranges for the calibrated model of Lake Mogan with PCLake

parameter	default	min	max	parameter	default	min	max
cThetaSet	1.01	1.01	1.212	cMuMaxGren	1.5	1.5	1.8
cThetaMinW	1.07	0.856	1.284	cMuMaxBlue	0.6	0.48	0.49
cThetaMinS	1.07	0.856	1.284	cTmOptBlue	25	25	30
cThetaNitr	1.08	0.864	1.296	cFiltMax	4.5	4.5	5.4
fDepth1Veg	0	0	0.1	hFilt	1	1	1.2
cTmInitVeg	9	7.2	10.8	cPrefDiat	0.75	0.6	0.8
cMuMaxVeg	0.2	0.18	0.19	cPrefBlue	0.125	0.125	0.15
cMuMaxDiat	2	1.6	1.8	fDAssZoo	0.35	0.35	0.42

4.2.3. Scenario design

In order to simulate the impact of climate change on Lake Mogan, a double approach was used. Firstly, temperature and precipitation were changed based on the downscaled MPI-ESM-MR **(**MPI) global climate model **(**GCM) for 2020-2090. Secondly, general climate scenarios were implemented where temperature and precipitation were transformed uniformly.

For the first approach, the projections of two representative concentration pathways **(**RCP 4.5 & RCP 8.5) in the MPI model were used as the basis for the climate scenarios in SWAT. Bias correction was applied for the temperature projection using the relationship between the observed and simulated daily average temperature in the reference period (1985-2000). For precipitation, scenarios for 2020-2090 were designed using a seasonal correction based on the difference between the future projection and the reference period. No projections for changes in wind speed, solar radiation or relative humidity were available and long-term daily average data were used instead. The projections for temperature and precipitation for both scenarios are given in Table 4.5. Temperature change in the first decades is negative due to the relatively high temperatures observed in the reference period.

Based on the projection of the MPI-RCP 4.5 and 8.5 scenarios, five temperature scenarios (A-E: -1ºC, 0 ºC, +1 ºC, +2 ºC, +3 ºC) and five precipitation scenarios (1-5: -30%, -20%, -10%, 0, +10%) were designed, resulting in 25 general scenarios when combined (A1-E5). These scenarios were used to elucidate how changes in temperature and precipitation interact to influence flow rates and nutrient loading of the streams.

Table 4.5: Temperature (a) and precipitation (b) projections per decade for 2020-2090 and relative change compared with the reference period (2000-2010) for MPI-RCP 4.5 and 8.5.

a)

	4.5 Temp (ºC)	8.5 Temp (ºC)	4.5 ΔTemp (ºC)	8.5 ΔTemp (ºC)
2020-2030	12.41	12.50	-0.44	-0.35
2030-2040	12.71	12.64	-0.15	-0.21
2040-2050	12.79	13.23	-0.06	0.38
2050-2060	13.16	13.77	0.31	0.92
2060-2070	13.18	14.39	0.33	1.54
2070-2080	13.20	14.92	0.35	2.07
2080-2090	13.27	15.39	0.42	2.54

b)

	4.5 Precipitation (mm)	8.5 Precipitation (mm)	4.5 ΔPrecipitation (%)	8.5 ΔPrecipitation (%)
2020-2030	356	323	-5	-14
2030-2040	346	355	-8	-5
2040-2050	362	354	-4	-6
2050-2060	349	337	-7	-10
2060-2070	363	276	-3	-16
2070-2080	348	294	-7	-12
2080-2090	349	294	-7	-12

For every climate scenario, the relative effect on flow rates and nutrient loading in the streams compared with the baseline scenario was determined and consequently used as input in the climate scenarios for PCLake.

For the climate scenarios, water temperature in PCLake was calculated by adding the temperature scenario to the observed air temperature and calculating the new daily water temperature based on the regression relation between both. Evaporation was changed in accordance with the change in potential evaporation as calculated by SWAT in the different scenarios. Inflow to the lake was adjusted based on the change in flow rates simulated by SWAT for each climate scenario.

The outflow of Lake Mogan is regulated and in order to estimate the lake water level as inflow levels change, a simple decision model was created that simulates control of the outflow gate. The decision model sets maximum flow rates and average ratios between inflow and outflow rates for different water levels, based on observed flow rates for 1997-2013. Regulation of the outflow is aimed at protecting the low-lying area around the lake from flooding in spring, while trying to prevent severe water level drops in summer due to evaporation. The outflow model therefore projects a near closing of the outflow gate if the lake water level drops below 3.5 m, while it would be opened stepwise to a maximum capacity of 5 m^3 s^{-1} above 4.5 m. Based on this outflow model, the relation between simulated and observed daily water levels had a bR^2 value of 0.66. This decision model was used to simulate lake outflow rates under different inflow scenarios generated by the climate scenarios in SWAT. Nutrient loading input was adjusted based on the output of the SWAT model for the respective climate scenarios.

For the general scenarios, average N, P and chl-a concentrations were determined by taking the average of all iterations and all years. For each ensemble of 250 runs in every scenario, the proportion was determined of runs that predict a macrophyte-dominated state (>30% vegetation coverage in all years), a phytoplankton dominated state (<10% vegetation coverage in all years) as well as intermediate states with macrophyte development or macrophyte loss (an increase or decrease of 15% in vegetation coverage in any given year).

For the MPI-scenarios, annual average TP, DIN and chl-a concentrations were calculated for 2020-2089 by taking the average of the 250 iterations for every time step, while excluding the 2.5% largest and smallest results. For every year, the percentage of iterations that predicted macrophyte coverage above 50% or below 5% was also determined.

Table 4.6: Monthly performance statistics for the calibrated and validated SWAT model of the Lake Mogan basin

		flow rate	NO3 load	SRP load
2000-2005	bR2	0.47	0.14	0.26
	NS	0.57	-0.08	0.00
2006-2010	bR2	0.65	0.44	0.29
	NS	-0.08	0.52	-0.01
2000-2010	bR2	0.50	0.21	0.23
	NS	0.44	0.31	0.01

4.3. Results

The relation between the simulated and observed total flow rate to Lake Mogan between 2000 and 2010 had a bR^2 of 0.50 and a NS-value of 0.44 (Table 4.6). During the calibration period (2000-2005), a reasonable good fit was obtained (Figure 4.2; Figure 4.3) with an average observed flow rate of 0.23 $m^3 s^{-1}$ and a simulated flow rate of 0.31 $m^3 s^{-1}$ (bR^2 = 0.47, NS = 0.57). During the validation period, the observed flow rates were much lower at 0.12 $m^3 s^{-1}$ and the simulated model overestimated flow rates (bR^2 = 0.65, NS =-0.08), especially in the Çölovası subbasin.

SRP and NO_3 loading to the lake was simulated reasonably well by the model (Figure 4.2). The average monthly observed SRP load was 72 kg and the model simulated 77 kg (bR2 = 0.21, NS = 0.31). Simulated monthly NO_3 loading was 739 kg compared with 846 kg NO_3 per month observed (bR^2 = 0.23, NS =0.01).

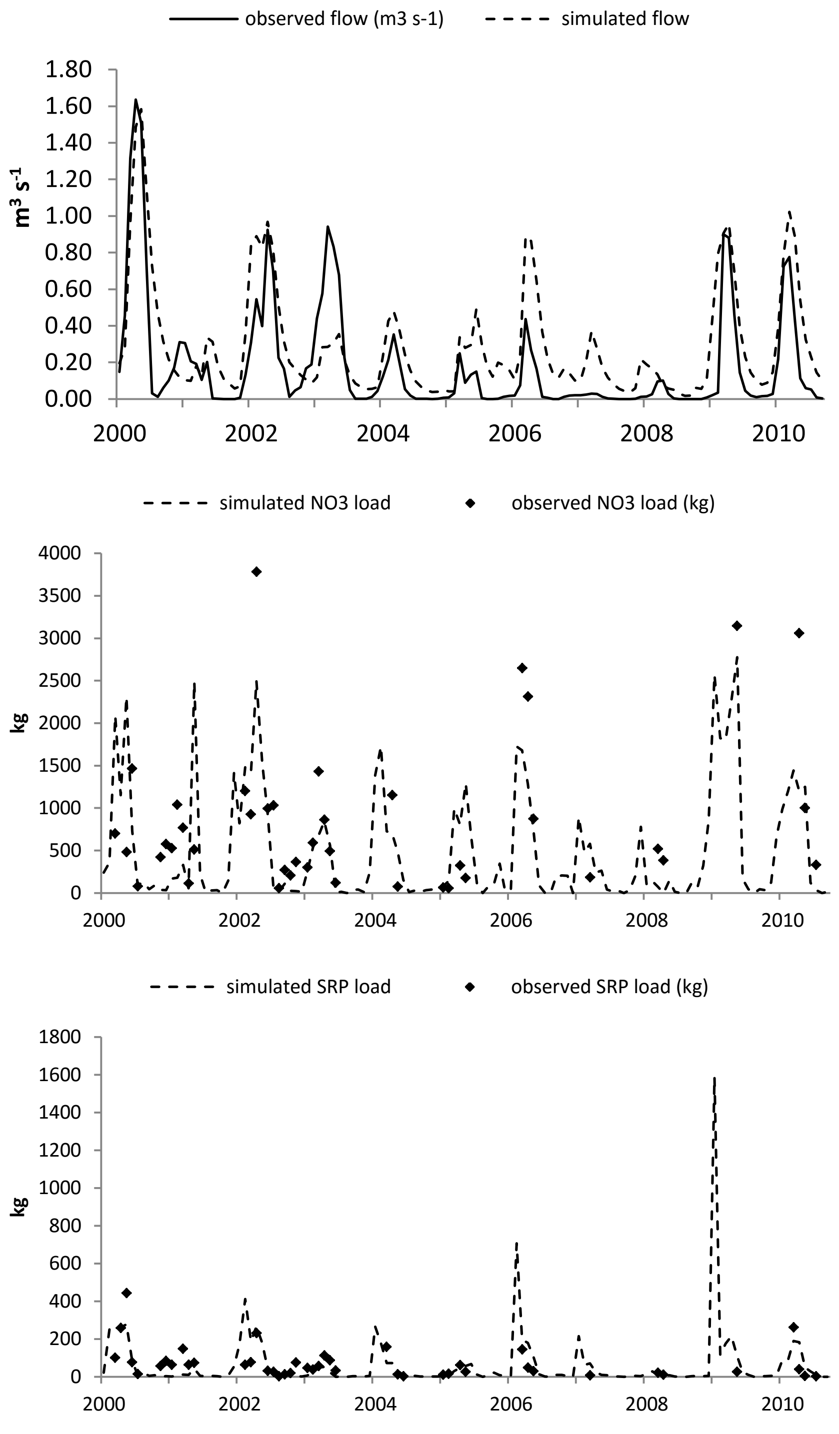


Figure 4.2: Comparison between observed and simulated monthly average flow rates, monthly NO_3 loading and monthly SRP loading of the Lake Mogan basin

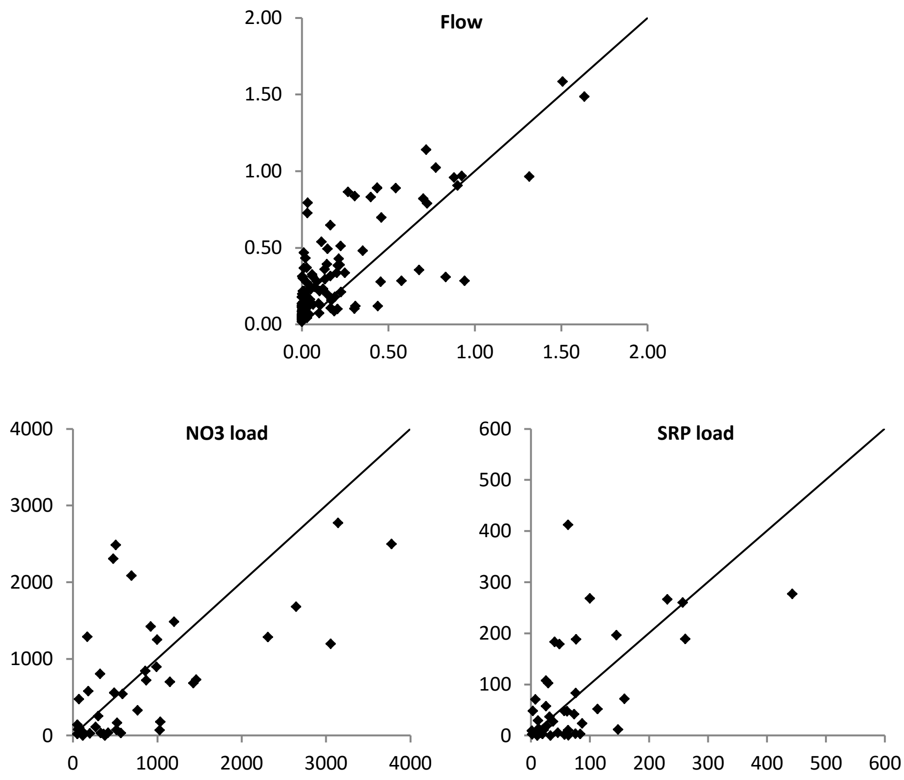


Figure 4.3: Scatterplots of the relationship between observed (X-axis) and simulated (Y-axis) flow rate, NO3 loading and SRP loading with the line representing a 1:1 relationship shown.

The calibrated model for Lake Mogan in PCLake included 90% of macrophyte coverage observations and 81% of chl-a measured values. With regard to the lake nutrient concentrations, 63% of SRP, 41% of TP, 56% of NH4 and 59% of NO_3 measurements were included in the simulation bands, respectively (Figure 4.4).

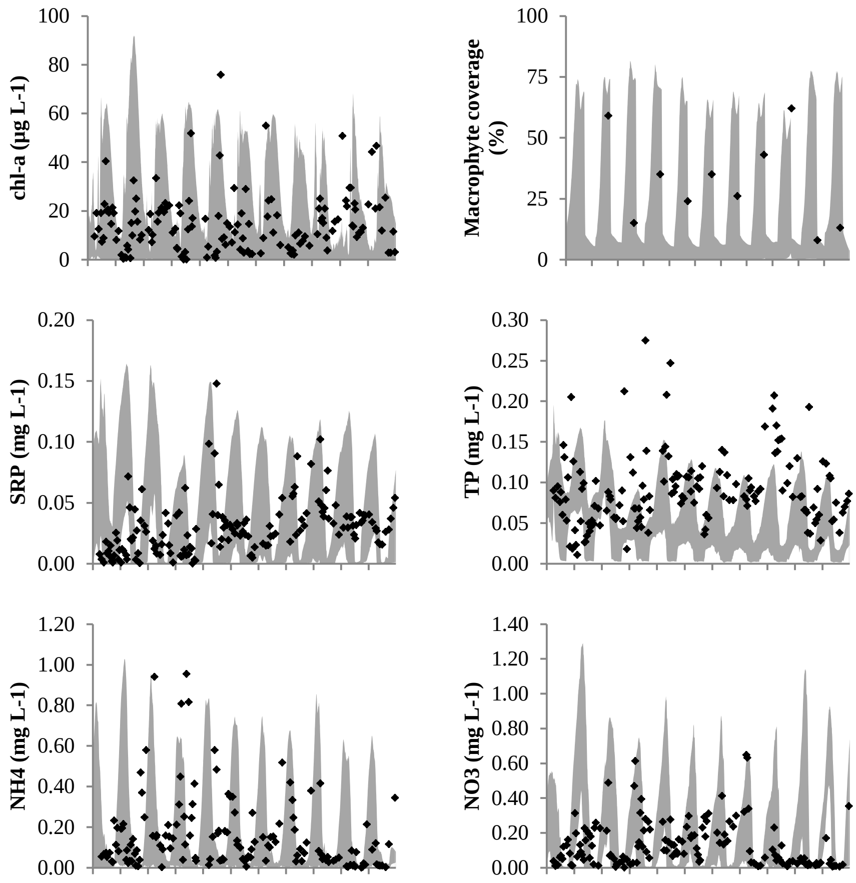


Figure 4.4: Performance of the PCLake model for Lake Mogan against observed values. Grey bands represent the range between the 2.5th percentile and the 97.5th percentile on a daily basis based on 250 iterations of the model with parameter values inside the calibration ranges.

The SWAT results of the MPI-RCP4.5 and 8.5 scenarios are given in Table 4.7. Flow rates decrease in most decades compared with the baseline period, up to -33% in the 4.5 scenario and -60% in the 8.5 scenario. NO_3 and SRP loading are expected to decline as well (-15 to -11% on average), although in certain simulated years increased NO_3 and SRP loading were found even if inflow rates declined on average, due to large rainfall events during some years leading to increased runoff and nutrient loading.

Table 4.7: Average results per decade from the SWAT-model on Lake Mogan basin for the MPI-RCP 4.5 and MPI-RCP 8.5 scenarios (2020-2090)

	4.5			8.5		
	flow rate change (%)	NO3 load change (%)	SRP load change (%)	flow rate change (%)	NO3 load change (%)	SRP load change (%)
2020-2030	-23	-22	-41	-43	-31	-63
2030-2040	-33	-24	-49	-28	-26	-38
2040-2050	-10	-7	-26	-11	-11	-15
2050-2060	-18	-21	-10	-11	+9	-1
2060-2070	+7	-3	+35	-60	-42	-66
2070-2080	-18	-5	-7	-35	-24	-31
2080-2090	-13	+2	+23	-4	+23	+114

Based on the predicted flow rates and the outflow decision model, changes in mean water level were calculated (Figure 4.5). In the MPI-RCP 4.5 scenario, periods of low water levels are predicted on several occasions with water levels below 2 m, and especially between 2030 and 2040 a prolonged drought period is simulated. The MPI-RCP 8.5 scenario predicts four severe drought periods and drying out of the lake in the second half of the century.

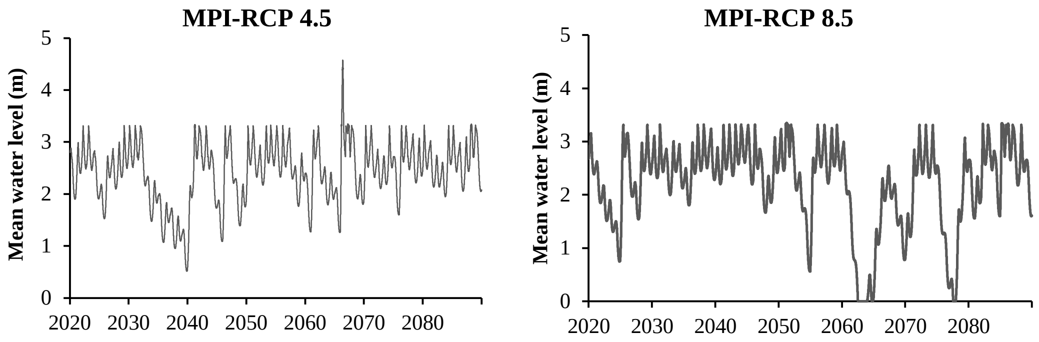


Figure 4.5: Predicted mean water levels of Lake Mogan (2020-2090) under the MPI-RCP 4.5 and 8.5 scenarios.

Results for the general climate scenarios in SWAT are shown in Figure 4.6. Average flow rates to the lake decrease with increasing temperature and decreasing precipitation to a minimum of 16% of the baseline flow in the E1 scenario. An increase in flow can be expected if precipitation increases with 10% or if mean annual temperature decreases with 1 degree. Maximum monthly flow rates changed to the same extent as average flow rates with a decrease to 17 % of the baseline scenario in the E1 scenario. NO_3 loading is mostly determined by runoff and is therefore more linked with changes in precipitation than with changes in temperature. All precipitation decrease scenarios result in a decrease in NO_3 loading to around 45% in case of a 30% decrease in precipitation. SRP loading decreases in almost all scenarios to a minimum of 8% of the baseline loading in the E1 scenario.

The results of the MPI scenarios in PCLake are shown in Figure 4.7 and Figure 4.8. Results for the MPI-RCP8.5 scenarios are only given until 2060 before the scenario predicts the lake to dry out.

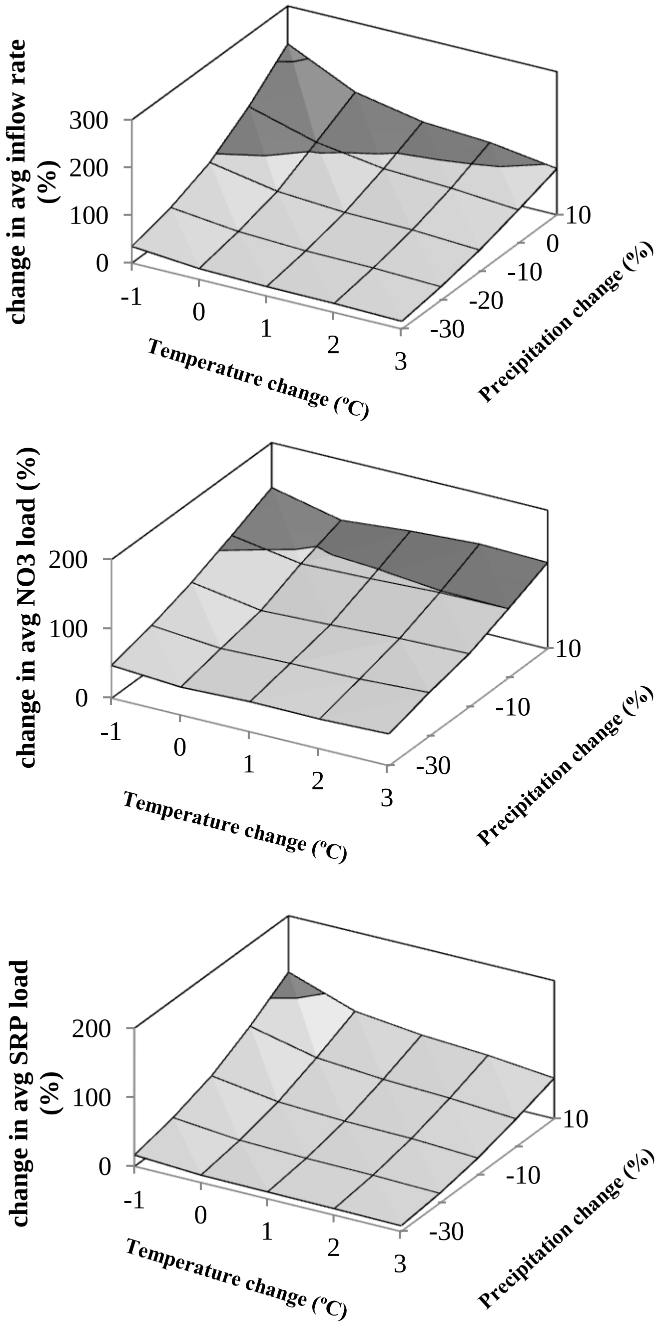


Figure 4.6: SWAT scenario results for relative change in average inflow rate, annual NO_3 load and annual SRP load. Dark grey shading represents simulated values larger than the baseline model, while light grey shading shows simulations with values smaller than the baseline model.

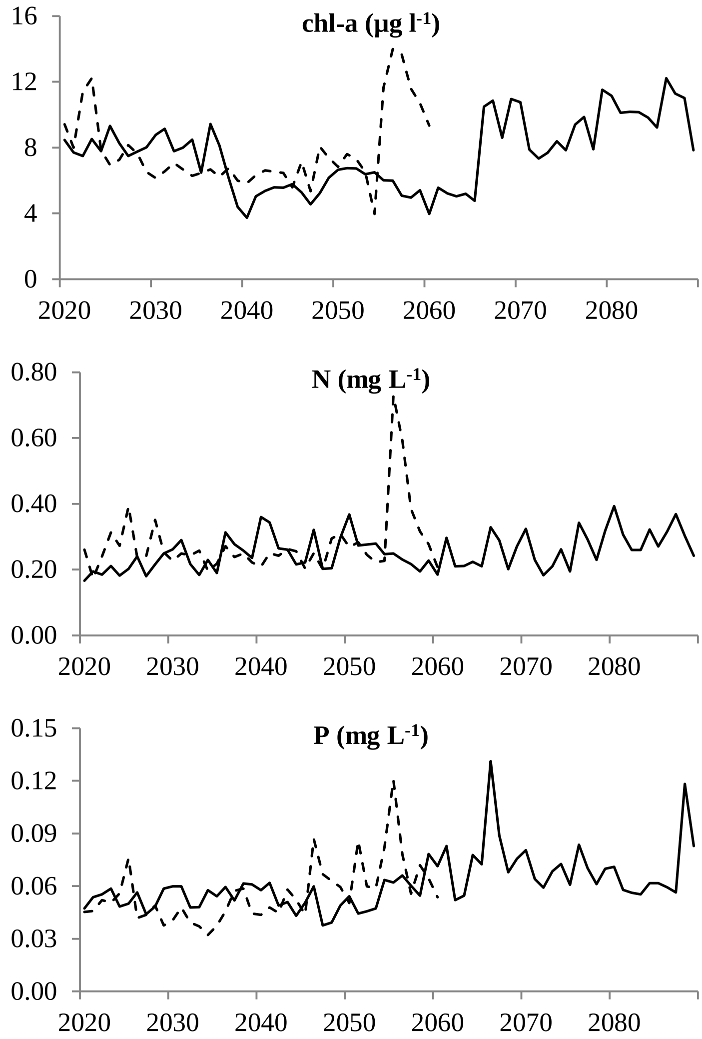
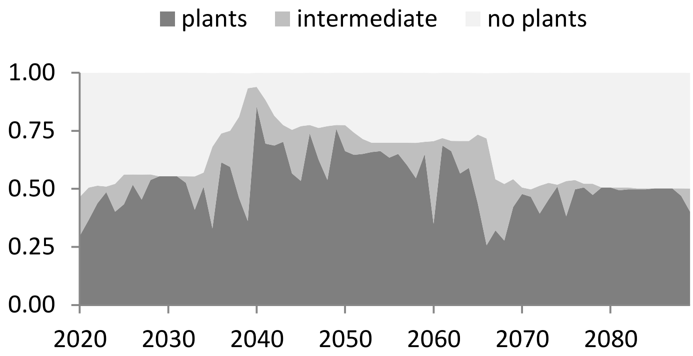


Figure 4.7: Predicted chl-a, phosphorus (P) and nitrogen (N) concentrations in the PCLake model of Lake Mogan for the MPI-RCP4.5 (full line) and 8.5 (dashed line) scenarios (2020-2090).

The results of the MPI scenarios in PCLake are shown in Figure 4.7 and Figure 4.8. Results for the MPI-RCP8.5 scenarios are only given until 2060 before the scenario predicts the lake to dry out.

a)



b)

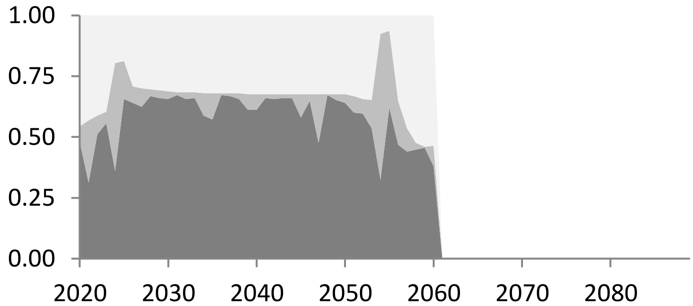


Figure 4.8: Percentage of iterations predicting plants (>50% macrophyte coverage) or no plants (<5% macrophyte coverage) for the a) MPI-RCP4.5 and b) 8.5 scenarios (2020-2089)

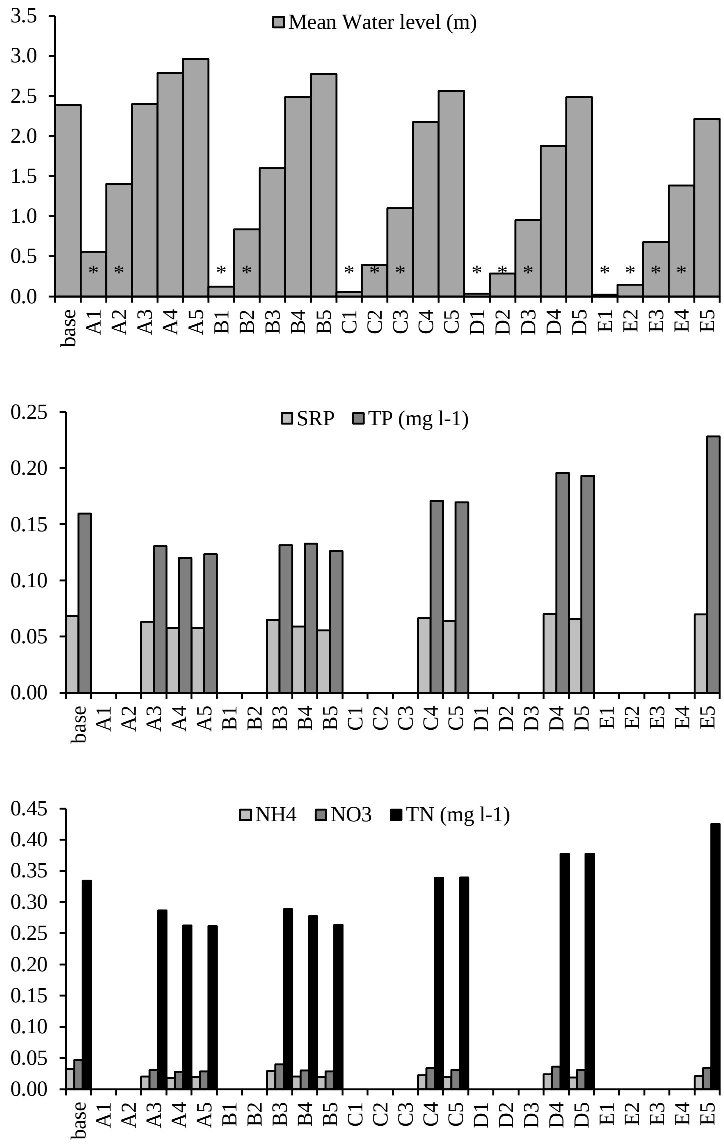


Figure 4.9: Mean water level, N and P concentrations for the baseline simulation and the general scenarios (* marks scenarios predicting lake dry out at some points during the 10-year simulation)

Figure 4.9 shows the changes in mean water level in the general scenarios. The results show decreasing water level with increasing temperature (A-E) and decreasing precipitation (5-1) and drying out of the lake in many scenarios. Both in-lake TN and TP increase with increasing temperature (and thus decreasing water level), while differences between the precipitation scenarios are small. The proportion of runs predicting a macrophyte-dominated state did not change much between the general scenarios (Figure 4.10).

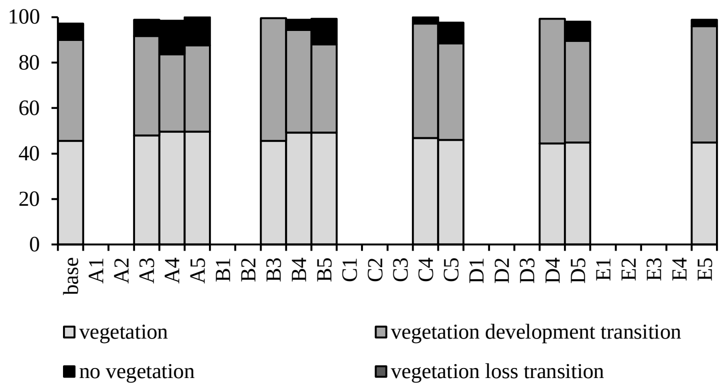


Figure 4.10: Vegetation development for the baseline simulation and the general scenarios.

4.4. Discussion

We used a combined approach of catchment modelling and lake modelling to study the effects of changes in temperature and precipitation, as expected by projected climate change, on a shallow lake in a semi-arid climate. Our results show that flow rates into the lake can decline up to 60%, with an average decline of 15-27%, depending on the RCP projection, leading to dry periods with low water levels and even potential drying out of the lake. NO_3 and SRP loading are expected to decline as well due to reduced runoff and diminished groundwater flow. General scenarios showed that inflow rates decrease with both temperature and precipitation, while the NO_3 and SRP load responded mostly to changes in precipitation.

The results of the lake model exhibited similar predictions in N, P and chl-a concentration in both RCP scenarios. Sharp increases in nutrient concentrations are predicted when water levels drop, unless nutrient loading decreases as well. Higher precipitation and higher nutrient loading also augment lake nutrient concentrations and can lead to a decrease in plant coverage and an increase in chl-a. Prolonged periods of low water levels seem to encourage macrophyte development. The results of the general scenarios confirm the risk of drying out of the lake with changes in precipitation and temperature and indicate higher TP and TN with increasing temperature. No strong effect on submerged macrophyte development was observed in the general scenarios.

The calibration performed in SWAT was considered acceptable based on the review made by Moriasi et al. (2007), with a NS-estimate near 0.50 for the entire 2000-2010 period. A better calibration fit was hindered by a combination of factors, mainly related to the Çölovası subbasin. The Çölovası stream connects with the lake through a wetland area. Water loss through evaporation from this wetland area was underestimated by SWAT, which led to overestimation of flow rates especially during the dry years. The two reservoirs in the Çölovası subbasin also had an unknown water release regime during the calibration period, complicating modelling of the flow contributions from large parts of the subbasin. During winter periods with high precipitation, the lake also extended through the wetland area up to the Çölovası monitoring point, causing low measured flow rates, a phenomenon which could not be incorporated into the SWAT-model. For this reasons, the calibration was first focussed on the Yavrucak and Sukesen subbasin to fix the basin-wide parameters and later applied to the Çölovası subbasin to obtain the best fit for total inflow to the lake.

The calibration for the catchment with SWAT resulted in one set of values for every parameter for the calibrated subbasins. This deterministic approach can fail to take into account uncertainties related with the available data and knowledge on the characteristics of the catchment and this uncertainty should be taken into account when interpreting the results of the model.

The results of the catchment model predict lower flow rates in the stream, lower runoff and lower nutrient loading, which is in line with the results of other studies undertaken in the Mediterranean climate zone (Nunes et al., 2008; Molina-Navarro et al., 2014). The large reductions or increases in flow rates with modest changes in precipitation are in line with observations from the 1997-2010 reference period, with, for example a 25% precipitation decrease from 2006 to 2007, leading to a 80% reduction in the inflow to the lake. Lower nutrient loading during years with lower hydraulic loading is also supported by observations between 1997 and 2010, due to lower runoff and reduced groundwater flow to the lake.

Lower nitrate loading in the climate scenarios with lower precipitation seemed to be associated with lower runoff, which corresponds with the results of Panagopoulos et al. (2011). On the other hand, inflow nitrate concentrations increased because inflow volumes decreased to a larger extent. SRP concentrations did not vary much between the different scenarios and SRP loading seemed to be more associated with lateral groundwater flow than with surface runoff. The MPI-RCP projections show an enhanced nutrient loading in certain decades, even when average flow rates do not increase, caused by more extreme rainfall events that lead to higher runoff and nutrient loading.

The scenarios did not include any changes to any other primary meteorological parameters except temperature and precipitation. Land use changes were also not incorporated into this modelling study. While this study focuses on providing insight into the effects of changes in climate, changes in land use in the next 70 years will have large effects on the hydraulic and nutrient loading in the catchment and can both mitigate or enhance the effects of changes in climate. Management practices can be adopted to reduce the nutrient loading from the catchment, although it can be remarked that agriculture in the catchment is not intensive and fertilizer use is already low.

In contrast to the deterministic calibration of the SWAT model, a more stochastic approach was used for PCLake, following Nielsen et al. (2014). The main reason was the difficulty in modelling the fast year-to-year changes in submerged macrophyte coverage in Lake Mogan with PCLake. The model was originally developed to model the dynamics of alternative stable states in northern temperate lakes and focuses on nutrients and light availability as the primary determining factors (Janse & Van Liere, 1995). In shallow semi-arid lakes, water level fluctuations also play an important role in determining the macrophyte occurrence (Havens et al., 2004; Beklioğlu et al., 2006), both directly through the changes in area available for submerged plant growth at different water levels and indirectly through the effect on nutrient concentrations in the water column. PCLake in its current form reduces the lake to a 0D water column and determines plant biomass for one standard square meter before converting it to an estimate of total plant coverage in the lake. In a V-shaped lake, like Lake Mogan, the area available for plant coverage fluctuates relative to light penetration and water depth and the current 0D approach of PCLake cannot represent these processes. Therefore we did not aim to model the yearly changes in macrophyte coverage exactly, but rather to create a simulation band including both runs with and without macrophytes. The new FABM approach to PCLake, allowing taking into account three dimensional differences in the lake into account, might remedy this problem in the future (Hu et al., 2016).

The calibration of the PCLake model was moderately successful. The model was able to incorporate many of the observed values, but the simulation band was wide even though parameters did not vary more than 20% from the default value. This indicates that a few parameters, mostly related to macrophyte and phytoplankton growth, can have a very large effect on the modelled nutrient concentrations.

The food web structure in PCLake is focussed on the food web structure in northern temperate lakes, and while the effect of macrophyte biomass on zooplankton predation (through the refuge effect) is incorporated in the model, changing the relevant parameters did not have a significant effect on the model outcome; nor did it improve calibration of chl-a. This can be because the refuge effect of macrophytes was not important for zooplankton abundance in Lake Mogan, since Tavşanoğlu et al. (2012) showed that the sediment is a more important refuge in Mediterranean lakes, or that the top-down effects on zooplankton in PCLake are underestimated.

The peaks in chl-a, P and N in Lake Mogan projected by the MPI-RCP4.5 and 8.5 scenarios are caused by periods of low water levels, with higher internal loading and lower plant biomass as well as periods with high water levels and high external loading. The general scenario results observed from the PCLake model can be explained by the increase in phytoplankton biomass, and particularly cyanobacteria at higher temperature (Paerl & Huisman, 2008), leading to higher biomass and higher organic N and P.

Salinity of the lake is not included in PCLake and the effect of high salinity that can be expected on submerged macrophytes, zooplankton and phytoplankton communities due to increased evaporation and increased hydraulic residence time, as observed in Lake Mogan in the dry period (2004-2008), could not be incorporated in the modelling exercise.

4.5. Conclusion

Our combined approach of catchment modelling and lake modelling demonstrate the threats to shallow lakes in the Mediterranean semi-arid climate region. Reduced precipitation in combination with higher evaporation creates lower inflows to the lake, higher water loss and an increased risk of low water levels and drying out. Due to reduced runoff, nutrient loading is expected to decrease, although extreme events with high precipitation may lead to enhanced external loading to the lake. The effects on the lake are increases in nutrient concentrations during periods with low water levels and high external loading and increased phytoplankton biomass due to higher temperature. Further research including salinity, spatial development of macrophytes and changes in the food web is necessary to be able to undertake further model projections on the impact of climate change on warm shallow lakes.

CHAPTER V

CONCLUSIONS

The aim of this research was to study the relationship between climate, hydrology and the nutrient dynamics in shallow lakes in the Mediterranean, semi-arid climate zone. For this purpose we used three different approaches to test hypotheses related with external nutrient loading, internal nutrient loading, nutrient retention or loss and nutrient concentrations in lakes and how these nutrient processes are influenced by changes in hydrology and the relationship to the climate and climate change. The combined results of a space-for-time mesocosm experiment, analysis of a 20 year time series of two shallow lakes and a combined catchment and lake modelling approach allowed us to determine how changes in precipitation and temperature affect hydraulic and nutrient loading to the lakes and how it influences internal lake processes.

The results of the climate scenarios in the SWAT catchment model of Lake Mogan showed that relatively small decreases in precipitation (-30%) can lead to drastically lower inflow volumes to the lake, leading to low water levels that can persist for many years or can even lead to the potential drying out of the lake. The long term water budgets of Lakes Eymir and Mogan confirmed this and showed that inflows can decrease to a large extent in drier years and, given that annual evaporation accounts for nearly half of the average lakes' volume, low water levels cannot be avoided even with outflow control.

Due to their relatively small volumes, shallow lakes in the semi-arid climate zone are very sensitive to changes in inflow volumes and natural fluctuations in hydraulic loading and water level are part of their dynamics. Due to this sensitivity, man-made changes in the catchment can cause a big threat to the lakes. Irrigation agriculture in the Eymir-Mogan catchment is limited, but two reservoirs are already present and with climate warming an increase in crop irrigation may be expected. Planned and realised urbanisation around the lake shore of Lake Mogan, in the Imrahor valley and in low-lying parts of Ankara downstream of Lake Eymir puts further constraints on water level fluctuations in the lakes. To protect the low lying built-up areas, water level increases during winter and spring are limited by increasing the potential outflow volume. This limits the hydrological buffer capacity of the lakes to deal with the water loss during summer or with prolonged drought periods. An additional flood gate is also under construction by DSI just upstream of the Çölovası monitoring point to further protect the low-lying urban areas, but any reduction in inflow volumes during the critical winter and spring period would have large effects both on the Çölovası wetland area and the lakes. Management of this flood gate should therefore aim to minimize evaporative water loss and to release the retained water as soon as possible after the storm peak.

Low water levels are also not just a hydrological problem and have direct effects on nutrient concentrations and potential eutrophication problems in shallow lakes. When inflow volumes decrease as a result of lower precipitation, external nutrient loading decreases as well due to lower runoff and lower base flow. But nutrient concentrations can still increase under these conditions. Internal loading of nutrients, mostly P, stored in the sediment increases under low water level conditions, especially when large nutrient stocks are present as the result of high historic nutrient loading, as in the case of Lake Eymir. External nutrient also still plays a role and if absolute nutrient loading to the lake (in kg) decreases to a lesser extent than hydraulic loading the result will be low inflow volumes, with higher nutrient concentrations leading to nutrient enrichment of the lake. Lake modelling confirmed this process and showed that lake nutrient concentrations can increase during dry periods as a result of internal processes or as a result of external loading with high nutrient concentrations.

The results of the pan-European mesocosm experiment showed that external loading is directly linked with the extent of nutrient retention or nutrient loss in shallow lakes. The long-term nutrient budgets of Lake Mogan on the other hand showed that nutrient retention was higher in years with low external loading and low water levels, while internal loading occurred in Lake Eymir during the low water level years. This was not observed in the mesocosm experiment since the water level effect could not be quantified due to use of different nutrient loading levels in the shallow and deep mesocosms. The lakes nutrient budgets seem to show though that water level can have a strong effect on retention quantities independent from external loading. Low water levels allow for a relatively larger contact between the sediment and the water column, potentially enhancing retention. Periods of the low water levels in the lakes are also associated with very long water residence times, due to closing of the outflow gates, further enhancing retention processes.

Although the analysis of the data from the mesocosm experiments did not allow quantifying the different processes of retention (sediment uptake, biomass uptake, denitrification), analysis of retention in relation to development of macrophyte and phytoplankton biomass seems to indicate a large effect of biomass in these closed mesocosm ecosystems. In shallow mesocosms, higher temperatures lead to lower water levels which benefitted development of macrophyte which took up larger quantities of P and N from the biomass. In deep mesocosms higher temperatures lead to higher phytoplankton biomass and higher organic N and P production as a result leading to lower total retention.

Water levels, submerged macrophytes and nutrient concentrations are in a complex relationship with many interactions. Lower water levels lead to higher nutrient concentrations due to evaporative water loss concentrating more nutrients in a smaller water volume and due to higher internal loading. Higher nutrient concentrations generally benefit phytoplankton because more nutrients are available for uptake in the water column giving them a competitive advantage over macrophytes, leading to a shift to the turbid-water phytoplankton dominated eutrophic state. But the stability of this turbid water state depends in part on the capacity of phytoplankton to reduce the light availability and shade submerged macrophytes. Low water levels increase the light availability at the bottom of the lake and increase the percentage of the lake area with suitable light conditions for macrophyte growth. In a further feedback mechanism, macrophytes can create conditions favourable for nutrient retention and thus lower nutrient concentrations in the water column by direct uptake from nutrient of the water column and by providing favourable conditions for denitrifying bacteria.

The final result of low water levels on nutrient concentrations therefore depends on many different factors and while the mesocosm experiment showed a positive effect on macrophyte development and retention (at least in the shallow mesocosms) the field observation in Lake Eymir and Lake Mogan are not so straightforward. More work on the different processes of retention and the role of macrophytes and low water levels is therefore required.

The balance between phytoplankton and macrophyte may also be further disturbed by increasing temperatures as expected by climate change projections. Lake modelling showed that higher temperature promotes phytoplankton growth although the effect on potential macrophyte development did not seem large. Nonetheless, warmer temperatures are expected to increase the risk of eutrophication even if lower water levels potentially enhance macrophyte growth.

Overall the research showed the importance of changes in the hydrological processes on the nutrient dynamics in Mediterranean lakes and the relation with changes in precipitation and temperature. A future drier and warmer climate with more human pressure on shallow lakes will complicate lake management efforts to keep shallow lakes in a desirable low nutrient state.

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APPENDIX A

MODELLING OF THE HYDROLOGICAL CYCLE IN SWAT

An overview of the hydrological cycle as modelled by SWAT is presented in the Figure A.1.

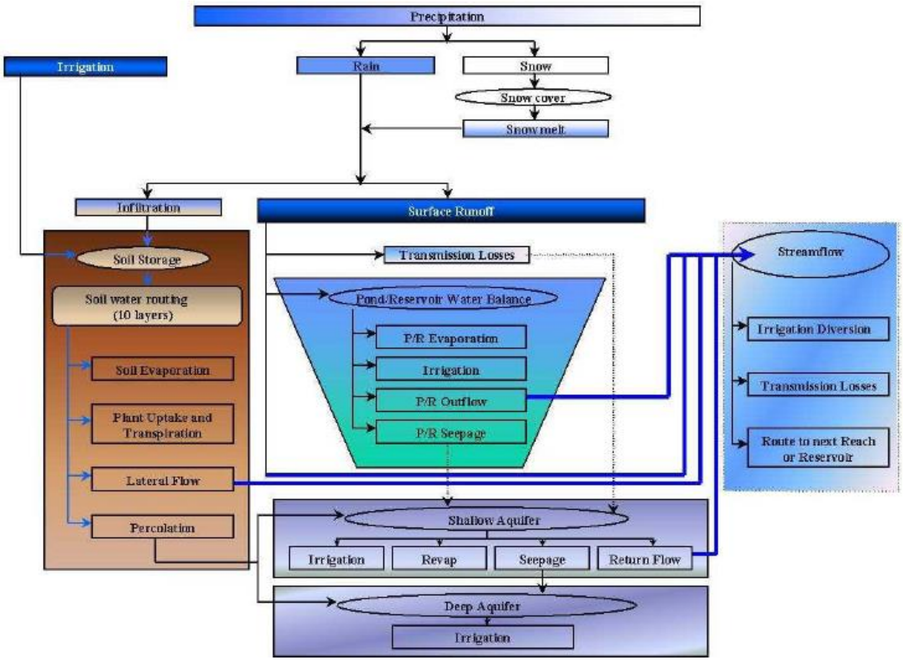

Figure A.1: Hydrological cycle in SWAT (from Neitsch et al., 2011).

The different components of the water cycle in SWAT are:

- Canopy storage: the amount of precipitation that is intercepted and made available for evaporation by canopy. It is either taken into account in the Curve Number for the surface runoff calculation or it is calculated separately by considering the Leaf Area Index of the vegetation type and the maximum storage capacity of the canopy.
- Infiltration: the uptake of water by the soil. The infiltration rate is larger when the soil moisture content of the soil is low and equal to the hydraulic conductivity of the soil and full saturation. In the Curve number calculation method infiltration is not modelled directly but calculated as the difference between precipitation and surface runoff. In the the Green & Ampt calculation method infiltration can be modelled if precipitation data on a sub-daily basis are made available.
- Redistribution: refers to the movement of water in the soil caused by differences in moisture content in different parts of the soil profile. Percolation to lower layers is simulated if the upper soil layer is at field capacity and the lower layer is not saturated. The percolation rate is determined by the hydraulic conductivity of the soil layer and by soil temperature.
- Evapotranspiration: the sum of water loss through evaporation from open water surface, through transpiration from plant leaves and sublimation from ice and snow. SWAT contains three methods to calculate potential evapotranspiration (Hargreaves, Priestley-Taylor and Penman-Monteith). Potential evapotranspiration is then used to calculate soil evaporation and plant transpiration by using the leaf area index, soil depth and water content.

- Lateral subsurface flow: the flow of water below the surface but above the saturated groundwater table. A kinematic model is used to calculate subsurface flow from every soil layer simultaneously with the redistribution based on hydraulic conductivity, water content and slope.
- Surface runoff: the overland flow phase of the water cycle. SWAT calculates surface runoff volumes and peak runoff rates for every HRU in the basin by using the Curve Number method or the Green & Ampt method.
- Tributary channels: after the water is routed to the channel it drains to the basin outflow. Secondary channels rout water within a subbasin to main channel which transports the water to the subbasin outflow. Main channels also receive groundwater return flow. Transmission losses in the channel bed are simulated based on channel width, length and flow duration.
- Return flow: the base flow is the groundwater contribution to the stream flow. The groundwater in SWAT is divided into a shallow aquifer, which contributes to return flow to streams within the catchment and a deep aquifer, which does not. Water in the shallow aquifer may also replenish the soil water content in upper soil layers under certain conditions.

For further information on the equations used in SWAT to model the described processes we refer to the SWAT-manual (Neitsch et al., 2011). An overview of the water cycle for the Lake Mogan basin as modelled in SWAT is presented in Figure A.2.

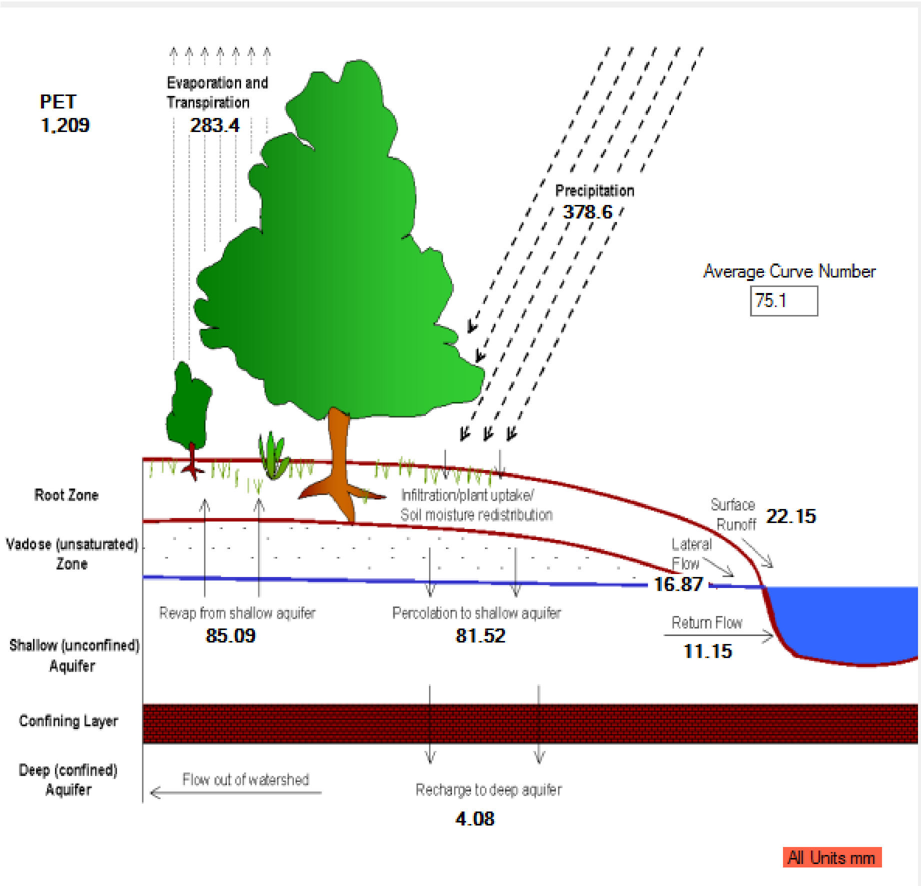

Figure A.2: Hydrological cycle for Lake Mogan in SWAT

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PUBLICATIONS

1. **Coppens, J.**, Özen, A., Tavşanoğlu, Ü.N., Erdoğan, Ş., Levi, E.E.,Yozgatlıgil, C., Jeppesen, E. & Beklioğlu, M., 2016. Impact of alternating wet and dry periods on the long-term seasonal phosphorus and nitrogen budgets of two shallow Mediterranean lakes. Science of the Total Environment 563–564: 456–467.

2. **Coppens, J.,** Hejzlar, J., Šorf, M., Jeppesen, E., Erdoğan, Ş., Scharfenbergerg, U., Mahdy, A., Nõges, P., Tuvikene, A., Baho, D.L., Trigal, C., Papastergiadou, E., Stefanidis, K., Olsen, S., & Beklioğlu, M., Hydrobiologia, in press. The influence of nutrient loading, water level and climate on nitrogen and phosphorus retention in lakes: a pan-European mesocosm experiment. DOI: 10.1007/s10750-015-2505-9

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6. Özen, A., Bucak, T., Tavşanoğlu, Ü.N., Çakıroğlu, A.I. , Levi, E.E., **Coppens, J.** Jeppesen, E. & Beklioğlu, M., 2014.Water level and fish-mediated cascading effects on the microbial community in eutrophic warm shallow lakes: a mesocosm experiment. Hydrobiologia 740: 25-35.

7. Van Orshoven, J., **Coppens J.** and Terres, J.M., 2007. Possibilities and limitations of AVSWAT2000 for the assessment of environmental impacts of farming practices. Proceedings of the 4th International SWAT conference, UNESCO-IHE, Delft, the Netherlands, 4-6-july-2007: 24-31.

CONFERENCE PRESENTATIONS

1. **Coppens, J.,** Bucak, T., Tavşanoğlu, Ü.N., Trolle, D. & Beklioğlu, M. Impact of climate change on the hydrological and nutrient balance of a shallow Mediterranean lake using catchment and lake modelling approaches. 2015 ASLO Aquatic Science Meeting, Spain. 2015

2. **Coppens, J.,** Erdoğan, Ş., Bucak, T., Özen, A., Jeppesen, E. & Beklioğlu, M. Impacts of wet and drought periods on the long-term seasonal phosphorus and nitrogen budget of two shallow Mediterranean lakes. 8th Shallow Lakes Conference, Turkey. 2014

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