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Is Anthropogenic Nutrient Input Jeopardizing Unique Lake Ohrid? - Mass Flux Analysis and Management Consequences

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Abstract

Lake Ohrid is comparably large (surface area A ~358 km^2), deep (maximal depth z_{max} ~289 m) and one of the most voluminous (volume V ~55 km^3) lakes in Europe. It is a transboundary lake, with two-thirds in Macedonia and one-third in Albania. Including the underground karst connection to upstream Lake Prespa (A ~254 km^2, z_{max} ~48 m, V ~3.6 km^3), the watershed of Lake Ohrid extends into Greece. The overall karst groundwater inflow – recharged by Lake Prespa and local precipitation – contributes more than 50 % to the lake's water balance. Because of the comparably small watershed and the dry climate, water exchange is slow with a mean residence time of ~70 years.

While a groundwater-fed, large water body in Mediterranean climate is exceptional in itself, the most outstanding quality of Lake Ohrid is its endemism (> 200 species). During the estimated two to five million years of existence, species covering the whole food chain have evolved or persisted in the lake. Given its importance as a global hotspot of biodiversity and being the only ancient, long-lived lake in Europe, Lake Ohrid was declared a UNESCO world heritage site in 1979.

However, the unique ecosystem could be jeopardized by human impacts. A particular concern is potential eutrophication of the currently oligotrophic lake from population growth. It is feared that eutrophication could threaten the endemic flora and fauna, which has adapted to nutrient-poor conditions. However, assessment of eutrophication is not straight forward: Because of the long water residence time, changes will be slow. Moreover, expected variations in nutrient conditions are in the range of measurement error because of the low present-day concentrations. As there is a lack in high-quality, long-term data, different measurement schemes must be combined to assess eutrophication.

During this work, a basic monitoring program was established, covering the lake, its inflows, as well as settling particles and accumulated sediments. Moreover, available historic information was collected and evaluated. Indeed, slow eutrophication could be detected in sediment cores over the past century. Combining all information in a linear model shows a ~3.5 fold increase in phosphorus (P) concentration from 1.3 mg-P m^{-3} to currently 4.6 mg-P m^{-3}. As was expected, the mean P residence time is comparably high at ~5 years, signifying that it takes around 15 years until a new P equilibrium is reached after an increase in input. Major P-input was identified from domestic point sources, which enter the lake diffusively and via tributaries. Agricultural sources seem to be of secondary importance at the moment but may potentially increase in the future if artificial fertilizers become affordable to local farmers.

Compared to the P inputs from tributaries and diffusive sources, the groundwater inflows – though being the dominant water source – contribute only ~9 % to the P budget. However there is an eutrophication potential in the connection to upstream, mesotrophic Lake Prespa, which has ~7 times higher P concentrations than Lake Ohrid. The quantification of the Lake Prespa water share in accessible groundwater inflows

was possible thanks to a distinct signal in natural tracers, such as Cl^- and stable isotopes. Combining results from different spring locations indicates that most of the water from Lake Prespa flows into Lake Ohrid, contributing ~20 % to its water balance. However 65 % of P is held back during the underground passage by filtering and potentially precipitation processes. Thanks to these natural cleaning mechanisms only ~6 % of the total P input to Lake Ohrid stems from Lake Prespa. It was shown that this P input increases with P-load to Lake Prespa but also with decreasing water level from water abstraction. A worst case scenario of a future four-fold increase could have a negative effect on Lake Ohrid leading to a 1 mg-P m^{-3} (20 %) P-increase. While being a potential future threat to Lake Ohrid, Lake Prespa itself is substantially endangered by ongoing eutrophication and a water level decrease.

Having quantified the current P budget the sensitivity of Lake Ohrid to eutrophication in comparison with other human impacts – such as water abstraction and global warming – is important. As a result we assessed the effects of – (a) doubling of current P-concentrations, (b) extended use of water for irrigation during summer season and (c) predicted 0.04 °C yr^{-1} atmospheric warming – on the physical lake properties. As expected, eutrophication has been identified as a major threat. On the one hand, it would support growth of non-endemic plankton forms close to the surface. As a result, light conditions would worsen for endemic diatom Cyclotella fottii, which is currently dominant between 30 and 150 m depth. On the other hand, higher productivity will increase mineralization and DO consumption at the sediment. This could even lead to anoxia, particularly in the deep, non-regularly mixed hypolimnion, which would have serious impacts on the unique bottom fauna of Lake Ohrid. While the prevention of eutrophication must be the emphasis of lake management, the negative effects would be significantly amplified by global warming. A warming in air temperatures of 0.04 °C yr^{-1} would reduce the probability of deep convective mixing events and stabilize the water column, basically leading to an isolation of the deep water in the long term. This increase in stratification is due mainly to the deep water temperatures, which lag behind the warming of the surface water. Thus it is the rate of warming that reduces lake mixing and not higher temperature. In contrast, deep convective mixing would be even more frequent than today under a higher temperature equilibrium, as a result of the temperature dependence of the thermal expansivity of water. Although water abstraction may change local habitats, e.g., karst spring areas, its effect on overall lake properties was shown to be of minor importance.

Based on the sensitivity analysis, management steps to prevent eutrophication must be adapted to the extent of global warming. In order to evaluate the individual and combined effects of global warming and eutrophication, a set of scenarios was defined and simulated via the coupling of a physical and a bio-geochemical lake model. It was found that at least a 50 % reduction in P input must be reached to keep DO > 4 mg L^{-1} in the deep hypolimnion for the next 50 years at predicted 0.04 °C yr^{-1} atmospheric warming. However, as has been shown in the sensitivity analysis, the deep hypolimnion will turn increasingly isolated, notwithstanding the P reduction. As a result, global warming would have to be slowed down within the next five decades to prevent anoxia in Lake Ohrid.

The presented analysis was only possible in close cooperation with the Macedonian Hydrobiological Institute (HBI). Apart from the joint monitoring work, a program for scientific exchange was established to support the use of new analytical infrastructure at HBI and a regular future monitoring of Lake Ohrid. Project performance was successful in (i) assisting the regular and appropriate use of sampling equipment for lake monitoring, (ii) training young scientists on high-tech equipment through internships abroad and (iii) supporting international cooperation with HBI. However, the project did not influence national relations of HBI. In an overall assessment, the combination of a PhD with the responsibility for the organization of a small project in an international setting is highly suggested.

Kurzfassung

Der Ohridsee ist ein grosser, tiefer und einer der volumenreichsten Seen Europas (Oberfläche A ~358 km^2, maximale Tiefe z_{max} ~289 m, Volumen V~55 km^3). Die Oberfläche des Grenzsees teilen sich die Länder Mazedonien und Albanien im Verhältnis zwei zu eins. Durch die Karstverbindung zum höher gelegenen Prespasee (A ~254 km^2, z_{max} ~48 m, V ~3.6 km^3) erstreckt sich das Einzugsgebiet zusätzlich bis nach Griechenland. Die vom Prespasee und lokalem Niederschlag gespeisten Karst-Grundwasserzuflüsse bilden auch den grössten Beitrag zur Wasserbilanz des Ohridsees (> 50%). Aufgrund des relativ kleinen Einzugsgebietes, sowie des trockenen, mediterranen Klimas ist der Gesamtzufluss allerdings gering, was zu einer langen Wasseraufenthaltszeit von durchschnittlich ca. 70 Jahren führt.

Ein grundwassergespeister See dieser Grössenordnung im trockenen südeuropäischen Klima ist an sich schon erstaunlich. Der herausragendste Aspekt des Ohridsees ist aber seine endemische Flora und Fauna (> 200 Arten). Während seiner geschätzten zwei bis fünf Millionen jährigen Geschichte haben Tier- und Pflanzenarten der gesamten Nahrungskette im Ohridsee überdauert und sich weiterentwickelt. Die Bedeutung des Ohridsees als einziger Ursee Europas und globales Biodiversitäts-Zentrum wurde 1979 durch seine Aufnahme als Weltnaturerbe der UNESCO unterstrichen.

Das einzigartige Ökosystem ist allerdings verschiedenen menschlichen Einflüssen ausgesetzt. Insbesondere wird als Resultat des regionalen Bevölkerungswachstums eine Eutrophierung des momentan oligotrophen Sees befürchtet. Diese könnte ihrerseits die endemischen Arten gefährden, welche sich im Laufe ihrer Entwicklung an nährstoffarme Bedingungen angepasst haben. Die Frage, ob der See sich in einem Eutrophierungsprozess befindet ist allerdings nicht einfach zu beantworten. Einerseits bewirkt die lange Wasseraufenthaltszeit eine verzögerte Reaktion auf Veränderungen. Andererseits liegen erwartete Konzentrationsschwankungen im nährstoffarmen See im Bereich des analytischen Messfehlers. Wegen einem Mangel an langfristigen, qualitativ hochwertigen Seemonitoring-Daten müssen daher verschiedene Messungen kombiniert werden, um etwas über eine potentielle Eutrophierung aussagen zu können.

Im Rahmen dieser Arbeit wurde deshalb ein Messprogramm durchgeführt, welches die Wassersäule des Sees, seine Zuflüsse, sowie absinkende Partikel und akkumuliertes Sediment umfasst. Zusätzlich wurden vorhandene historische Daten gesammelt und in die Auswertung miteinbezogen. Tatsächlich konnte eine langsame Eutrophierung während der letzten hundert Jahre in Sedimentkernen festgestellt werden. Eine Kombination aller Informationen in einem linearen Modell zeigt einen 3.5-fachen Anstieg der Phosphor (P)-Konzentration im See von ~1.3 mg-P m^{-3} zu aktuellen ~4.6 mg-P m^{-3}. Wie erwartet ist die mittlere P-Aufenthaltszeit im See mit fünf Jahren vergleichsweise hoch. Entsprechend dauert es bis zu 15 Jahre bis sich nach einer Veränderung des P-Inputs ein neues Gleichgewicht eingestellt hat.

Hauptverantwortlich für den beobachteten Anstieg sind ungeklärte Siedlungsabwasser, welche diffus oder über Zuflüsse in den See gelangen. Landwirtschaftliche Quellen scheinen im Moment sekundär. Sie könnten jedoch in Zukunft zunehmen, wenn Kunstdünger für lokale Landwirte erschwinglich werden.

Im Gegensatz zu den Zuflüssen und diffusen Quellen tragen die die Wasserbilanz dominierenden Karst-Quellzuflüsse lediglich ~9 % zum gesamten P-Input bei. Allerdings besteht ein beachtliches Eutrophierungspotential in der Verbindung zum mesotrophen Prespasee, welcher siebenmal höhere P-Konzentrationen aufweist als der „stromabwärts" liegende Ohridsee. Im untersuchten Quellwasser konnte der vom Prespasee stammende Wasseranteil dank eines klaren Signals in Konzentrationen stabiler Isotopen und Cl^- quantifiziert werden. Der Vergleich verschiedener regionaler Karstquellen hat gezeigt, dass der grösste Teil des Ausflusses des Prespasees in den Ohridsee fliesst und ~20 % zu dessen Wasserbilanz beiträgt. Das mitgeführte P wird jedoch zu 65 % durch natürliche Filter und möglicherweise Fällungsprozesse auf dem Weg zurückgehalten. Dank dieser natürlichen Wasserreinigung macht der Zufluss vom Prespasee nur gerade 6 % des gesamten P-Inputs des Ohridsees aus. Diese P-Fracht würde sich vergrössern, wenn mehr P in den Prespasee gelangt oder sich dessen Wasserspiegel absenkt, wie in den letzten Jahren beobachtet. Im schlimmsten Fall könnte die Verbindung der beiden Seen zu einem 1 mg-P m^{-3} (20 %) Anstieg der P-Konzentrationen im Ohridsee führen. Während letzterer daher nur bedingt gefährdet ist, wird der Prespasee durch zunehmende Eutrophierung und ungebremste Wasserentnahme stark bedroht.

Nachdem die Wasser- und P-Bilanz quantifiziert wurde, stellt sich die Frage wie sensitiv der Ohridsee auf Eutrophierung, im Vergleich zu anderen menschlichen Veränderungen, wie zunehmender Wasserverbrauch oder globale Klimaerwärmung, reagiert. In einer Sensitivitätsanalyse wurden deshalb die Auswirkungen von (a) einer Verdoppelung der aktuellen P-Konzentration, (b) einem erhöhten Wasserverbrauch zur Bewässerung im Sommer und (c) der vorhergesagten 0.04 °C yr^{-1} Erwärmung der Atmosphäre, auf die physikalischen Bedingungen im Ohridsee ausgewertet. Wie erwartet zeigte sich eine weitergehende Eutrophierung als Bedrohung des Sees. Auf der einen Seite würde sie das Wachstum von nicht-endemischen Planktonarten an der Oberfläche einseitig fördern. Dieses Wachstum wiederum führte dann zu schlechteren Lichtbedingungen für die endemische Kieselalge Cyclotella fottii welche zwischen 30 und 150 Metern dominiert. Andererseits führt ein erhöhtes Algenwachstum zu mehr Mineralisation von organischem Material am Sediment und damit zu grösserer Sauerstoffzehrung. Das tiefe, nur unregelmässig durchmischte Hypolimnion könnte gar anoxisch werden, was verheerende Folgen für die endemische Bodenfauna hätte. Zusätzlich würden die negativen Auswirkungen einer Eutrophierung durch eine globale Klimaerwärmung verstärkt. Eine atmosphärische Erwärmung von 0.04 °C yr^{-1} würde die Auftretenswahrscheinlichkeit einer kompletten, konvektiven Durchmischung des Sees im Winter stark reduzieren und durch die generell erhöhte Stabilität der Wassersäule praktisch zu einer Isolation des Tiefenwassers führen. Der Effekt erklärt sich aus der verzögerten Reaktion des Tiefenwassers auf Veränderungen an der Oberfläche. Wenn sich die Oberfläche schnell erwärmt, so wird die Differenz zur Temperatur des Tiefenwassers immer grösser. Die Mischung des Sees und somit der Sauerstoffnachschub in grössere Tiefen ist also sehr sensitiv auf die Geschwindigkeit einer atmosphärischen Erwärmung und nicht auf die absolut höhere Temperatur. Es konnte gar gezeigt werden, dass in einem neuen Gleichgewichtszustand bei höherer Temperatur, wegen der Zunahme der Wärmeexpansivität von Wasser, die Durchmischungshäufigkeit grösser wäre als im heutigen Zustand. Im Gegensatz zu

Eutrophierung und globaler Erwärmung hätte eine Erhöhung des Wasserverbrauchs allenfalls lokale Effekte, zum Beispiel auf die Karst-Quellgebiete.

Entsprechend der Sensitivitätsanalyse müssen Massnahmen zur Verhinderung einer Eutrophierung des Ohridsees das Ausmass einer globalen Erwärmung berücksichtigen. Um die einzelnen und kombinierten Effekte der beiden Veränderungen zu quantifizieren, wurden verschiedene Szenarien mittels der Kopplung eines physikalischen und eines bio-geochemischen Seemodells getestet. Die Resultate zeigen, dass die P-Inputs um 50 % reduziert werden müssen, um auch bei einer atmosphärischen Erwärmung von 0.04 °C yr^{-1} während den nächsten 50 Jahren noch überall im See mehr als 4 mg L^{-1} gelösten Sauerstoff zu finden. Jedoch würde selbst bei Erreichen einer solchen P-Reduktion das Tiefenwasser immer isolierter. Entsprechend müsste die Erwärmungsrate in den nächsten 50 Jahren verlangsamt werden, um langfristig anoxische Verhältnisse im Ohridsee zu verhindern.

Die vorgestellte Arbeit war nur in enger Zusammenarbeit mit dem Mazedonischen Hydrobiologischen Institut (HBI) möglich. Neben der gemeinsamen Feld- und Laborarbeit wurde ein Programm zur Unterstützung des HBIs bei der Anwendung neuer Analysegeräte und beim Aufbau eines zukünftigen See-Monitorings durchgeführt. Das Projekt hat erfolgreich (i) die regelmässige und sachgerechte Verwendung von Geräten zur Probennahme motiviert, (ii) die Ausbildung von jungen Wissenschaftern des HBI an der neuen Hightech-Ausrüstung durch Praktika im Ausland gefördert und (iii) die Anbindung des HBI an internationale, wissenschaftliche Projekte deutlich verbessert. Hingegen konnte das Projekt die nationale Vernetzung des Instituts nicht positiv beeinflussen. Abschliessend kann ich die Kombination einer Doktorarbeit mit der Verantwortung für ein kleines, internationales Projekt nur empfehlen, sowohl was die Projektergebnisse als auch meinen persönlichen Erfahrungsgewinn anbelangt.

Chapter 1

Introduction

1.1 Lake Ohrid – A Unique Lake Ecosystem

1.1.1 Geography and Origin

The region of Lake Ohrid (693 m asl) is situated near 41° latitude in south-eastern Europe. Roughly two thirds of its surface area belong to Macedonia and one third to Albania. Through underground connection to upstream Lake Prespa (849 m asl) the catchment area of Lake Ohrid further extends to Greece. The two lakes are bounded to their East and West sides by high mountain chains, reaching to about 1500 m asl to the West of Lake Ohrid ("Mokra Mountain" chain), 2250 m asl between the two lakes ("Galicica Mountain" and "Mali i Thate Mountain" chains) and 2600 m asl to the East of Lake Prespa ("Baba Mountain" chain). Whereas the mountain ranges to the West of Lake Ohrid and to the East of Lake Prespa are mostly crystalline with limited water permeability, the rocks separating the two lakes are karstified carbonates forming aquifers with high hydraulic conductivity (Cvijic 1908; Eftimi et al. 2001).

Lake Ohrid and Lake Prespa are graben-type lakes, which are the result of collapsed, karstic poljes (Stankovic 1960). They are believed to have formed in mid Pliocene, roughly two to five million years ago (Stankovic 1960; Meybeck 1995). The reason why in particular Lake Ohrid still forms a deep permanent water body lies certainly in a large original depth and small sedimentation rate. In addition the Ohrid-Korca graben to the south of the lake is still tectonically active and might compensate sedimentation by subduction (Stankovic 1960). The activity of the zone is also underlined by regular earthquakes (Wagner et al. 2006).

Today, Lake Ohrid is a large ($A = 358\ km^2$), deep ($z_{mean} = 155$ m, $z_{max} = 288.7$ m) and one of the most voluminous lakes in Europe ($V = 55\ km^3$). In contrast to the deep and regular topography of Lake Ohrid, Lake Prespa ($A = 254\ km^2$, $V = 3.6\ km^3$) is shallow ($z_{mean} = 14$ m) with a few deeper holes ($z_{max} = 48$ m).

1.1.2 Hydrology

The relatively dry, Mediterranean climate and the small drainage basin (catchment/lake surface ratio of ~7) of Lake Ohrid results in a long hydraulic residence time scale of ~70 yr. The water balance is dominated by the inflow from karst aquifers (~50%) with smaller shares from rivers and direct precipitation. The river runoff was even lower by 5.5 $m^3\ s^{-1}$ (~70%), before 1962 when River Sateska was deliberately diverted from the north into the lake. The karst aquifers are charged from mountain range precipitation and from Lake Prespa, as revealed by natural tracers (Anovski et al. 1992; Eftimi & Zoto 1997). Apart from springs, which flow directly into Lake Ohrid on its shore, about 50% of the aquifers are expected to be sublacustrine. It is noteworthy that 50% of the already small catchment area is made up of the underground connection to Lake Prespa, which has no surface outflow. The water leaves Lake Ohrid by surface outflow in the north (~60%) and by evaporation (~40%) (Ivanova 1974; Watzin et al. 2002). Annual evaporation from the lake surface clearly outweighs direct precipitation.

1.1.3 Lake Ohrid and its Life

The top 150 to 200 m water column of Lake Ohrid follows the usual temperature stratification seasonality of deep, temperate lakes, whereas the deep hypolimnion is stably stratified by salinity. The stability due to the salinity gradient allows complete mixing only roughly once per decade during cold winters (Stankovic & Hadzisce 1953; Hadzisce 1966).

Both in terms of nutrient concentration, as well as biological parameters Lake Ohrid qualifies as oligotrophic (Naumoski 2001; Watzin et al. 2002). Thanks to this oligotrophy and the “filtered” spring inflows, the water is exceptionally clear with average Secchi depth of ~14 m (Patceva 2005). Despite the lack in annual deep water exchange from complete overturn or plunging rivers, dissolved oxygen (DO) never drops below ~6 mg L^{-1} (Naumoski 2000). While Lake Ohrid is special as such, by far the most spectacular quality is its impressive endemism. Similar to Lake Baikal, Lake Ohrid harbors endemic species covering the whole food-chain, from phytoplankton (e.g., *Cyclotella fottii*) over zooplankton (e.g., *Cyclops ochridanus*), cyprinid fish (e.g., *Pachychilon pictus*), to predatory fish (e.g., *Salmo letnica*) and finally its diverse endemic bottom fauna (e.g. *Ochridagammarus solidus*) (review in Salemaa 1994). Whereas most of the endemic species descriptions are based on morphological and ecological characteristics, some recent applications of molecular genetic techniques underline the specificity of the fauna (Korniushin et al. 2000; Sywula et al. 2003; Sell & Spirkovski 2004). The large number of described endemic species (> 200) make Lake Ohrid a hotspot of freshwater diversity (UN: World Conservation Monitoring Centre 1998; LakeNet: Duker and Borre 2001). The importance of the lake was further emphasized by UNESCO when the region was declared a World Heritage site (UNESCO 1979).

1.1.4 Socio-Cultural Importance

Throughout history many peoples – Illyrians, Macedonians, Romans, Slavs, Ottomans, Serbs, only to name a few – have passed through the area and left their traces in terms of culture and monuments (Poulton 2000). In particular the town of Ohrid played an important role as a cultural and religious center on an Eastern European scale. During the ninth century Ohrid had a thriving university with more than 3500 students and was made the first archbishopric by the Assembly of Constantinople (Watzin et al. 2002). As a result the area boasts a large number of temples, churches and fortresses bearing witness of almost 3000 years of agitated history.

The historic monuments, as well as the pristine lake environment make the area around Lake Ohrid a prime site for tourism. In the 1980s more than 200’000 national and international tourists went on a literal pilgrimage to the Macedonian lake side every year (Watzin et al. 2002). During the Yugoslav crisis and particularly after the interethnic conflicts within Macedonia in 2001 international tourism collapsed but has been slowly recovering during the past two years (Watzin et al. 2002; local information 2001 - 2005). Even though many of the above visitors are staying for a weekend only, tourism makes an important share of local economy (~1 visitor/inhabitant).

It is important to note that Lake Ohrid is regarded by most Macedonians as an "affair of state", if not as the outstanding national symbol of pride and identity. As a result every rumor on swan diseases, radio-active pollution or amazing fish catches quickly finds its way into national newspapers, notwithstanding the source of the information. However the high level of national attention is also an excellent starting point for mitigation efforts.

1.2 Is Lake Ohrid Jeopardized by Human Activities?

In the past 50 years population in the lake catchment has grown from about 70'000 to 170'000 inhabitants (Watzin et al. 2002; Macedonian State Statistical Institute, personal communication, 2003). As a result human pressure on Lake Ohrid is expected to have increased in parallel.

1.2.1 Eutrophication

A particular concern is potential lake eutrophication. Indeed, the negative effect of human activities can be observed in some heavily polluted inflows (Naumoski 2001; Veljanoska-Sarafiloska 2002). The decrease in water quality in the vicinity of such inflows and possible negative impacts on the special biology have been discussed in several puclications since the 1970s (Cado 1974; Ocevski 1974a; Taylor & Gerking 1978; Taylor et al. 1981; Serafimova-Hadzisce 1985; Cohen 1994). However, only more recent contributions are concerned about a possible eutrophication of the entire lake in the future (Ernst Basler and Partners 1995; Naumoski 2001). Because of the slow adaptation of the endemic species to nutrient-poor conditions, eutrophication could in turn lead to irreversible losses. Observations of shifts away from endemic species in polluted littoral sites reinforce this fear of extinction (Watzin et al. 2002). Moreover the negative impacts from eutrophication would be prolonged and intensified by the special mixing properties and the long water residence time of Lake Ohrid.

1.2.2 Water Abstraction

In the two riparian countries, Macedonia and Albania, which are both in political transition, agriculture is still a major source of economy. Given the semi-arid climate of the area (European Environment Agency 2003), irrigation puts strong pressure on the local surface waters. During the summer season, water is abstracted from all tributaries of both upstream Lake Prespa and Lake Ohrid, reducing those river flows to a mere trickle or setting them completely dry. Moreover there are groundwater pumps and systems to take water directly from Lake Prespa. The growing need for irrigation water has its most obvious effect on the water level of Lake Prespa, which has dropped by ~6 m over the past decade (Hollis & Stevenson 1997; Anovski 2001).

1.2.3 Global Climate Change

While local human impacts can affect Lake Ohrid directly, global climate change might also be important. Regional climate simulations for increasing atmospheric CO_2 concentrations predict warming above average for the Balkan Peninsula (Giorgi et al. 2004; Räisänen et al. 2004). Warmer temperatures lead in turn to an intensification of the hydrological cycle, which can affect storage in lakes, particularly in semi-arid regions (IPCC 2002). Finally an increase in water temperatures has been observed to lead to prolonged isolation of lower waters in deep, stratified lakes (IPCC 2002; O'Really et al. 2003).

1.2.4 Activities of Lake Management

In the framework of an initiative by the World Bank (Ernst Basler and Partners 1995) actions have been started to tackle potential eutrophication. In particular, the collector system was improved to cover a larger share of the Macedonian catchment and to prevent leakage from existing sewerage. For the Albanian catchment, where no canalization exists at the moment, possible solutions are evaluated by the German KfW Development (R. Sampson, personal communication).

Whereas the above activities go into the right direction it is important to know whether they are enough to preserve the lake and its unique species for future generations. This thesis aims to support planning of future management activities by answering the following questions:

- Is Lake Ohrid in a process of eutrophication?
- Is the present nutrient load inflicting changes on the lake system?
- Do other human impacts – water abstraction or global warming – interfere by amplifying or reducing the effects from eutrophication?
- What would be a "sustainable" nutrient load, which can preserve Lake Ohrid as a suitable habitat for its endemic species into the future?

1.3 Approach & Outline

The above questions require a careful systems analysis of Lake Ohrid, as well as a modeling approach to make predictions about the future. Both steps involve a number of specific scientific challenges, which are tackled in a step-by-step approach from chapter 2 through 4. Each of the three chapters also represents an individual manuscript, which has been submitted to an international, peer-reviewed journal. In the following overview the main content of each chapter, as well as its implication for the overall aim are briefly outlined.

1.3.1 Chapter 2: Is Lake Prespa Jeopardizing the Ecosystem of Ancient Lake Ohrid?

The karst springs are the largest inflow to Lake Ohrid, yet little is known about them. In particular the water stemming from upstream, mesotrophic Lake Prespa has a large current and future eutrophication potential for Lake Ohrid. Based on monitoring of Lake Prespa, as well as several spring inflows, chapter 2 assesses the underground water and phosphorus transport. With the help of natural tracers contributions from Lake Prespa and local precipitation are distinguished. Finally a detailed phosphorus (P) balance allows the evaluation of the historic, current and potential future P loading from groundwater inflows to Lake Ohrid. The results on the groundwater share of the water and P balance of Lake Ohrid are the pre-condition for the analyses in chapters 3 and 4.

1.3.2 Chapter 3: Sensitivity of Ancient Lake Ohrid to Local Anthropogenic Impacts and Global Warming

Whereas many aspects of Lake Ohrid have been described over the past century, physical lake processes were mostly neglected. To fill this gap chapter 3 gives a detailed account of major physical lake properties based on four years of observation.

Lake physics also form an important boundary condition for the special biological communities of Lake Ohrid, e.g., through light availability or vertical exchange of dissolved oxygen. As a consequence they are used as a basis to test the sensitivity of Lake Ohrid to expected human impacts, eutrophication, water abstraction and global warming.

On the one hand, chapter 3 sets the physical boundary conditions for model approaches. On the other hand human impacts are identified to which the unique species are expected most sensitive.

1.3.3 Chapter 4: Eutrophication of Ancient Lake Ohrid – Global Warming Amplifies Detrimental Effects of Increased Nutrient Inputs

Finally the state of eutrophication of Lake Ohrid is assessed, based on regular inflow measurements, sediment cores and traps, as well as vertical nutrient profiles from the lake.

Having assembled information on the water balance including the karst aquifers (chapter 2), physical boundary conditions such as mixing processes (chapter 3) and internal cycle and human sources of phosphorus (chapter 4) allows the establishment and calibration of a numerical, bio-geochemical lake model. In the second part of chapter 4, different scenarios are established including the variation of local eutrophication and global warming - two processes which were shown to be important in chapter 3. Using dissolved oxygen in the hypolimnion as a goal function (expressing the ecological needs of some of the endemic species), sustainable phosphorus loads are defined, dependent on the extent of global warming.

1.3.4 Chapter 5: Capacity Building

One major task of the PhD thesis was the setup of a monitoring program in collaboration with the Hydrobiological Institute (HBI) in Ohrid. Moreover – through the main financial support of the Swiss Secretariat for Economic Affairs (seco) – the aim of the project went beyond the scientific analysis and included capacity building at HBI via the organization of scientific exchange on various levels. Chapter 5 accounts for this important project part, which is not yet reflected in scientific publications.

Chapter 2

Is Lake Prespa Jeopardizing the Ecosystem of Ancient Lake Ohrid?

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Abstract

Lake Prespa and Lake Ohrid, located in south-eastern Europe, are two lakes of extraordinary ecological value. Although the upstream Lake Prespa has no surface outflow, its waters reach the 160 m lower Lake Ohrid through underground hydraulic connections. Substantial conservation efforts concentrate on oligotrophic downstream Lake Ohrid, which is famous for its large number of endemic and relict species. In this paper, we present a system analytical approach to assess the role of the mesotrophic upstream Lake Prespa in the ongoing eutrophication of Lake Ohrid. Almost the entire outflow from Lake Prespa is found to flow into Lake Ohrid through karst channels. However, 65% of the transported phosphorus is retained within the aquifer. Thanks to this natural filter, Lake Prespa does not pose an immediate threat to Lake Ohrid. However, a potential future four-fold increase of the current phosphorus load from Lake Prespa would lead to a 20% increase (+ 0.9 mg P m^{-3}) in the current phosphorus content of Lake Ohrid, which could jeopardize its fragile ecosystem. While being a potential future danger to Lake Ohrid, Lake Prespa itself is substantially endangered by water losses to irrigation, which have been shown to amplify its eutrophication.

2.1 Introduction

Lake Ohrid and Lake Prespa form a very unusual lake system. Situated in south-eastern Europe between Albania, Macedonia and Greece (Figure 2.1), they are both of tectonic origin with an estimated age between 2 and 35 million years (Jakovljevic 1935; Stankovic 1960; Meybeck 1995). According to Stankovic (1960), the two lakes formed one lake at their earliest stages of existence. Today, Lake Prespa lies about 160 meters higher than Lake Ohrid (Table 2.1), separated by the Mali i Thate / Galicica mountain range, which consists of karst rock structures (Eftimi et al. 2001). Stable isotope

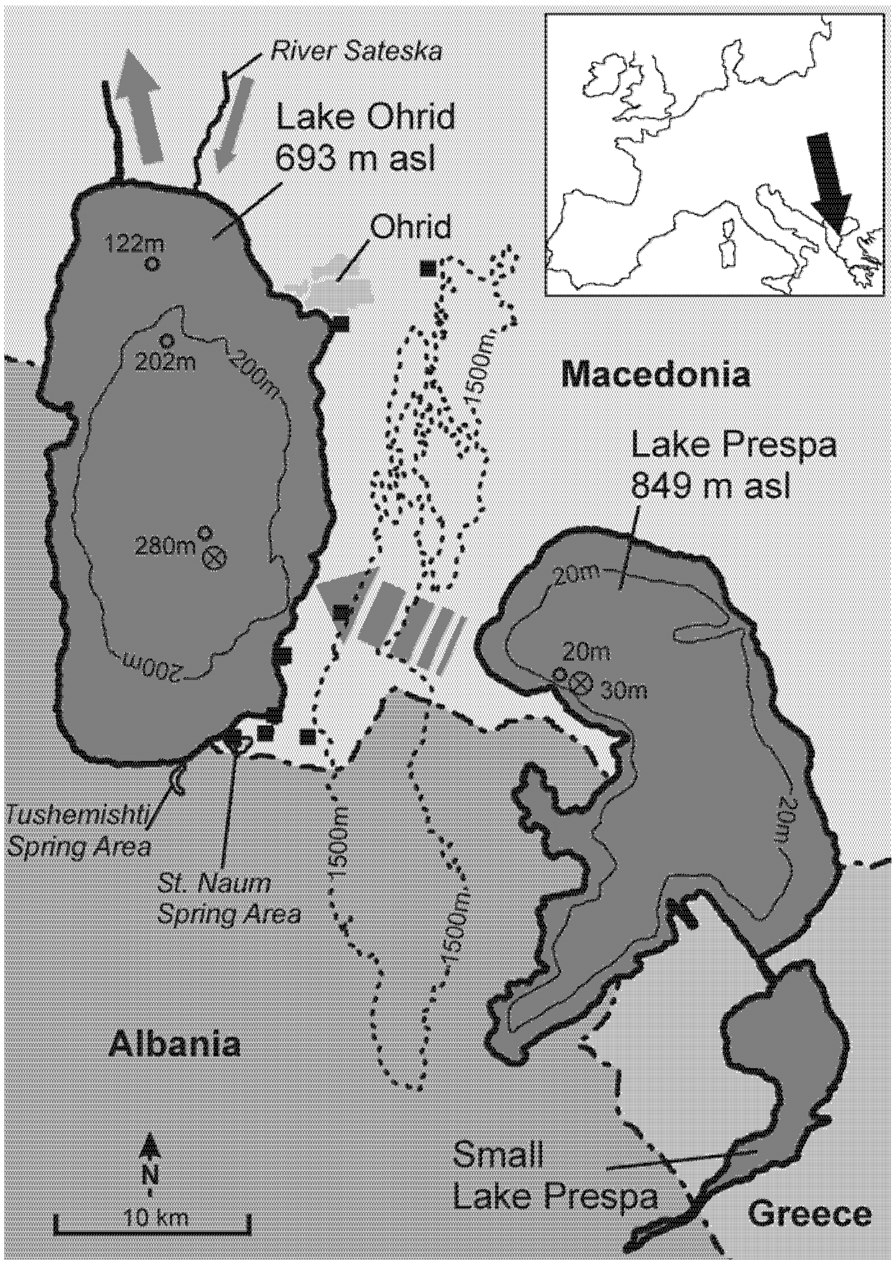

Figure 2.1: Overview of study area indicating the different sampling sites. The grey arrows indicate flow direction (the broken arrow signifies underground flow). Crossed circles indicate main lake sampling sites, empty circles are sediment core locations and filled squares are sampled karst springs. The inset map shows the location of the study area within Europe.

measurements, as well as recent tracer experiments have revealed that water from Lake Prespa is flowing to Lake Ohrid through karst channels (Anovski et al. 1980; Eftimi & Zoto 1997; Zoto, pers. comm. 2003).

Lake Ohrid harbors a large number of endemic species, which have been studied extensively since the beginning of the 20th century. Most of the endemic species are benthic forms (e.g., *Pyrgula macedonica* (freshwater snail; Stankovic 1960), *Ochridaspongia rotunda* (freshwater sponge; Gilbert & Hadzisce 1984)), but there is also a number of endemic plankton species (e.g., *Cyclops ohridanus* (zooplankton), *Cyclotella fottii* (phytoplankton), both described by Stankovic 1960) as well as fish (e.g., *Salmo letnica* (Ohrid trout; Sell & Spirkovski 2004)). Through rising international interest, the research institution "Station Hydrobiologique – Ohrid" was founded in Ohrid (Figure 2.1) in 1935, dedicated to the study of Lake Ohrid and its extraordinary species (Serafimova-Hadzisce 1985). The importance of Lake Ohrid was further emphasized by UNESCO when the region was declared a World Heritage site (UNESCO 1979).

With increasing public attention concerns about the conservation of Lake Ohrid are also growing. Particular focus is the preservation of its oligotrophic state, which seems to be in jeopardy due to rising population and tourism. Indeed a slow eutrophication can be detected in sediment cores of Lake Ohrid over the past ~100 years (Matzinger et al. 2004). Moreover, changes in biological communities have been observed over the past decades (Watzin et al. 2002). In 1997 a project was launched by the Global Environment Facility to tackle future problems with (i) the improvement of the urban waste water treatment system covering the major settlements in Macedonia, (ii) consolidation and extension of water quality monitoring activities and (iii) the establishment of bilateral lake management (Ernst Basler and Partners 1995). Apart from direct pollution from riparian towns, villages and agricultural fields, it is important to understand the influence of the underground inflow from Lake Prespa. Given that Lake Prespa contributes 50% of the total catchment of Lake Ohrid and that the total phosphorus (TP) concentration (Table 2.1) of Lake Prespa is seven times higher than in Lake Ohrid, the development of Lake Prespa is a worrying concern for the eutrophication of downstream Lake Ohrid.

Table 2.1: Characteristics of Lake Prespa and Lake Ohrid

Property	**Unit**	**Lake Prespa**	**Lake Ohrid**
Altitude	m asl	849 (854)[3]	693
Catchment area	km^2	1300[1]	2610[2]
Lake surface area	km^2	254 (282)[3]	358
Maximal depth	m	~ 48 (54)[3]	288
Mean depth	m	14 (19)[3]	155
Volume	km^3	3.6 (4.8)[3]	55.4
Hydraulic residence time	yr	~ 11 (17)[3]	~ 70
Average phosphorus concentration TP (2003)	mg P m^{-3}	31	4.5
Number of endemic species	-	~ 10 ?	> 150
Number of inhabitants in catchment area	-	~ 24,000	~ 174,000[2]
Number of tourists per year	-	< 1000 ?	~ 50,000

[1] including Small Lake Prespa and its catchment
[2] including Lake Prespa and its catchment
[3] value in parentheses: in the 1980s before recent water level decline of Lake Prespa

Although similar in surface area, Lake Prespa is much shallower than Lake Ohrid (Table 2.1). Archaeological findings indicate that its water level was subject to large variations in the past (Sibinovik 1987; Milevski et al. 1997). Compared to Lake Ohrid, much fewer endemic species have been described (Karaman 1971; Crivelli et al. 1997; Shapkarev 1997). With its nearly untouched shoreline and large reed-belts, Lake Prespa is an important breeding ground for various water birds, such as the rare Dalmatian Pelican *Pelecanus crispus* (Crivelli 1996; Nastov 1997). As a result Lake Prespa was declared a Ramsar site in 2000 (Ramsar 2000). However, serious concerns have been expressed about the recent water level decline, as well as potential eutrophication of the lake (Naumoski et al. 1997; Löffler et al. 1998; Goltermann 2001).

The goal of this paper is to quantify the role of these anthropogenic changes in Lake Prespa in regard to the ongoing eutrophication of Lake Ohrid. Such an assessment is important and urgent for defining potential management options and for making optimal use of the limited financial resources to protect the two lakes. Furthermore, it is crucial that mitigation measures are taken in time, given the sluggish dynamics of Lake Ohrid with its hydraulic residence time τ ~70 yr (Table 2.1). In our analysis we focused on phosphorus, as it is clearly the growth-limiting nutrient (N:P > 25:1).
A system analytical approach is presented, which begins with the assessment of the anthropogenic changes in upstream Lake Prespa, quantification of the underground transfer of water and phosphorus and finally evaluation of the effects on downstream Lake Ohrid in a linear phosphorus model.

2.2 Study Sites

2.2.1 Lake Prespa

Lake Prespa, situated at an altitude of ~849 m asl, is surrounded to the east and west by up to 2000 m high mountains. Symptomatic of the sparse knowledge of Lake Prespa is the broad range of values given by various authors for the areas of the lake (254 to 285 km^2) and its catchment (822 to 1775 km^2). In the following, we use the values from a recent compilation by Anovski (2001) given in Table 2.1, to which scientists of all three riparian countries contributed. The bathymetry shows that the lake is mostly shallower than 30 m with a few local deeper holes (Figure 2.2). A topographic map from the 1940s (provided by the Hydrobiological Institute Ohrid (unpublished data, ca. 1940)) indicates that the present water level is ~5 to 6 m lower than in the 1940s. Significant lake level decreases have also been observed since the late 1980s (Figure 2.3). To the south, Lake Prespa is connected to Small Lake Prespa (Figure 2.1) by a controllable man-made channel with a current hydraulic head of ~3 m (Hollis & Stevenson 1997). As Lake Prespa is relatively shallow compared to its large surface area, wind and convective mixing lead to complete destratification of the entire water column from September to April/May and consequently all dissolved substances are homogenized annually.

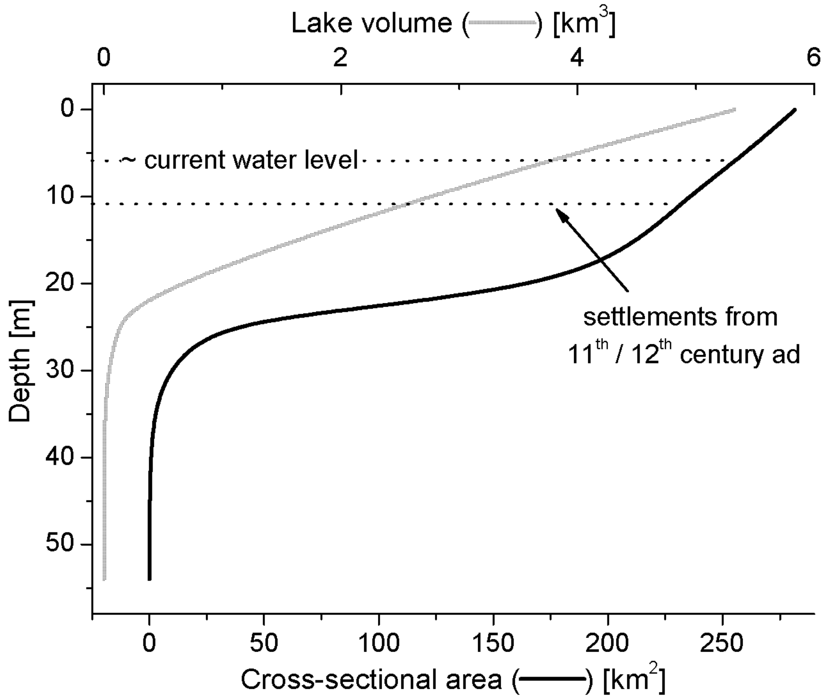

Figure 2.2: Hypsograph of Lake Prespa: Cross-sectional area and enclosed volume as a function of depth relative to the “historic” water level (~6 m above current level). Data are from a map from the 1940s (Hydrobiological Institute, unpublished data). The horizontal lines indicate known water levels of Lake Prespa.

2.2.2 Lake Ohrid

Lake Ohrid, situated between mountain ranges to the east and the west, is oligotrophic, deep (max. depth ~288 m), large (surface area ~358 km^2) and one of the most voluminous lakes (~55 km^3) in Europe (Table 2.1). Apart from its unique flora and fauna, a peculiarity is the long hydraulic residence time, which results from the relatively dry, Mediterranean climate and the small drainage basin. The water balance is dominated by inflow from karst aquifers (55%) with slightly smaller shares from river runoff and direct precipitation (Matzinger et al. 2004). The fraction of river runoff was even below 10% before 1962, when River Sateska was deliberately diverted into Lake Ohrid (Figure 2.1). In contrast to Lake Prespa, Lake Ohrid is an oligomictic lake with complete mixing occurring roughly once per decade (Hadzisce 1966).

2.2.3 Karst Springs

Along the western side of the Galicica / Mali i Thate Mountain Range (Figure 2.1), which separates the two lakes, numerous karst springs arise and often seep away shortly after their appearance or flow directly into Lake Ohrid. The springs, originating at an altitude higher than Lake Prespa, are charged from mountain range precipitation. Recent tracer experiments on the lower altitude springs have revealed that two particularly large spring areas, which flow into Lake Ohrid on its south-eastern shore, are partly fed from Lake Prespa. Each of the two areas, Tushemishti in Albania and St. Naum in Macedonia, consist of dozens of spring holes (Figure 2.1).

2.3 Materials and Methods

In order to assess the effects of a potential deterioration of Lake Prespa on Lake Ohrid, a sampling and measurement program was established in the area, as the basis of our system analytical approach. The sampling program, analytical methods, as well as calculations are presented in the following. The declaration of materials and methods is arranged in the same order and under the same subtitles as the results in the "Results and Discussion" section. Table 2.2 gives an overview of all samples and analytical parameters within our program.

Table 2.2: Overview of sampling and measurement program

Site	**Sampling period**	**Number of samples**	**Parameters**
Phosphorus Load from Lake Prespa[1]			
Lake Prespa (0, 5, 10, 15, 20, 25, 30 m)	12/2002-9/2003	44 36	SRP, TP, DO NO_2^-, NH_4^+, TN
Lake Prespa sediment core	5/2002	43 15 18	TC, TIC, TN, Water content TP ^{137}Cs, ^{210}Pb
Transfer to Lake Ohrid[1]			
Lake Prespa (0, 5, 15 m)	4/2001-9/2003 (3 different dates for isotopes)	3 21 32	$\delta^{18}O$, δD Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-} TP, SRP
Lake Ohrid (0, 5, 50, 250 m)	4/2001-10/2002	4	$\delta^{18}O$, δD
St. Naum Springs	4/2001-9/2003	7 11 10 12	$\delta^{18}O$, δD Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-} TP SRP
Other springs	4/2001-9/2003	11 10 10 21	$\delta^{18}O$, δD Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-} TP SRP
Rain samples	5/2001-10/2002	8	^{18}O, D
Effect on Lake Ohrid[1]			
Lake Ohrid (0, 10, 20, 30, 40, 50, 75, 100, 150, 200, 250, 275 m)	7/2002-9/2003	79	TP
Lake Ohrid sediment cores	5/2002 & 4/2003	76 13	TP, Water content ^{137}Cs, ^{210}Pb

[1] Titles in italics correspond with the subtitles in "Materials and Methods" and "Results and Discussion" Sections.

2.3.1 Lake Prespa Underground Outflow

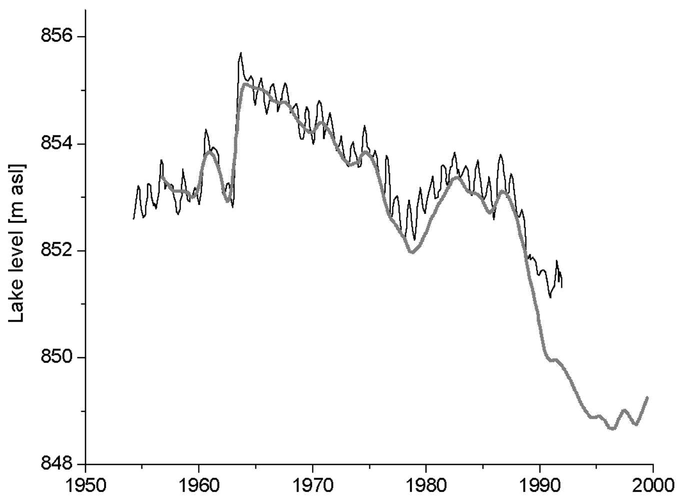

Figure 2.3: Development of Lake Prespa water level over the past five decades. The fine line shows seasonal fluctuations according to Greek measurements presented in Hollis and Stevenson (1997). The bold, grey line represents annual means from Albanian measurements, adapted from Anovski et al. (2001).

Before we can assess the potential effect of Lake Prespa, the underground outflow must be quantified. Whereas a rough water balance for Lake Prespa can be drawn from earlier publications, a simple model was developed to account for the effect of the observed level decline. The continuous decrease (Figure 2.3) indicates that a large share of the underground channels is located below the present lake surface, making Lake Prespa particularly vulnerable to changes in water input. As a result, any additional water output $Q_{out,add}$, for example for irrigation, will lead to a decrease in water level z until a new equilibrium is reached at the level z = h (z is the vertical coordinate, positive upward; highest level: z = 0 m, deepest spot: z = -54 m). In our model, equilibrium level h is attained when

$$Q_{out,add} = Q_{out,undgr}(z=0) - Q_{out,undgr}(z=h) + \underbrace{E_{net} \cdot \left(A(z=0) - A(z=h)\right)}_{(i)} \qquad (2.1)$$

where $Q_{out,add}$ [$m^3\ yr^{-1}$] is the additional water consumption, which leads to a change in the water balance, $Q_{out,undgr}$ [$m^3\ yr^{-1}$] is the underground outflow for water level z, A [m^2] is the lake surface area, and E_{net} [$m\ yr^{-1}$] is the net evaporation from the lake surface (= evaporation – precipitation = 0.37 $m\ yr^{-1}$). If we assume that the outflow channels are distributed evenly over the western half of the lake bottom and that the groundwater velocity scales with the hydrostatic pressure, the underground outflow $Q_{out,undgr}$ in equation (2.1) can be expressed for an arbitrary water level z = h as

$$Q_{out,undgr}(z=h) = Q_{out,undgr}(z=0) \cdot \left(\underbrace{\frac{A(z=h)}{A(z=0)}}_{ii} \cdot \underbrace{\sqrt{\frac{V(z=h) \cdot A(z=0)}{V(z=0) \cdot A(z=h)}}}_{iii} \right) \qquad (2.2)$$

where V [m^3] is the lake volume. If we combine equations (2.1 and 2.2), three stabilizing processes are considered that are effective during the water level decline (letters correspond with the terms in equations (2.1 and 2.2)): (i) reduced evaporation due to smaller lake surface, (ii) karst outflow channels that are set dry, (iii) decrease in hydrostatic pressure (assumed relative to average lake depth, V/A).

If the recent water level decline is the result of an additional output (e.g., irrigation) rather than an increase in evaporation, a decrease in the lake level will change the total water outflow $Q_{out,tot}$ from Lake Prespa. Using equations (2.1 and 2.2) we find:

$$Q_{out,tot}(z=h) = Q_{out,add} + Q_{out,undgr}(z=h) = Q_{out,undgr}(z=0) + E_{net} \cdot \left(A(z=0) - A(z=h)\right) \quad (2.3)$$

According to equation (2.3), $Q_{out,tot}$ will increase during water level decline by the reduction in net evaporation from the lake surface, although $Q_{out,undgr}$ decreases. Under that assumption the level-dependent bulk water residence time

$$\tau(z=h) = \frac{V(z=h)}{Q_{out,tot}(z=h)} \quad (2.4)$$

will decrease with declining water level.

2.3.2 Phosphorus Load from Lake Prespa

The concentration of phosphorus (P) in Lake Prespa was monitored in order to know the export loads from the upstream lake by underground outflow. In addition nitrogen (N) species and dissolved oxygen (DO) were measured to assess the extent of eutrophication in Lake Prespa. Furthermore, sediment samples were analyzed and sedimentation rates determined to reconstruct the history of eutrophication and the related potential increase in the outflow of P.

Water Samples – At one of the deep sites in Lake Prespa (depth ~ 30 m; Figure 2.1) water samples were collected ~bimonthly at five meter depth intervals using a Niskin bottle (Table 2.2). The campaign started in December 2002 and was continued until September 2003. Samples were stored in new or acid-rinsed plastic bottles and cooled for transport. Total phosphorus (TP), total nitrogen (TN), soluble reactive phosphorus (SRP), nitrite (NO_2^-) and ammonium (NH_4^+) were analyzed colorimetrically using standard analytical methods (DEW 1996). Mean measurement errors were 1.9 mg P m^{-3} for TP, 20 mg N m^{-3} for TN, 0.5 mg P m^{-3} for SRP, 0.2 mg N m^{-3} for NO_2^- and 2 mg N m^{-3} for NH_4^+. DO was measured with the Winkler method (Table 2.2).

For average lake concentrations annual or perennial averages of the volume integrated single profiles were used. Volume integration was performed numerically using

$$\langle X \rangle_h = \int_{z=z_{max}}^{h} X(z) A(z) dz \quad \{ \int_{z=z_{max}}^{h} A(z) dz \}^{-1} \quad (2.5)$$

where X [mg m^{-3}] is the concentration of the considered parameter, e.g., TP, $\langle X \rangle_h$ is the volume-average of X within the lake water between the surface level z = h and the maximal depth $z = z_{max}$. The cross-sectional areas A(z) are taken from the function plotted in Figure 2.2.

Sediment Sampling – One sediment core was retrieved from a depth of 20 m in Lake Prespa, close to the water sampling site (Figure 2.1), using a gravity corer (Kelts et al.

1986) and subsequently sectioned to 1 to 2 cm long vertical segments. For each segment, the water content was measured by weight loss after freeze-drying. TP was measured photometrically after digestion with $K_2S_2O_8$ in an autoclave for 2 hours at 120 °C (DEW 1996). Total carbon (TC) and total nitrogen (TN) were analyzed with a combustion CNS-Analyzer (EuroVector Elemental Analyzer). Total inorganic carbon (TIC) was measured by infrared absorption of CO_2 after acidifying the sample with 3M HCl (Skoog et al. 1996). Total organic carbon (TOC) was calculated as TOC = TC – TIC.

For the dating of the core ^{137}Cs and ^{210}Pb activities were established from gamma-counting in Ge-Li borehole detectors (Hakanson & Jansson 1983). As no clear peaks could be identified in the ^{137}Cs profile, only ^{210}Pb was used for the dating. In the top few cm of the core, the ^{210}Pb activity was practically constant, probably because of bioturbation by benthic organisms. Below the homogeneous layer, the ^{210}Pb signal decreased exponentially to background activity. The exponential fit led to a sedimentation rate SR of 0.075 ± 0.008 cm yr^{-1} (correlation R^2 = 0.96; the method is detailed in Wieland et al. (1993) and Doskey & Talbot (2000)).

Sediment accumulation rates were calculated using the following equation:

$$S_M = (1 - POR) \cdot SR \cdot \rho_{sed} \tag{2.6}$$

where S_M [kg m^{-2} yr^{-1}] is the mass accumulation rate, POR [-] is porosity calculated from the water content, SR [m yr^{-1}] is the sedimentation rate from ^{210}Pb dating and ρ_{sed} = 2500 kg m^{-3} is the sediment density established by pycnometer. The average S_M in the dated section of the core (top 13 cm) was used to calculate the current accumulation rates for TP, TN and TOC, by multiplication with their respective measured proportions. For total lake accumulation the rates above were multiplied with the surface area A(z = h).

Phosphorus Balance – A balance of TP was used to assess the effect of anthropogenic changes. The phosphorus balance of a lake is generally expressed as:

$$V \frac{\partial \langle TP \rangle}{\partial t} = P_{inp} - P_{sed,net} - P_{out} \tag{2.7}$$

where <TP> [mg m^{-3}] is the volume-averaged concentration of TP [equation 2.5], ∂<TP>/∂t [mg m^{-3} yr^{-1}] is the rate of change of <TP>, P_{inp} [mg yr^{-1}] is the annual phosphorus input, $P_{sed,net}$ [mg yr^{-1}] is the area-integrated net sedimentation, and P_{out} [mg yr^{-1}] is the outflow of TP. Given the simplicity of equation (2.7) all parameters represent annual or perennial averages. For our purposes, we used the linear model by Vollenweider (1969):

$$\frac{\partial \langle TP \rangle}{\partial t} = \frac{1}{V} P_{inp} - \sigma \cdot \langle TP \rangle - \frac{\beta}{\tau} \cdot \langle TP \rangle \tag{2.8}$$

where $P_{sed,net}$ from equation (2.7) is assumed proportional to the total phosphorus content with the sedimentation rate constant σ [yr^{-1}] and the outflow P_{out} from equation (2.7) is expressed by the water outflow (V/τ) times the average outflow concentration β·<TP> (where $\beta = TP_{surface}/\langle TP \rangle$). From equation (2.8), the residence time of P in the water column results as $\tau^* = ((\beta/\tau) + \sigma)^{-1}$, where τ* quantifies the e-folding time with which the equilibrium concentration $\tau^* \cdot P_{inp}/V$ is approached. For Lake Prespa V and τ depend on the water level h. For changes in h, V was adapted according to the function in Figure 2.2 and τ was replaced by equation (2.4).

2.3.3 Transfer to Lake Ohrid

In order to understand the transfer from Lake Prespa to Lake Ohrid, different tracers, as well as nutrients, were measured in samples from Lake Prespa and in springs which flow into Lake Ohrid.

Water from eight different karst spring areas (Figure 2.1), Lake Prespa and Lake Ohrid was collected from 2001 to 2003. In addition rain water was collected at different altitudes (Table 2.2). The springs at St. Naum were sampled at a ~three month interval. The other springs were sampled a total of between two and eight times, whenever opportunity was given, as some of them are hard to access. Samples were stored in new or acid-washed plastic bottles. TP and SRP were measured as described for Lake Prespa above. Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-} were determined with ion chromatography (IC 690 equipped with a Super-Sep column for cations; IC 733 with 753 suppression module for anions, all Metrohm, Switzerland; methods in Weiss 2004). Because of an unfortunate loss of samples only four springs were analysed for TP and ions (Table 2.2).

Water samples for stable isotope analysis were stored in glass vials and cooled immediately after sampling. $^{18}O/^{16}O$ and D/H ratios were analysed by isotope ratio mass spectrometry (GV Instruments IRMS (Manchester, UK) in continuous flow mode) at the Limnological Research Center Kastanienbaum. $\delta^{18}O$ and δD isotope compositions of the water samples were expressed using the delta-notation (δ) as a per mille deviation from the internationally accepted Vienna Standard Mean Ocean Water (VSMOW). The analytical errors are 0.3 ‰ and 0.8 ‰ for $\delta^{18}O$ and δD, respectively. Samples were equilibrated with a CO_2 and He mixture for $\delta^{18}O$ and with a H_2 and He mixture for δD analysis at 40 °C for at least 12 hours prior to the measurement. The method used is adapted from Werner & Brand (2001).

The share of Lake Prespa water in the St. Naum Springs flow was calculated using different tracers with the following equation:

$$r_{PR} = \frac{C_{SN} - C_{PS}}{C_{PR} - C_{PS}} \cdot 100\% \qquad (2.9)$$

where r_{PR} [%] is the share of Lake Prespa water and C [mg m^{-3}] is the tracer concentration; subscript SN stands for St. Naum Springs, PS for precipitation-fed springs and PR for Lake Prespa. Deviations from the mean were individually calculated for each place and parameter. For r_{PR} the individual errors were combined through error propagation. Based on these errors Δr_{PR}, an error-weighted average was calculated from all conservative tracers.

2.3.4 Effect on Lake Ohrid

In order to evaluate potential effects on Lake Ohrid, its P balance of the form of equation (2.8) was also established. For that reason, TP was also measured in the water column and in the sediment of Lake Ohrid.

Water profiles were drawn from May 2002 to September 2003 (Table 2.2). Water samples were treated and analysed for TP using the same methods as for Lake Prespa.

Three sediment cores were taken along the North-South axis of Lake Ohrid (Figure 2.1) in 2002 and 2003. An exponential fit to the measured ^{210}Pb activities (R^2 = 0.97) of one core resulted in a sedimentation rate of 0.089 ± 0.006 cm yr^{-1}. For the analysis of TP in the sediment, the same methods described for Lake Prespa were applied. P accumulation rates $P_{sed,net}$ were calculated by multiplying measured TP contents with S_M (equation 2.6) for all three cores. For equation (2.8), the average, estimated between 1.5 and 2.5 cm sediment depth, was used as the current value of $P_{sed,net}$.

2.4 Results and Discussion

2.4.1 Lake Prespa Underground Outflow

To assess the current underground outflow from Lake Prespa through the karst aquifers and its potential variability, we need to understand the overall lake water balance. The three major contributors to water input are river runoff from numerous small streams, direct precipitation onto the lake surface and inflow from Small Lake Prespa. The loss terms in the water balance are evaporation, diversion for irrigation as well as outflow through karst aquifers and fissures on its western shore (Eftimi et al. 2001; Figure 2.1).

Table 2.3: Water balance of Lake Prespa from Anovski et al. (2001)

Process	**Inflow** [Mio m^3 yr^{-1}]	**Outflow** [Mio m^3 yr^{-1}]	**Fraction during dry season**[1] [%]	**Fraction during wet season**[1] [%]
Runoff from sub-area in E and SE (q = 13 L km^{-2} s^{-1})	199		37	63
Runoff from karst sub-area in W and SW (10% of total precipitation)	31		37	63
Runoff from sub-area in plain in N (q = 9 L km^{-2} s^{-1})	69		37	63
Precipitation on lake surface (average from 6 stations along shore ~ 735 mm yr^{-1})	186		37	63
Inflow from Small Lake Prespa and from aquifers in Greece	49		50	50
Evaporation (average from three different approaches ~ 1100 mm yr^{-1})		279	78	22
Irrigation with lake water		10	100	0
Underground outflow (from balance)		245	50	50
Total	534	534		

[1] Dry season is from April to September, wet season from October to March. Relative contributions of river runoff, precipitation and evaporation are based on Figure 2.4.

Anovski et al. (2001) compiled measurements of hydraulic loads q [L km^{-2} s^{-1}] (runoff generated per land surface area) from various authors and combined them with their own measurements to calculate a water balance (Table 2.3). For a consistency check, we compared this water balance with the seasonal level fluctuations of Lake Prespa measured by Hollis & Stevenson (1997) (Figure 2.3). The de-trended water level shows a very regular seasonal oscillation with a period of one year and average amplitude of ~0.3 m. The lake level oscillations are the result of the seasonal changes in the relative contribution of the two major terms in the water balance, evaporation and precipitation (Figure 2.4). If we assume that river runoff follows a similar pattern as precipitation at lake level, we would expect a net water input from October to March and a net water loss from April to September. Each term of the water balance was split up between the dry and wet seasons, based on the relative fractions of evaporation and precipitation in Figure 2.4 (third and fourth columns in Table 2.3). Groundwater in- and outflows were assumed to be constant, whereas irrigation was considered only during dry seasons. The separate balance for two time periods resulted in a seasonal volume change of ± 145 Mio m^3. This corresponds to a surface height variation of ± 0.29 m, which fits with the observed annual oscillation in Figure 2.3. This agreement indicates that the water balance by Anovski et al. (2001) is within the correct range.

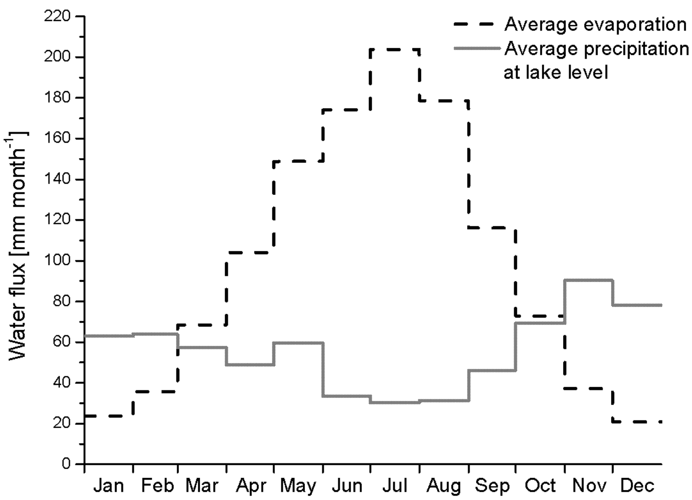

Figure 2.4: Seasonal evaporation (from Anovski et al. 2001) and precipitation (from Hollis & Stevenson 1997; Ristevski et al. 1997) at the surface of Lake Prespa. 100 mm month^{-1} corresponds to 0.8 m^3 s^{-1}.

However, this consistent water balance cannot explain the recent decrease of the Lake Prespa water level. This water loss, apparent in Figure 2.3, continued resulting in a total decrease of ~6 m between 1986 and 1996 (Chavkalovski 1997; Löffler et al. 1998; Anovski et al. 2001). From 1996 to 1999, the level stabilized (Anovski et al. 2001) but decreased further from 2000 to 2002 (Cakalovski, pers. comm. 2002). According to Figure 2.2, a decrease of ~5 m corresponds to a loss of ~ 1.2 km^3 of water, which is roughly 25% of the total lake volume. Different explanations have been proposed for the observed decrease (Chavkalovski 1997; Hollis & Stevenson 1997; Anovski et al. 2001): (a) a tectonic subduction of the lake bottom, (b) an opening of underground channels, due to seismic activity, (c) climatic variability or (d) increased irrigation. Comparing a topographic map (probably from the late 1940s) with our own measurements, the lake depth has decreased by ~ 5 to 6 m, which contradicts point (a). Archaeological findings imply that the water level has been several meters below the present elevation for extended time periods during the past 1000 years (Milevski et al. 1997). While options (b) and (c) cannot be ruled out completely regarding the current level decline, Chavkalovski (1997) calculated that the combined irrigation water consumed by the three riparian countries since the mid 1960s could indeed lead to several meters decrease in lake level. He assumed consumption from Lake Prespa of 10 Mio m^3 yr^{-1} from

Macedonian territory, and a water uptake from Small Lake Prespa of 35 and 10 Mio m^3 yr^{-1} from Albanian and Greek territory, respectively.

Using equations (2.1 and 2.2) it is possible to extrapolate the underground outflow to the level of 1965 and to estimate the effect of an additional water outlet $Q_{out,add}$ on the water level. In such a scenario a former underground outflow of ~313 Mio m^3 yr^{-1} would be 68 Mio m^3 yr^{-1} higher than today. The consumption of 55 Mio m^3 for irrigation water each year would decrease the surface water level in Figure 2.2 by ~2 m in 10 years and by ~4.5 m until a new equilibrium is reached after several decades. This rudimentary analysis shows that

i) any additional water loss, artificial or natural, can lead to a drastic decline in the water level of Lake Prespa;
ii) the long-term underground outflow of Lake Prespa will be reduced by such an additional water loss;
iii) although the exact reasons for the observed decline are not known, irrigation practices are a plausible hypothesis.

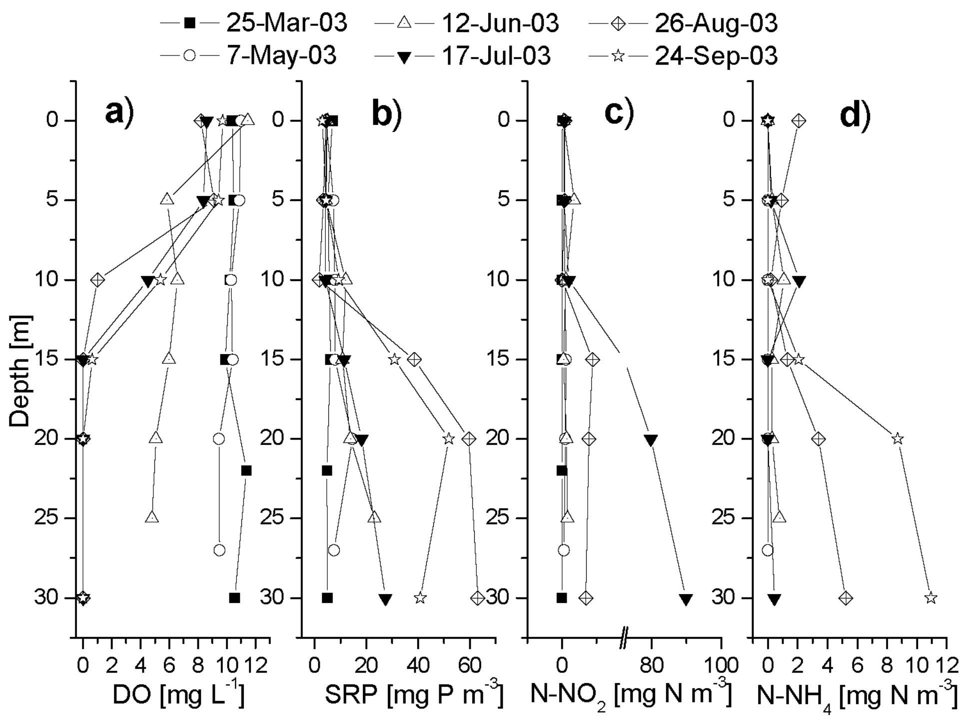

Figure 2.5: Development of (a) DO, (b) SRP, (c) NO_2^- and (d) NH_4^+ in the water column of Lake Prespa in the course of the productive season 2003.

2.4.2 Phosphorus Load from Lake Prespa

To assess the role of Lake Prespa in the eutrophication of Lake Ohrid, we need to know how the phosphorus loads from Lake Prespa have changed in the recent past and how they are likely to change in the near future. A key to these questions lies in the change of the trophic state of Lake Prespa. Several indicators of eutrophication in Lake Prespa are subsequently discussed.

Dissolved Oxygen (DO) – Figure 2.5a shows the development of the vertical distribution of DO in the course of one year for a deep site in the Macedonian part of Lake Prespa (Figure 2.1). Anoxic conditions prevail below 15 m from July to September 2003. This pattern has also been documented by others (Naumoski et al. 1997; Loeffler et al. 1998; Jordanoski et al. 2002). Along with the depletion of DO, the accumulation of mineralization products such as SRP, NO_2^- and NH_4^+ (Figures 2.5b to 2.5d) indicates a significant input of organic material to the lake sediment. Figure 2.6 compares historic data from the work of Jakovljevic (1935) with recent measurements of DO from the same location. The most striking difference appears in the summer profile (Figure 2.6c). In 1931, DO close to the lake bottom never dropped below 6 mg L^{-1}. Such a strong change over the past 70 years indicates that the observed summer anoxia is partly of recent anthropogenic origin.

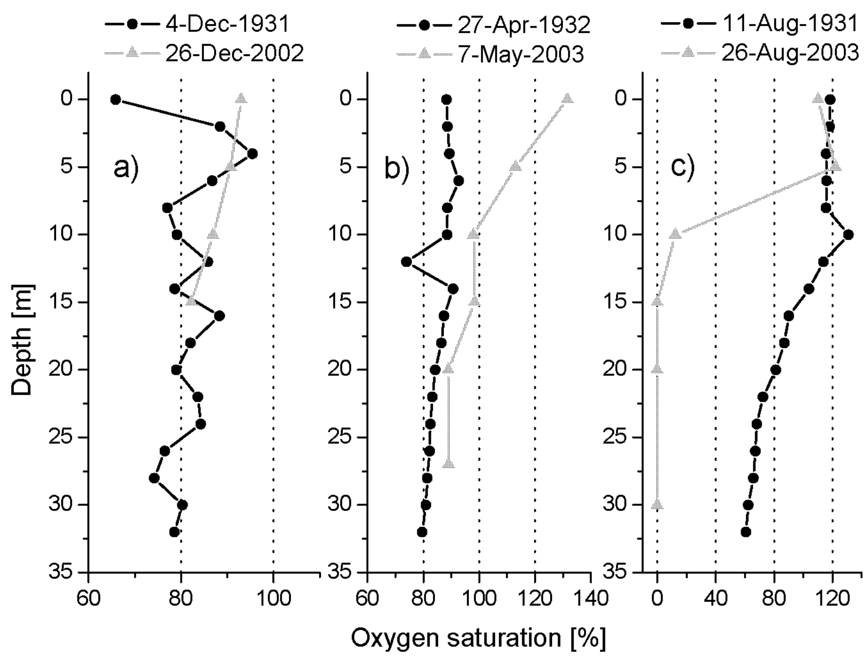

Figure 2.6: Comparison of oxygen saturation in profiles of Lake Prespa between historic results from Jakovljevic (1935) and recent measurements, for winter (a), spring (b) and summer (c). The surface point from December 1931 in Figure 2.6a is most probably a measurement error.

Phosphorus in the Water Column – As in Lake Ohrid, P is the growth-limiting element in Lake Prespa, as the molar N:P ratio in the top 10 m is about 25:1 on average, well above the Redfield ratio of 16:1 (Redfield 1958). The average concentration of TP in Lake Prespa was 31 mg P m^{-3} in 2003, which is consistent with findings by Jordanoski et al. (2002) for the years 2000 to 2002 (Figure 2.7). Naumoski et al. (1997) measured an average TP concentration of 20 mg P m^{-3} between January 1992 and January 1993, which signifies an increase of 11 mg P m^{-3} within 10 years. The concurrent 0.6 km^3 decrease in lake volume would explain a TP increase of 2.3 mg P m^{-3}, if we assume that the P-content of the lake and the P-input to the lake have both remained constant for this 10-year period. Thus an increase of 11 - 2.3 = 7.7 mg P m^{-3} or 0.8 mg P m^{-3} yr^{-1} must be caused by a raised P-load to Lake Prespa.

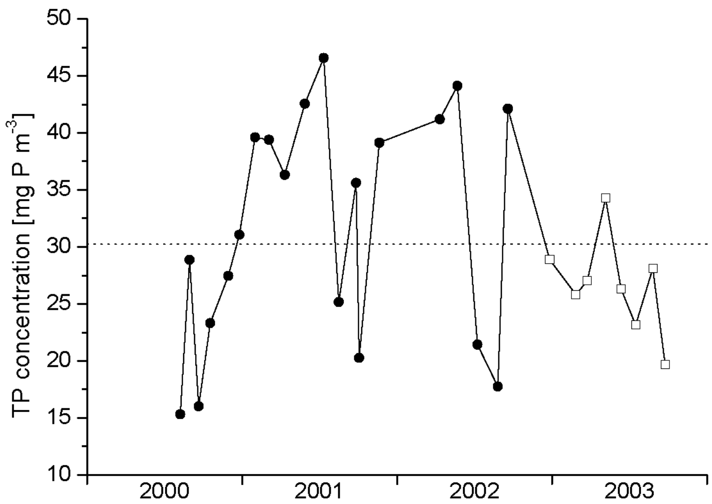

Figure 2.7: Seasonal development of average TP concentration in the top 15 m of Lake Prespa. The full circles are measurements by Jordanoski et al. (2002), the empty squares are our measurements. The horizontal line is the average concentration over the whole period.

Phosphorus in the Sediments – Another approach investigating the eutrophication history is provided by sediment cores. In a sediment core, taken at a depth of 20 m, an increase in the contents of TOC, TN and TP was observed over the top 12 cm (~150 years; Figure 2.8). Such an increase in organic components towards the sediment surface is expected as a result of early diagenesis (e.g., Meyers & Ishiwatary 1995; Hupfer et al. 1995). However, while mineralization stays at high rates for a few years only, nutrient release should be terminated after two decades at the most (Lotter et al. 1997; Urban et al. 1997). Thus, apart from the top few centimeters a real increase in TOC, TP and TN content has occurred. Nevertheless the reason for the increase is not straightforward. The most obvious explanations would be a change in allochthonous input or

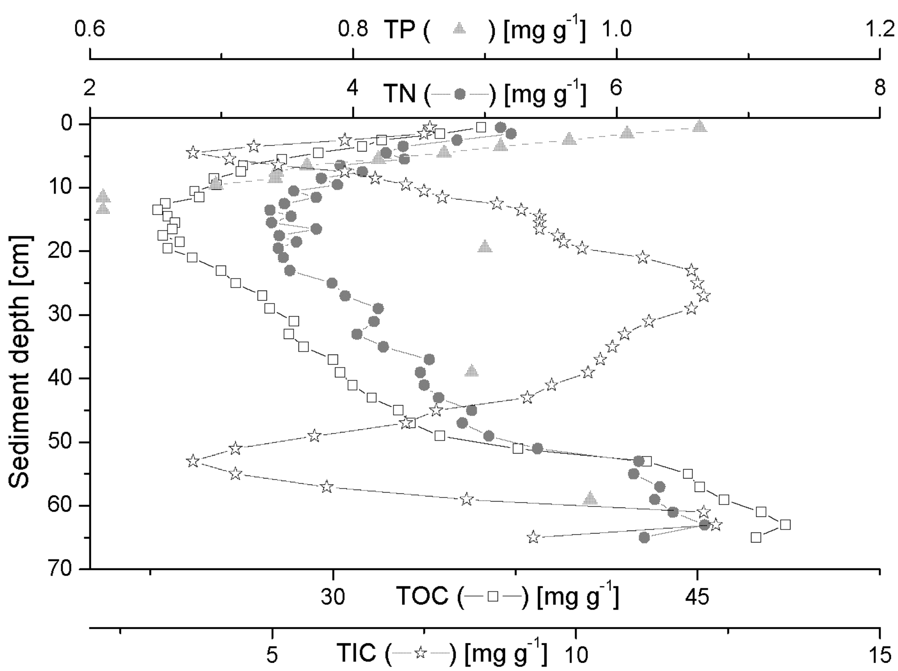

Figure 2.8: Content of TP, TN, TOC and TIC as measured in the sediment core from Lake Prespa, taken at 20 m depth in 2002 (Figure 2.1).

eutrophication. In the case of the latter a concurrent increase in calcite-precipitation, and thus a dilution effect, would be expected for Lake Prespa hard water (Dittrich & Koschel 2002). Indeed, such an effect can be seen in the measured TIC concentrations (Figure 2.8). On the other hand the observed increase in TIC supports the scenario of eutrophication. Furthermore the level fluctuations of Lake Prespa could have a strong influence on sedimentation. Net sedimentation rates were estimated with linear lake surface correction based on equation (2.6), using the lake surface area as in Figure 2.2 for 12 cm depth and the current surface area for the top of the core. The results show an increase over the past ~150 years from 46 to 72 t P yr^{-1} for TP, from 253 to 346 t N yr^{-1} for TN and from 1710 to 2440 t C yr^{-1} for TOC.

Surprisingly the contents of TOC, TN and TP increase with growing depth of the core below 20 cm (Figure 2.8). A possible reason is again water level fluctuation, which could change the mixing regime of the lake. At much lower lake level, benthic-pelagic coupling would be expected to be enhanced throughout the summer season, increasing the available nutrients and thus lake productivity. The increase in organic components in Figure 2.8 towards the bottom of the core might thus indicate much lower water levels several centuries ago. Archaeological findings of settlements from the 11th century in the lake, located below the present water level (Sibinovik 1987; Milevski et al. 1997), support such a scenario (Figure 2.2).

Lake Prespa Phosphorus Balance – In order to establish a phosphorus balance for Lake Prespa, the influence of the recent water level fluctuations on the terms V and τ in equation (2.8) have to be included. If we know the water level, TP concentration in the lake water and $P_{sed,net}$, P_{inp} can be estimated for equilibrium conditions based on equations (2.4 and 2.8).

Results of these calculations are shown in Table 2.4. In this table the historic situation, before the recent increase of TP in the sediment core (Figure 2.8), is compared to 2003. Because there is little historic data, the TP concentration in 1992 (Naumoski et al. 1997) was used for the historic conditions. Table 2.4 implies an increase in TP input from 52-70 t P yr^{-1} from 1992 to 2003 and a current net sedimentation rate of 60 t P yr^{-1}. This net sedimentation rate indicates that about 60 – 46 = 14 t P yr^{-1}, or about 50% of the observed 26 tP yr^{-1} increase of TP in the sediment core, is likely the result of the higher P input. It further implies that Lake Prespa is reducing the potential P loads to Lake Ohrid by $P_{sed,net}/P_{inp}$ ~86% (Table 2.4).

Results from equations (2.4 and 2.8) can be checked for plausibility with the observed internal balance. Figure 2.7 shows the seasonal P dynamics of Lake Prespa. During the productive season the average TP concentration decreases by ~28 mg P m^{-3} in the top 15 meters, which indicates loss by sedimentation. This loss corresponds to a gross sedimentation ($P_{sed,\,gross}$) of ~90 t P yr^{-1}. If the calculated net sedimentation $P_{sed,net}$ in Table 2.4 is correct, the fraction of the released P from sedimenting particles and from the sediment, $P_{rel} = P_{sed,\,gross} – P_{sed,\,net}$, equals 30 t P yr^{-1}. Such a release of phosphorus can be observed in an increase of SRP below the thermocline over the summer season (Figure 2.5b). By depth-integrating the SRP profiles below 10 m (equation 2.5), an increase of 31 t P was found from March to August/September 2003, which compares very well with the result of our internal balance. The ratio P_{rel} / $P_{sed,gross}$ would then be ~30%, which is within the expected range (Hupfer et al. 1995; Moosmann et al. 2006). The sum of the calculated input fluxes P_{inp} and P_{rel} is 101 t P yr^{-1}, which is very close to the sum of the output fluxes $P_{sed,gross}$ and P_{out} of 99 t P yr^{-1}.

Table 2.4: P balance of Lake Prespa and Lake Ohrid

P Balance[1]	Lake Prespa		Lake Ohrid
	"historic"[3]	2003	2003
Equilibrium P concentration [mg P m^{-3}][2]	*20*	*31*	*4.5*
Net sedimentation [t P yr^{-1}]	*46*	**60**	*38*
σ (sedimentation rate constant = net P sedimentation per total lake content) [yr^{-1}]	**0.5**	*0.5*	**0.15**
β (concentration in outflow divided by volume-average concentration) [-]	*1*	*1*	**0.77**
P-input P_{inp} [t P yr^{-1}]	**52**	**70**	**40**
P-outflow P_{out} [t P yr^{-1}]	**6**	**9**	**3.5**

[1] P balance of Lake Prespa based on equations (2.2, 2.4 and 2.8) and of Lake Ohrid based on equation (2.8). Input data are in italics, results are in bold.
[2] Equilibrium P concentration is the annual average of mean lake concentrations (equation 2.5).
[3] For the historic situation of Lake Prespa the minimum P sedimentation rate and the 1992 TP concentration by Naumoski et al. (1997) were used.

Summary **–** Both the development of anoxic conditions in the bottom water and the increases in sedimentary P, as well as the linear lake P balance, point towards a eutrophication of Lake Prespa. This development is most probably the result of a combination of intensified agriculture (Chavkalovski 1997; Hollis & Stevenson 1997), lack of sewage treatment and increased use of P-containing detergents. However, the increase of P input cannot be linked directly to the population, which has decreased in the past 50 years (Berxholi 1997; Catsadorakis & Malakou 1997; Macedonian State Statistical Institute, personal communication 2003). Apart from the increased P loads, lake level changes can have a large influence on the water quality. Changes in the lake volume have a direct effect on the concentration of dissolved nutrients. If indeed the lake surface was much lower historically, P concentrations may have been even higher than today.

This leaves us with two main conclusions: (i) Lake Prespa is in a process of eutrophication and (ii) the water level of Lake Prespa has a large effect on its water quality. A further level drop could surpass the effect from increased nutrient input.

2.4.3 Transfer to Lake Ohrid

Having established a model for calculating the phosphorus load from Lake Prespa, we focus now on the fraction of P that is transferred to Lake Ohrid.

Water Transfer **–** Within a current IAEA project (Anovski 2001) colour tracer experiments have revealed that two large springs at the south-eastern end of Lake Ohrid have significant connections to Lake Prespa (Zoto, pers. comm. 2003). These two sources provide a total inflow of ~10 m^3 s^{-1} (Watzin et al. 2002). Part of this flow may not originate from Lake Prespa but instead from local precipitation seeping through the karst rocks and mixing with the Prespa water that flows downwards into Lake Ohrid. Several studies have been undertaken on the hydraulic connection between the two lakes using stable isotopes (Anovski et al. 1980; Anovski et al. 1992; Eftimi & Zoto 1997; Eftimi et al. 2001). The results for δD and $\delta^{18}O$ from Anovski et al. (1980) and Eftimi & Zoto (1997) are plotted in Figure 2.9, together with our own measurements from 2001 to 2003. Despite the dynamics of the underground system, the values have been surprisingly constant over more than two decades.

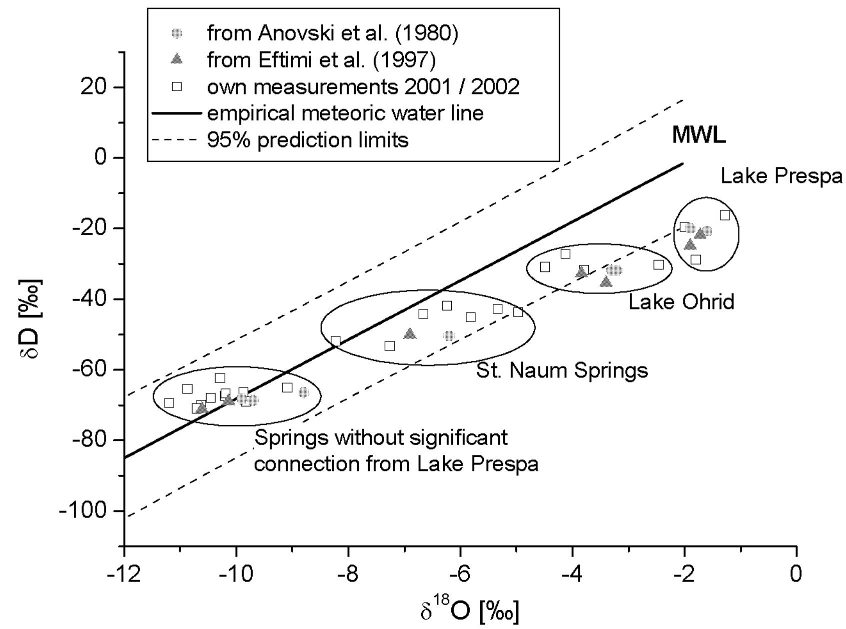

Figure 2.9: Ratios of stable isotopes δD and $\delta^{18}O$ in the two lakes, St. Naum Springs with a proven connection from Lake Prespa and various springs fed by precipitation only. The meteoric water line (MWL) is based on linear regression of local precipitation measurements by us and Anovski et al. (1980).

Lake Prespa and Lake Ohrid are enriched in heavy isotopes due to evaporative processes. Springs fed from precipitation are evenly distributed between the prediction bands of the empirical meteoric water line (MWL), which is based on local rain samples by Anovski et al. (1980) and by ourselves (Table 2.2, Figure 2.9). St. Naum Springs, the largest spring area, which is partly charged by Lake Prespa, is also enriched in heavy isotopes compared to the MWL, which further confirms that parts of its waters have been subject to evaporation.

The measurement of stable isotopes has been used to estimate the ratio of water that originates from Lake Prespa emerging at St. Naum (Figure 2.1) and of rainwater percolating through the karst rocks, based on equation (2.9). A similar balance can be done for all substances which are transported by underground water. In Table 2.5, the results for the stable isotopes δD and $\delta^{18}O$, several ions and TP are shown. The calculations are based on 5 to 30 measurements per site from the years 2001 - 2003 (Table 2.2). Measurements from the

Table 2.5: Water composition at St. Naum Springs

Substance	Share from Lake Prespa[1] [%]	Share from precipitation[1] [%]	Standard deviation [%]
$\delta^{18}O$	47	53	6
δD	45	56	4
Cl^-	35	65	8
SO_4^{2-}	12	88	4
Na^+	27	73	6
K^+	30	70	4
Ca^{2+}	56	44	49
Mg^{2+}	68	32	3
SRP	32	68	14
TP	11	89	5

[1] Ratios calculated, based on measured concentrations in Lake Prespa, in precipitation-fed springs and in St. Naum Springs using equation (2.9).

top 15 meters of Lake Prespa were included without volume-averaging, because no distinct profiles were observed and the position of the outlets is unknown. Most of the water constituents, listed in Table 2.5 (e.g., SO_4^{2-} or SRP), are not conservative since they can be influenced by biological processes or by ion exchange, as most cations. Thus, only stable isotopes and Cl^- were used to establish an error-weighted average of the mixing ratio. The result implies that 43 ± 5% of the spring water at St. Naum is fed from Lake Prespa and 57 ± 5% from local precipitation.

Table 2.6: Water flow from Lake Prespa in two spring areas

Spring area	Average discharge[1] [$m^3 s^{-1}$]	Share from Lake Prespa[2] [%]	Discharge from Lake Prespa [$m^3 s^{-1}$]
St. Naum Springs (MK)	7.5	43	3.2
Tushemishti Springs (AL)	2.5	52	1.3
Total	10	45	4.5

[1] from Watzin et al. (2002)
[2] from own measurements for St. Naum Springs and from Eftimi and Zoto (1997) for the Tushemishti Springs

Eftimi & Zoto (1997) found similar numbers for St. Naum Springs and a slightly increased share of Lake Prespa water in the Tushemishti Springs in Albania (Figure 2.1). In Table 2.6 the percentages of Lake Prespa water in St. Naum Springs from our own findings and in Tushemishti Springs from Eftimi & Zoto (1997) are multiplied with their respective inflows, presented by Watzin et al. (2002). In total, the two spring areas were found to contribute about 4.5 $m^3 s^{-1}$ of Lake Prespa water to Lake Ohrid. This corresponds to 58% of the total underground outflow from Lake Prespa (245 Mio $m^3 yr^{-1}$ = 7.8 $m^3 s^{-1}$) and leaves about 7.8 - 4.5 = 2.3 $m^3 s^{-1}$ (100 Mio $m^3 yr^{-1}$) unidentified (Tables 2.3 and 2.6). Although there are several small springs to the south of Lake Ohrid, outside of its catchment area, Eftimi & Zoto (1997) found that these springs are mainly fed from local precipitation. However, we have located subaquatic springs below the surface of Lake Ohrid along its eastern shoreline with CTD transects. Moreover, the water balance of Lake Ohrid indicates that as much as 10 $m^3 s^{-1}$ are expected from subaquatic springs (Matzinger et al. 2004). As a result, it will be assumed in the following that the total underground outflow of 7.8 $m^3 s^{-1}$ from Lake Prespa flows to Lake Ohrid.

Phosphorus Transfer – For assessing the contribution of this underground connection to the eutrophication of Lake Ohrid, the transfer of phosphorus from Lake Prespa to Lake Ohrid needs to be quantified. As for the determination of the water origin, we distinguish between springs in the catchment of Lake Ohrid with and without a proven connection to Lake Prespa. For the latter an average TP concentration of 4 ± 1 mg P m^{-3} was found based on our sampling (Table 2.2, Figure 2.10). According to this value and the water loads estimated from stable isotope and chloride measurements (Tables 2.5 and 2.6) a TP concentration of 11 mg P m^{-3} is calculated for the Lake Prespa water of the St. Naum Springs, which is significantly lower than in the lake itself. Based on the oxygen saturation in the spring water arising on the Lake Ohrid side we presume that organic matter is exposed to oxic mineralization during underground transport. Accordingly, virtually all P that reaches Lake Ohrid through underground transport is bio-available, which is also indicated by the insignificant difference between measured SRP and TP spring concentrations, based on a two sample paired t-test at level 0.05 (Figure 2.10). Lake Prespa outflow obviously passes a coarse filter, such as a sediment layer, which retains most particles. Within such a layer P mineralization and incorporation or sorption of SRP seem probable (Gächter et al.

2004). Moreover, SRP can be adsorbed to mineral surfaces such as Fe-oxyhydroxides, silicates or Ca on its way through the underground (Diaz et al. 1994; Ioannou & Dimirkou 1997; Dittrich & Koschel 2002).

We conclude that 65 % of the TP leaving Lake Prespa (average of 31 mg P m^{-3}) entering into the karst underground is retained. We assume this ratio to be constant for higher P concentrations as well. This seems plausible for filtering as well as for adsorption processes, since any solubility product between PO_4 and Ca species will be dominated by Ca (Ca:P > 1000).

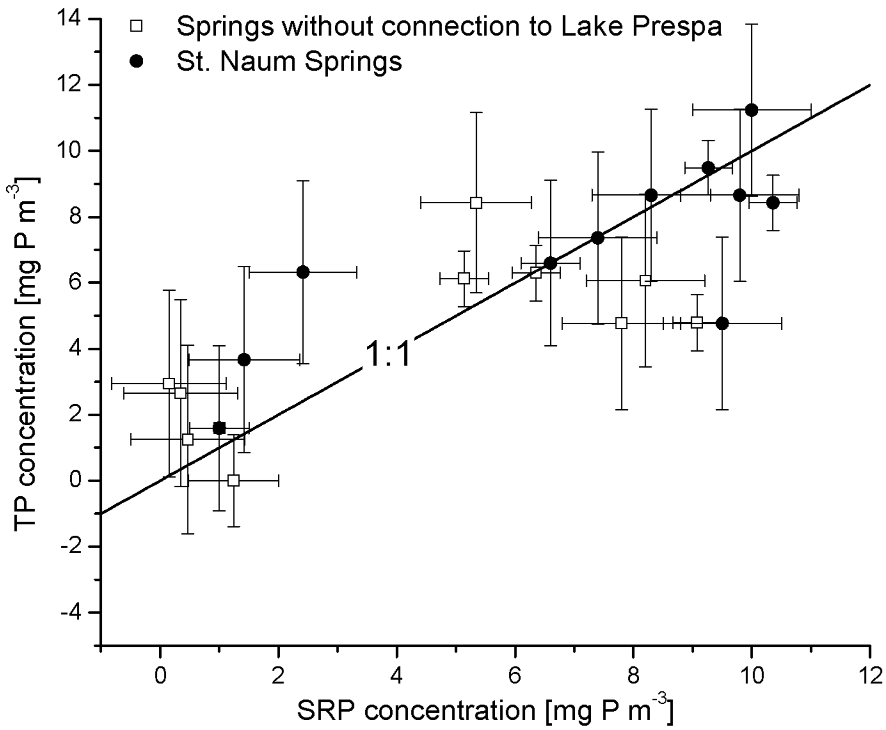

Figure 2.10: SRP and TP concentrations measured in St. Naum Springs and in springs without connection from Lake Prespa. Error bars show the respective measurement errors.

2.4.4 Effect on Lake Ohrid

Given the calculations above, about 20% of the Lake Ohrid water inflow and 7% of the TP load originate from Lake Prespa. The recent rapid water level decline of Lake Prespa most likely is the result of increased irrigation or net evaporation, rather than of an opening of additional underground channels. Hence, the water level drop most likely is reducing the underground flow. Such a decrease has no influence on the water level of Lake Ohrid. However, its outflow is reduced, and consequently the hydraulic residence time τ of Lake Ohrid increases. Based on equation (2.2), the underground flow can be calculated for different water levels (Figure 2.11). The decline of the Lake Prespa water level since the 1960s has reduced the annual Lake Ohrid inflow by ΔQ = 68 Mio $m^3\ yr^{-1}$. This decrease in inflow results in a 9% increase in the hydraulic residence time of Lake Ohrid ($1/\tau_{new} = 1/\tau_{old} - \Delta Q/V$). Based on our measured TP concentration in the water column of 4.5 mg P m^{-3} and a net P sedimentation of 38 t P yr^{-1} in Lake Ohrid (Table 2.2), we found the parameters $\sigma = 0.15\ yr^{-1}$ and $\beta = 0.77$ for equation (2.8) (Table 2.4). Using the linear P model of equation (2.8) with those parameters, a smaller water exchange leads to a P residence time scale τ^* of 6.06 yr, which corresponds to an increase in the TP concentration of Lake Ohrid by merely ~0.6 %.

However, the P load P_{inp} to Lake Ohrid does not remain constant, but changes with decreasing Lake Prespa water level as a result of three competing causes (compare equation 2.8). (i) A smaller subterranean water flow $Q_{out,undgr}$ (equation 2.2) consequently reduces P_{inp}, (ii) the decreasing Lake Prespa volume leads to an increase in P concentra-

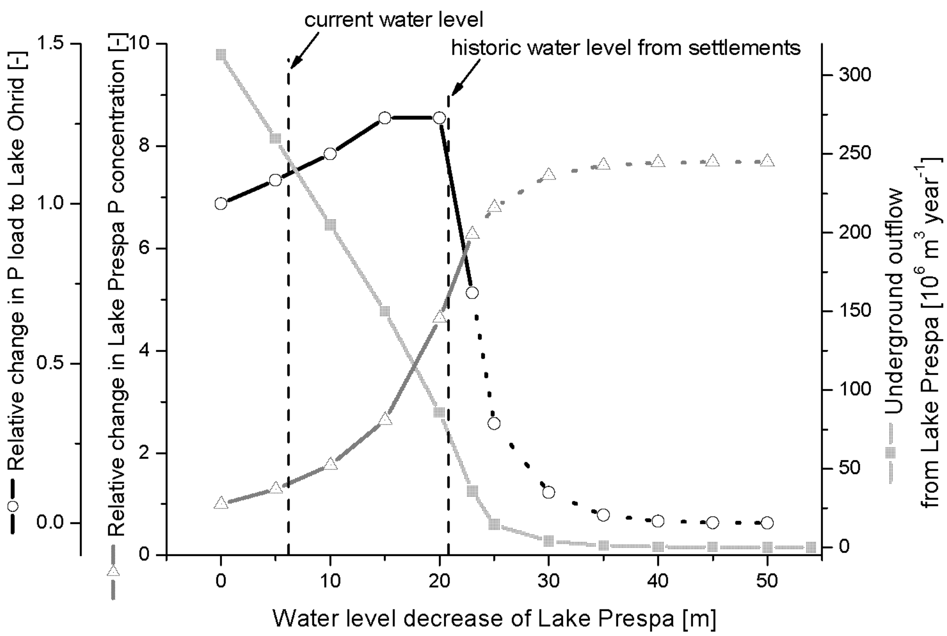

Figure 2.11: Effect of water level decline of Lake Prespa on its underground outflow, its P concentration and the P load to Lake Ohrid based on equations (2.1, 2.2 and 2.8). The P load to Lake Ohrid is proportional to underground outflow times Lake Prespa TP concentration. Note that the figure is only valid if water is extracted from Lake Prespa, but not if more water flows to Lake Ohrid. The dotted lines indicate that the use of equations (2.2) and (2.3) may no longer be appropriate.

tion in $Q_{out,undgr}$ and finally (iii) a shorter hydraulic residence time of Lake Prespa decreases this concentration. The last point is valid as long as σ in equation (2.8) remains constant. This estimate of σ was shown to be reasonable within the lake level range of the past decade and can be expected to provide sensible results under a further level decline. However, it is not known how a dramatic decrease to an empty lake would influence the phosphorus transport to Lake Ohrid. In Figure 2.11 the three effects (i) to (iii) on the phosphorus transport to Lake Ohrid are summarized using equations (2.2, 2.4 and 2.8). For a 20 m decrease in the water level, the P concentration in Lake Prespa increases almost five-fold, which leads to a 30% increase of the phosphorus load from Lake Prespa to Lake Ohrid, although the groundwater outflow decreases. With further level decline the predicted concentration in Lake Prespa goes up to 160 mg P m^{-3} but, as the groundwater flow declines rapidly, the P load to Lake Ohrid is also reduced (Figure 2.11). This decline in groundwater flow occurs only under the assumption of an additional outlet (or a reduced inflow) on the Lake Prespa side, such as water abstraction for irrigation or increased evaporation. In the case of an increased underground outflow to Lake Ohrid, the load would simply increase with the concentration in Lake Prespa.

Apart from the effects of the level decrease, the observed increase in P loads to Lake Prespa has to be taken into account. Adding both effects we find that the P loads to Lake Ohrid have increased by about 50% since the 1960s. Based on equation (2.8), this combined increase in the P load from Lake Prespa will augment the TP concentration of Lake Ohrid by merely 0.1 mg P m^{-3}. Even if we assume that the P load from Lake Prespa increases four times, the total increase of the TP concentration in Lake Ohrid would still be below 1 mg P m^{-3}. The transferred P is predominantly in readily bio-available form, which might amplify its effect on lake productivity. Still, the eutrophication potential of the underground connection between Lake Prespa and Lake Ohrid seems limited, thanks to the purifying effect of the underground. However, in historic times, when 90% of the inflow to Lake Ohrid was from direct precipitation and the karst aquifers, the SRP input from the latter could have been the major P source.

2.5 Conclusions

2.5.1 Lake Ohrid

Presently, Lake Prespa does not pose an immediate threat on Lake Ohrid, despite seven times higher TP concentrations and despite the large share of the Lake Ohrid catchment. The karst aquifers connecting the two lakes not only mineralize the entering phosphorus to SRP but also retain 65% of the P load from Lake Prespa. As a result, even a fourfold increase of the P load from Lake Prespa would lead to only 20 % increase (+ 0.9 mg P m^{-3}) of the actual 4.5 mg P m^{-3} TP concentration of Lake Ohrid. However, in ultraoligotrophic Lake Ohrid, an increase by 0.9 mg P m^{-3} might have significant effects on its fragile endemic community (Watzin et al. 2002). Moreover, the eutrophication potential of the spring inflow is increased, since almost all the P arrives as bio-available SRP.

Historically, the P load from Lake Prespa could have been important for the formation of the unique aquatic ecosystem of Lake Ohrid, due to the constant supply in directly bio-available SRP. This importance is indicated by specialized endemic phytoplankton species (e.g., *Cyclotella fottii*), which are dominant between 20 and 50 m depth (Ocevski & Allen 1977; Mitic 1985; Patceva 2001), exactly where the karst spring water intrudes during the productive summer season. This endemic community could change drastically with a substantial increase in the spring water P load.

2.5.2 Lake Prespa

The P balances above reveal that Lake Prespa would need to become eutrophic or even hypertrophic before it affects Lake Ohrid significantly. In fact, Lake Prespa is a very vulnerable system, because any additional consumption of water has a direct effect on its water level, which in turn affects not only the lake hydraulics but the entire lake ecosystem. A level decrease alone can cause an increase in the trophic state of Lake Prespa. If the external P loads increase simultaneously, the two combined processes can amplify. Such amplification is a realistic scenario in the case of further intensification of agriculture, where water consumption and fertilization increase in parallel. The recently observed anoxia in the deeper layers of the lake will most probably have a significant effect on its biodiversity.

2.5.3 Future Stakes

In the future, P and DO in the water column as well as the water level of Lake Prespa must be monitored on a regular basis, as they are essential indicators for the lake ecology. Moreover, the implications of these abiotic changes in Lake Prespa on its flora and fauna should be examined. Regarding Lake Ohrid, occasional measurements of P concentrations in the springs flowing to Lake Ohrid would help quantify changes in the P transfer from Lake Prespa. The main anthropogenic factor influencing Lake Prespa is agricultural development and in particular the abstraction of water, application of fertilizers, and enhanced soil erosion. Thus, it is important that the agricultural practices in all three riparian states are assessed in detail and management options to control the water use and the application of fertilizers are proposed.

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Chapter 3

Sensitivity of Ancient Lake Ohrid to Local Anthropogenic Impacts and Global Warming

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Abstract

Human impacts on the few ancient lakes of the world must be assessed, as any change can lead to an irreversible loss of endemic communities. In such an assessment, the sensitivity of Lake Ohrid (Macedonia/Albania; surface area A = 358 km^2, volume V = 55 km^3, > 200 endemic species) to three major human impacts – water abstraction, eutrophication and global warming – is evaluated.

It is shown that ongoing eutrophication presents the major threat to this unique lake system, even under the conservative assumption of an increase in phosphorus (P) concentration from the current 4.5 to a potential future 9 mg P m^{-3}. Eutrophication would lead to a significant reduction in light penetration, which is a prerequisite for endemic, deep living plankton communities. Moreover, a P increase to 9 mg P m^{-3} would create deep water anoxia through elevated oxygen consumption and increase in the water column stability due to more mineralization of organic material. Such anoxic conditions would severely threaten the endemic bottom fauna. The trend towards anoxia is further amplified by the predicted global warming of 0.04 °C yr^{-1}, which significantly reduces the frequency of complete seasonal deep convective mixing compared to the current warming of 0.006 °C yr^{-1}. This reduction in deep water

exchange is triggered by the warming process rather than by overall higher temperatures in the lake. In contrast, deep convective mixing would be even more frequent than today under a higher temperature equilibrium, as a result of the temperature dependence of the thermal expansivity of water. Although water abstraction may change local habitats, e.g., karst spring areas, its effects on overall lake properties was shown to be of minor importance.

3.1 Introduction

The following paper assesses the effects of three major human impacts – eutrophication, water abstraction, global warming – on the physical properties of Lake Ohrid (Table 3.1). Particular emphasis is given to the possible interaction between the local and global human impacts, as well as their relative importance. The analysis focuses on properties that form physical habitats for the endemic species of the lake.

Lake Ohrid, located between Macedonia and Albania (Table 3.1, Figure 3.1), is among the few ancient, long-lived lakes of the world that have provided continuous freshwater habitats for more than one million years, and the only one in Europe (Gorthner 1994; Martens et al. 1994; Meybeck 1995). It harbors a large number of relict and endemic species, which makes it also a hotspot of freshwater diversity (UN: World Conservation Monitoring Centre 1998; LakeNet: Duker and Borre 2001). Its importance was further underlined by UNESCO, by declaring it as a World Heritage site (UNESCO, 1979).

While Lake Ohrid is currently oligotrophic and not in immediate danger, it is being jeopardized by human activities (Watzin et al. 2002). In the two riparian countries, which are both in political transition, agriculture is still a major source of economy, putting pressure on surface water for irrigation given the semi-arid climate of the area (European Environment Agency 2003). The lack of economic growth also leads to pollution from sewage, because of missing or ineffective wastewater treatment (Ernst Basler and Partners 1995). The local human impacts are aggravated by an ongoing growth in population within the lake catchment (Albania: Watzin et al. 2002; Macedonia: Macedonian State Institute for Statistics, personal communication 2004). Moreover, regional simulations of global warming predict changes above average for south-

Table 3.1: Characteristics of Lake Ohrid

Property	Unit	Value
Latitude[1]	° N	41.1
Longitude[1]	° E	20.8
Altitude[1]	m asl	693.7
Catchment area[2]	km^2	2600
Surface area[3]	km^2	358
Volume[3]	km^3	54.9
Maximal depth[3]	m	288.7
Average depth[3]	m	155
Hydraulic water residence time[3,4]	yr	70
Age of lake existence[5]	10^6 yr	2 - 3
Endemic species[5]	#	> 200
Average phosphorus concentration[6]	mg-P m^{-3}	4.5

[1] Naumoski (2001)
[2] Including Lake Ohrid tributaries (Watzin et al. 2002), as well as Lake Prespa and its catchment (Anovski et al. 2001)
[3] Based on bathymetric data measured by the Smithsonian Institution (unpublished data, 1975)
[4] Macedonian Hydrometeorological Institute (unpublished data)
[5] Stankovic (1960)
[6] Own measurements (2002-2004)

eastern Europe (Giorgi et al. 2004; Räisänen et al. 2004). Finally the large depth of Lake Ohrid (Table 3.1) makes it prone to prolonged deep water isolation due to temperature increase (IPCC 2002; O'Reilly et al. 2003).

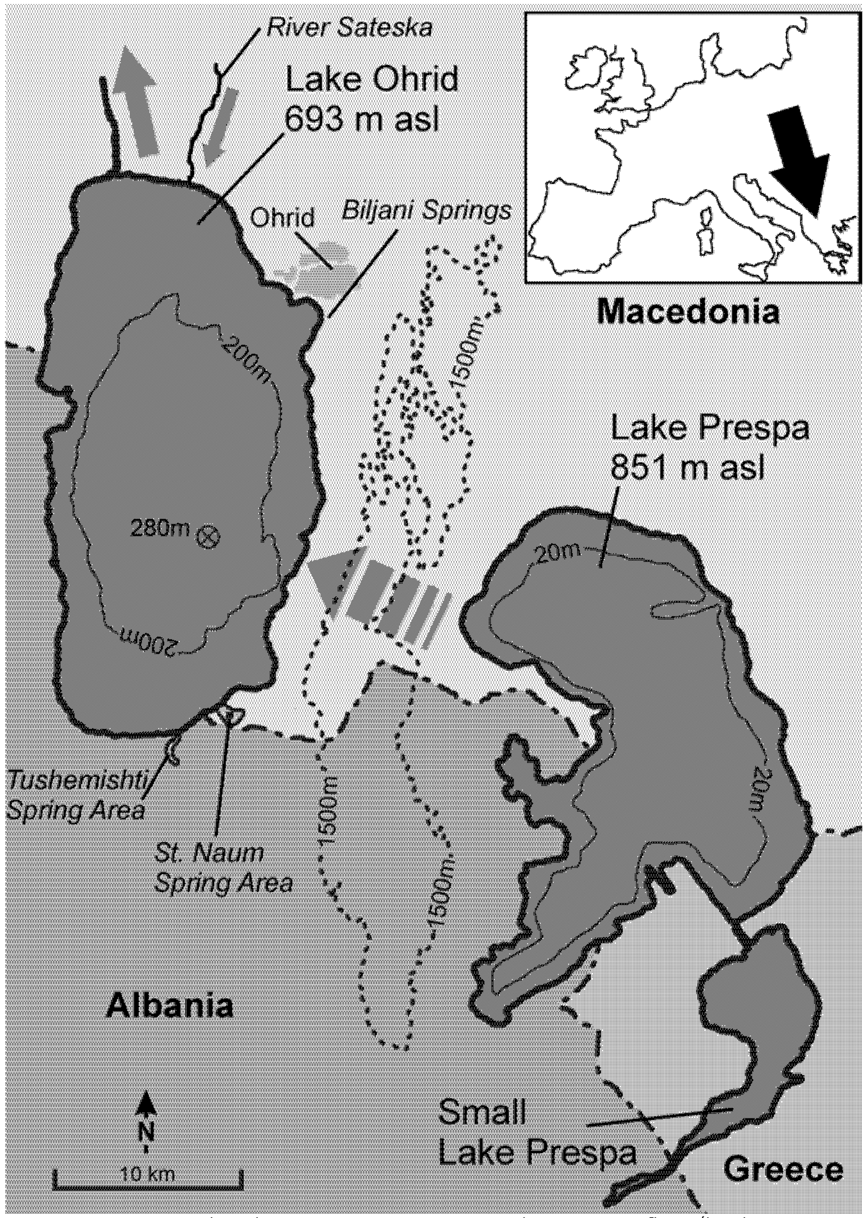

Figure 3.1: Geographical overview. Grey arrows show water flow (broken arrow = underground flow), crossed circle is main sampling site, inset map indicates location of study area in Europe.

The following case study provides an example of a system-analytical assessment, applicable to many other lakes that are exposed to similar threats. Indeed, pollution and scarcity of freshwater resources, followed by climate change, have been identified as the major environmental concerns of the future (UNEP 1999). Several recent publications have addressed the effect of these three environmental threats – pollution, water abstraction and climate change – on physical lake processes (e.g., Goudsmit et al. 2002; Sirjacobs et al. 2004; Lehmann 2002). Anthropogenically-induced changes in the physical lake properties have in turn been described to affect lake ecology (Hecky et al. 1994; O'Reilly et al. 2003) as well as economy (Lofgren et al. 2002). While some of the observed or predicted changes in lakes have been shown to be significant (Beeton 2002), comparison of impacts and interactions among them are often neglected. Impact assessment of these human pressures deserves special attention for the few globally unique ancient lakes, as any change might lead to irreversible losses (Cohen 1994; Beeton 2002).

Summarizing the above, there are three main motivations for our work:

(a) Lake Ohrid is an ideal site to study the influence and interaction of the three human impacts, water abstraction, eutrophication and climate change on physical lake properties.

(b) Its endemic communities are threatened by potential changes in the physical boundary conditions.

(c) In view of its international importance, the lake system is poorly documented, especially in terms of physical processes. A description of its status quo is required as a reference for future evaluations of changes.

3.2 Approach

The main management goal for Lake Ohrid is to conserve its rich and extensive endemism. We here base our analysis on properties, which form the physical habitat for two important groups of endemic species, phytoplankton and bottom fauna.

3.2.1 Phytoplankton

During the main productive season from April to October vertical phytoplankton distribution in Lake Ohrid is quite typical for oligotrophic lakes with green algae (*Chlorophyta*, *Chrysophyta*, *Pyrrophyta*) dominating the top 10 m of the water column and small forms of *Cyanophyta* taking over between 10 and 30 m (Patceva 2001; Kalff 2002). However the lake harbors highly specialized forms of pelagic diatoms (e.g. *Cyclotella fottii*), which show major growth between 20 and 50 m depth and are clearly dominant between 40 and 150 m (Stankovic 1960; Mitic 1985; Patceva 2005). In the oligotrophic lake (Table 3.1) high competition for bio-available soluble reactive phosphorus (SRP) is expected (N:P > 25:1, Matzinger et al. 2006a). Thanks to the efficient light use and/or the large size (> 1 mm) of endemic phytoplankton species (Mitic 1985), they are able to populate deep layers where most nutrients are available. This is also indicated by the shift of their maximal density to a depth of ~75 m from late autumn to early spring when the water has its highest clarity (Patceva 2005). Two criteria emerge: first the endemic diatoms depend on good light conditions at large depths and thus on high water transparency; and second, they are dependent on the supply of SRP between 20 and 150 m depth.

3.2.2 Bottom Fauna

About 90% of the endemic fauna described in Lake Ohrid are benthic organisms (Stankovic 1960). While some are found throughout the lake, most of them are limited to a specific depth zone. A surprising number of species is even restricted to large depth exceeding 150 m, such as some forms of gastropods, amphipods and ostracods (Stankovic 1960). Other species, such as the peculiar sponges *Ochridaspongia rotunda* and *Ochridaspongia interlithonis* live in deep water or at shallow sites in the vicinity of subaquatic spring inflows (Arndt 1937; Gilbert and Hadzisce 1984). Both areas are defined by cool and oxygen-rich water. Given the oligotrophic conditions all the endemic forms favor oxygen-rich conditions and are expected to be vulnerable to sediment anoxia.

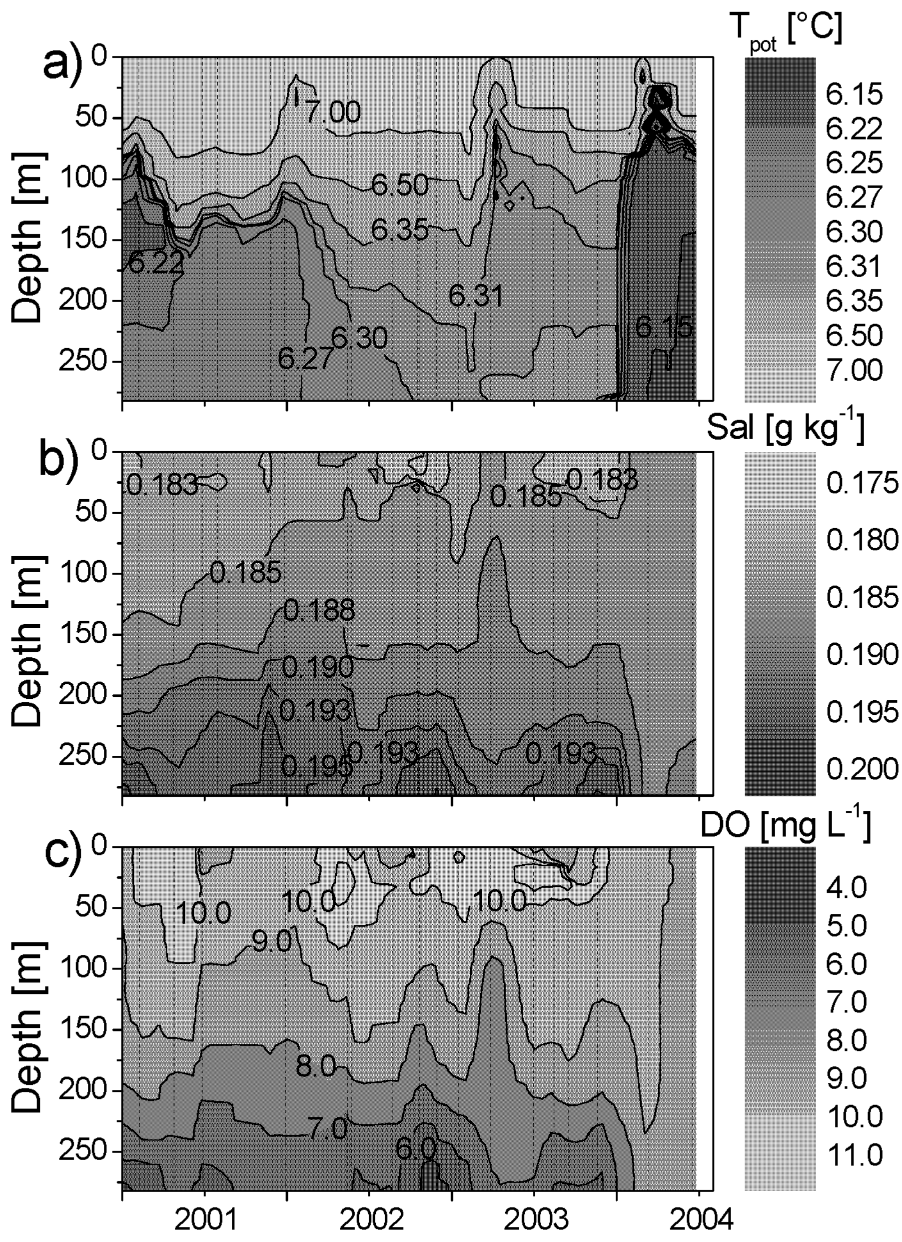

Figure 3.2: Contours from vertical profiles over the period 2001-2004. Dotted lines indicate dates of vertical profiles. (a) Temperature measured with a CTD probe, corrected for the surface level. (b) CTD conductivity transformed into salinity based on the average salt composition of Lake Ohrid. (c) Dissolved oxygen (DO) concentration from CTD measurements. Each DO profile is calibrated with a profile of Winkler samples.

As a result we assessed the effects of the three major human impacts – eutrophication, water abstraction, global warming – on (i) SRP supply between 20 and 150 m depth, (ii) water clarity in the trophogenic layer and (iii) oxygen supply to the hypolimnion.

The paper starts with a presentation of the status quo of physical properties of Lake Ohrid, which are important to understand the changes from human impacts. This description is based on basic parameters from regular CTD casts (Figure 3.2, Seabird SBE 19), thermistor measurements (Vemco and RBR Ltd.) and water samples for dissolved oxygen (DO) from 1999 – 2004. Following the physical lake characterization, human impacts are defined for historic, current and potential future situations. Finally, based on the physical properties, the effect of the defined anthropogenic changes is assessed.

3.3 Physical Characteristics of Lake Ohrid – Status Quo

3.3.1 Hydrology

Water Balance – The relatively dry, Mediterranean climate and the small drainage basin (catchment/lake surface ratio of ~7) results in a long hydraulic residence time scale of ~70 yr (Table 3.1). The water balance is dominated by the inflow from karst aquifers (~50%) with smaller shares from rivers and direct precipitation (Table 3.2). The river runoff was even lower by 5.5 $m^3\ s^{-1}$ (~70%), before 1962 when River Sateska was deliberately diverted into the lake from the north (Figure 3.1). The karst aquifers are charged from mountain range precipitation and from Lake Prespa, as revealed by using stable isotopes and Cl^- as natural tracers (Figure 3.1; Anovski et al. 1992; Eftimi and Zoto 1997; Matzinger et al. 2006a) and dye tracer experiments (J. Zoto, personal communication 2003). Apart from two large spring areas, which flow directly into Lake Ohrid on its southern shore (Figure 3.1), about 50% of the aquifers are expected to be sublacustrine (Table 3.2). It is noteworthy that 50% of the already small catchment area is made up of the underground connection to Lake Prespa (Figure 3.1). The water leaves Lake Ohrid by surface outflow in the north (~60%) and

Table 3.2: Water Balance of Lake Ohrid

Outputs	**Flow rate [$m^3\ s^{-1}$]**
Surface outflow River Crn Drim[1,2]	24.9
Evaporation (1145 mm yr^{-1})[3]	13.0
Total output	37.9
Inputs	**Flow rate [$m^3\ s^{-1}$]**
Precipitation on lake surface (773 mm yr^{-1})[3]	8.8
River inflows:	
Albanian catchment[3]	0.5
Macedonian catchment without R. Sateska[4,5]	1.9
R. Sateska (diversion into Lake Ohrid in 1962)[1,2,4]	5.5
Temporary inflows (estimation)[6]	1.0
Surface spring inflows[3]	10.3
Sublacustrine springs (from closing the balance)	9.9
Total input	37.9

[1] Ivanova (1974)
[2] Macedonian Hydrometeorological Institute (unpublished data, 2001)
[3] Watzin et al. (2002)
[4] Hydrobiological Institute Ohrid (unpublished data)
[5] Own measurements (2002 - 2003)
[6] Several small streams only develop after rain storms

by evaporation (~40%). Annual evaporation from the lake surface exceeds direct precipitation (Table 3.2).

Riverine and Spring Water Intrusions – Most of the tributaries - given their relatively small discharge of usually less than 1 $m^3 s^{-1}$ – are entrained directly into the surface water near the river inlets. During summer seasons even River Sateska, which contributes up to 13 $m^3 s^{-1}$ after continuous rain in winter and spring, is no more than a trickling creek, because of the dry climate and extensive upstream irrigation (Figure 3.3a). However during the cold season River Sateska can plunge deep due to its higher salinity and lower temperature (Figure 3.3b).

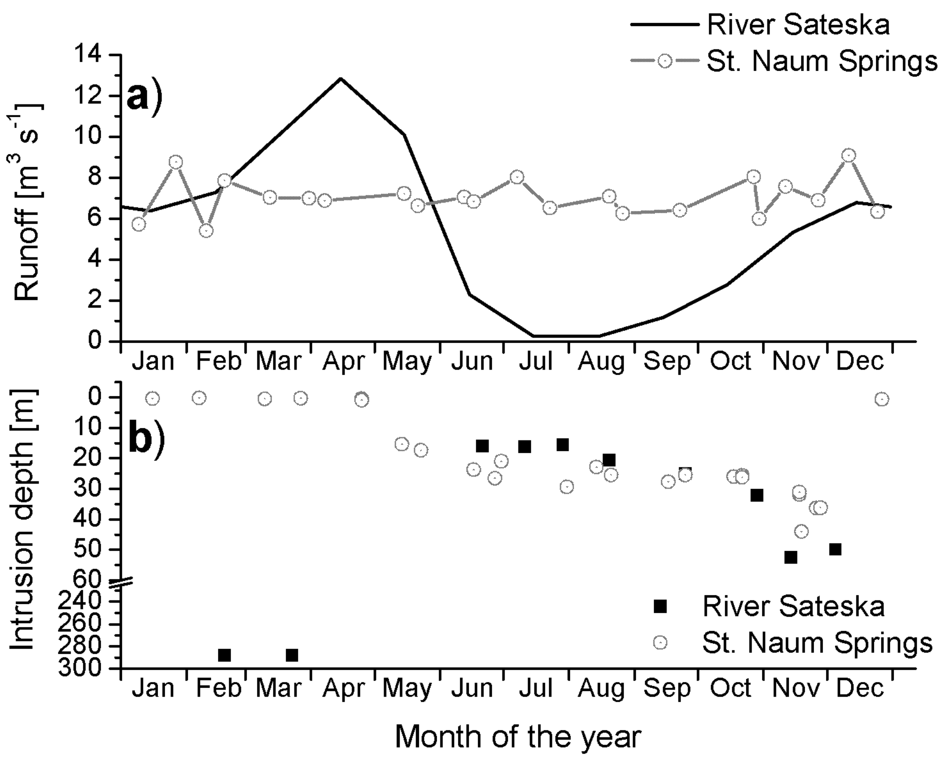

Figure 3.3: Seasonal discharge dynamics of two major inflows to Lake Ohrid. (a) Runoff based on combined measurements from 1996, 1997, 2002 and 2003 for St. Naum Springs (Hydrobiological Institute, unpublished data) and data from 1949-1969 (Ivanova 1974) and 1996-2000 (Hydrometeorological Institute, unpublished data) in combination with single measurements directly at lake inflow (Hydrobiological Institute, unpublished data) for River Sateska. (b) Calculated intrusion depth based on CTD profiles for the lake and salinity and temperature measurements from the inflows.

The inflow from the numerous karst springs seems very constant throughout the seasons. The surface spring area of St. Naum (Figure 3.1) shows standard deviations of the flow of only ~10% (Figure 3.3a), indicating significant storage capacity within the karst mountain. Though water property varies among different springs (e.g., temperature range ~9 to 12 °C), each single spring area shows remarkably constant parameters. Monthly measurements in springs, one in the north-east (Biljani Spring) and one to the south-east (St. Naum Spring) (Figure 3.1), showed a standard deviation of ~2 % in temperature (T), ~1 % in pH and ~10 % in salinity (S) (Hydrobiological Institute, unpublished results, 2000 to 2002 for pH and T, 2002 for S). Based on the stable water properties, the intrusion pattern of the spring water is also very constant within a season. In general the salinity of the groundwater is higher than that of the the lake by ~0.06 g kg^{-1}. Large surface springs, entering the lake with higher salinity and lower temperature than ambient lake water will plunge close to the level of neutral buoyancy into the thermocline. For the large spring area of St. Naum with an average flow of 7.5 $m^3 s^{-1}$ (Watzin et al. 2002; Table 3.2), the intrusion depth varies during the summer season from about 15 to 40 m (Figure 3.3b). Despite entrainment of ambient water a pronounced conductivity is seen kilometers from the spring inflow after several months of almost constant stratification in autumn (Figure 3.4a).

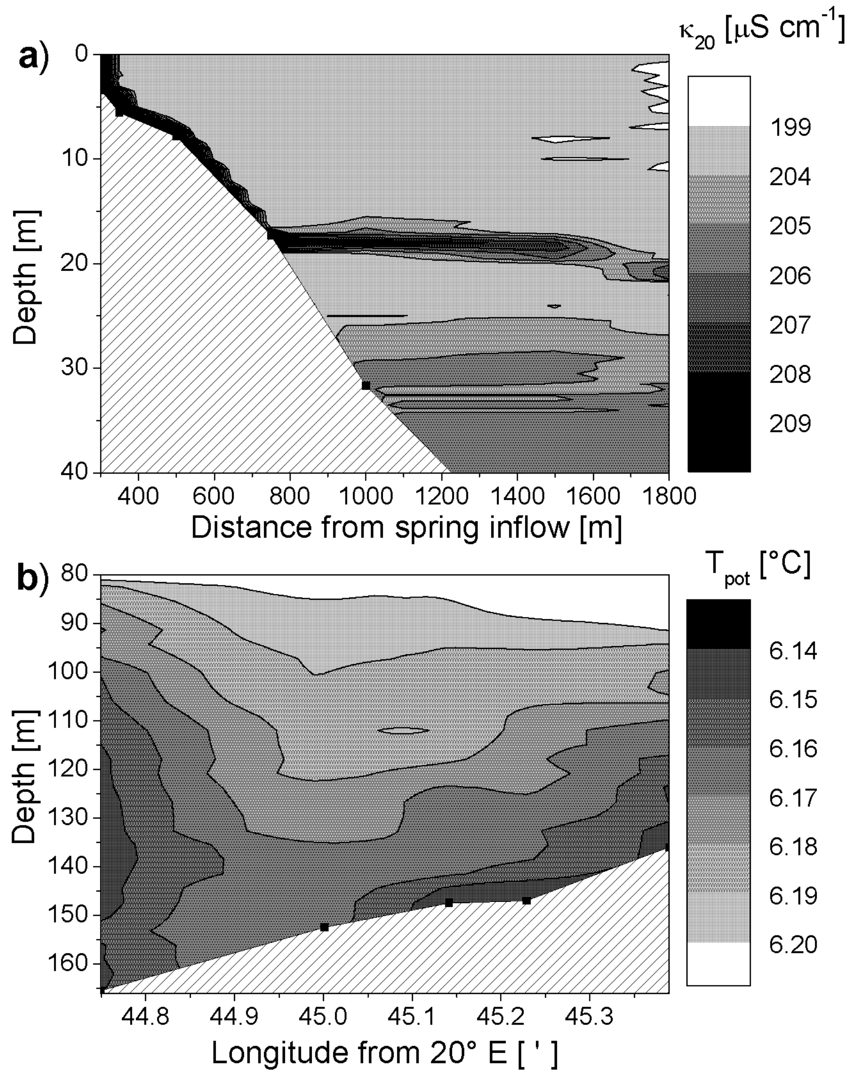

Figure 3.4: Mapping of spring inflow based on CTD transects. (a) Inflow from St.Naum Springs (location in Figure 3.1) plunging from the surface in terms of conductivity κ_{20} on 18-Oct-2002. (b) Temperature transect over a subaquatic spring area in the Northern part of Lake Ohrid on 22-Jun-2004.
Hatched area is lake bottom.

In winter the spring water is warmer and thus lighter than ambient (6 to 8 °C) and merges with the surface layer (Figure 3.3b). The situation is similar for the deep subaquatic springs, which enter the lake in a zone with temperatures between 6 and 7 °C throughout the year. As a consequence the warmer spring waters will rise and form plume-like structures. One such plume with distinct rise of the isotherms by up to 50 m over a short horizontal distance of ~200 m was observed in 150 m depth (Figure 3.4b). This same plume was detected on several instances over one year indicating continuous supply. A model approach by McGinnis et al. (2004) further verifies that subaquatic spring inflows would indeed create a structure similar to the one observed (Figure 3.4b).

3.3.2 Stratification and Mixing

Stratification – Whereas the surface temperature of Lake Ohrid underwent seasonal changes from 6.1 °C to 27 °C in the period 2001 to 2004, temperatures below ~150 m are almost constant around 6 °C (Figure 3.2a). The top 150 to 200 m water column follows the usual stratification seasonality of deep, temperate lakes. Below ~150 m in-situ temperatures show only small depth gradients between -0.02 and 0.15 °C per 100 m. The frequently observed temperature increase towards the bottom can be explained partly by isentropic compression with depth leading to an adiabatic lapse rate $\Gamma \approx$ 0.003 °C per 100 m (Figure 3.5a).

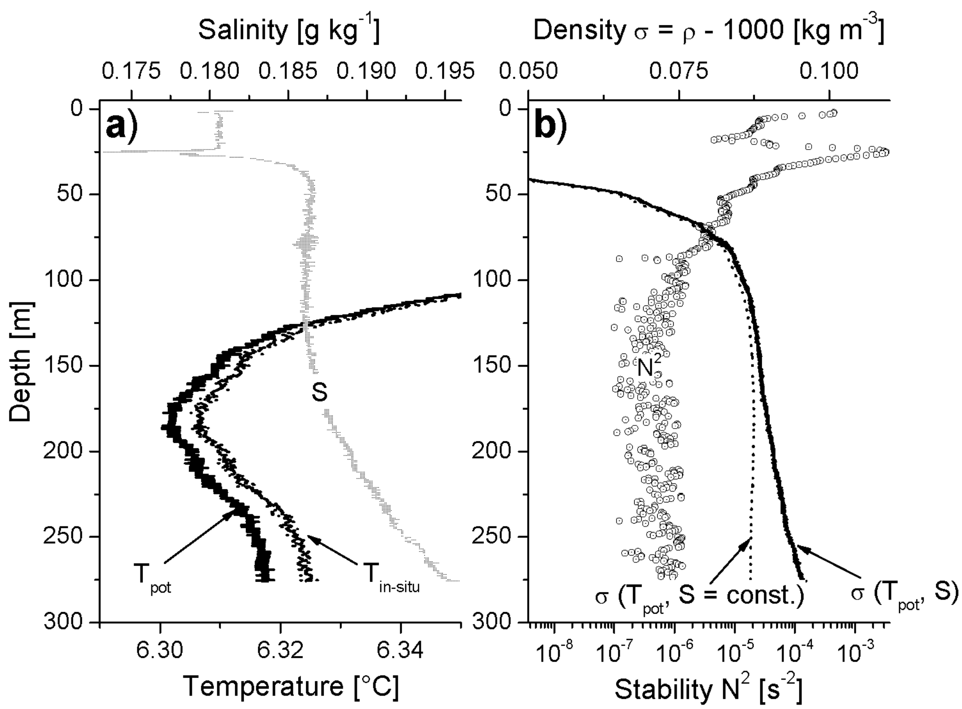

Figure 3.5: Stratification, based on a CTD-profile taken on 17-Sep-2003. (a) Lake salinity, based on measured conductivity, in-situ temperature, as well as potential temperature with reference to lake surface. (b) Calculated profile of potential density, as well as water column stability N^2.

Below ~150 m depth the water column is basically stratified by salinity (S) while temperature has but a small effect (Figure 3.5b). The salinity-induced stability explains the occurrences of inverse temperature stratification in the hypolimnion, as observed in 2001 and 2003 (Figures 3.2a, b and 3.5a). S accumulates in the hypolimnion through mineralization as a result of the productive season (see below). Every winter the salinity

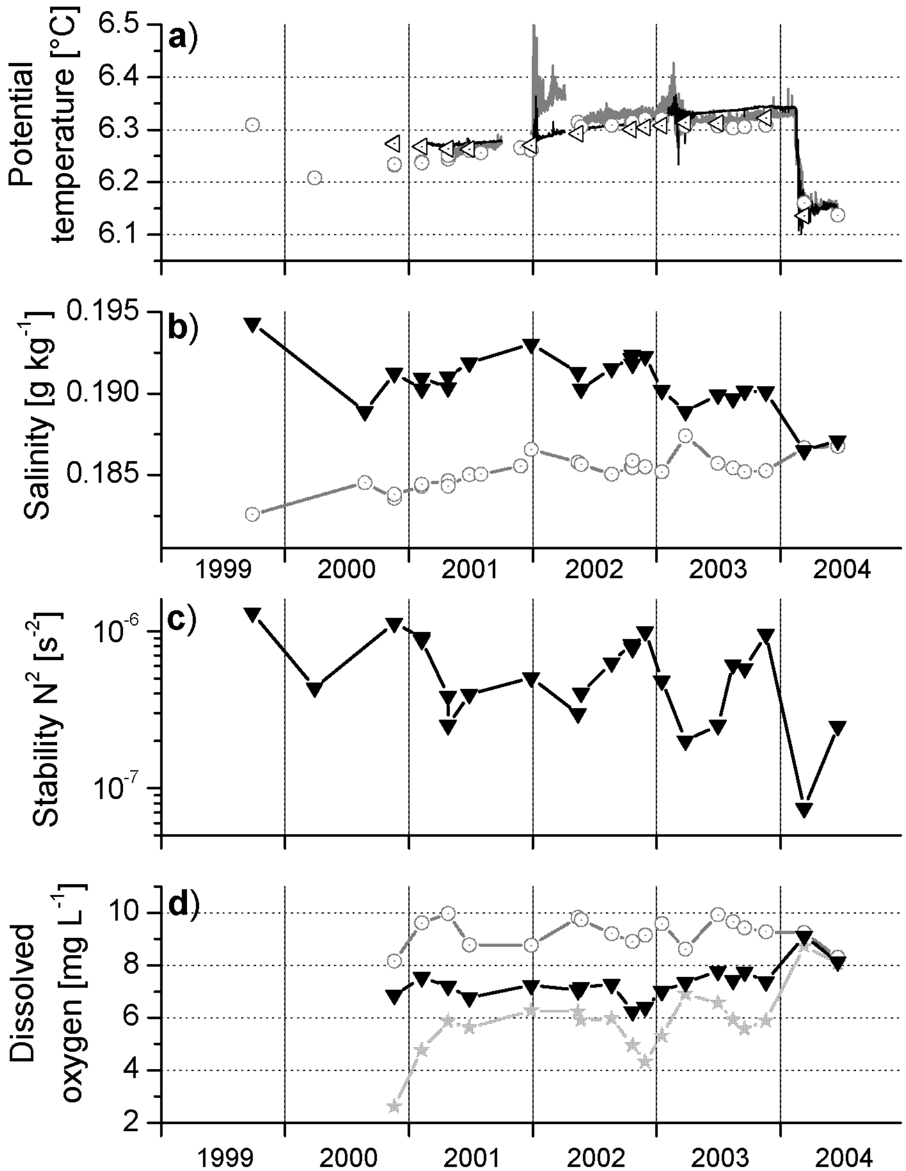

Figure 3.6: Temporal development in deep hypolimnion. (a) Change in temperature, from thermistors (lines) and CTD profiles (symbols). Grey line / circles are at 200, black line / triangles are at 277 m depth. The offset between CTD and thermistor measurements is because of smaller accuracy of the CTD probe. (b) Volume-averaged salinity, 0-200 m (grey circles), 200-250 m (black triangles). (c) Average stability of stratification between 200 and 250 meters depth from CTD measurements. (d) Volume-averaged content in dissolved oxygen from CTD profiles and Winkler measurements, 0-200 m (grey circles), 200-250 m (black triangles); DO directly above bottom (light grey stars).

gradient is partly mixed by increased turbulence. Within the surface layer S is reduced by phytoplankton up-take and calcite formation and increased by evaporation during summer time (Figure 3.2b). Dissolved oxygen (DO) shows a similar seasonal pattern as salinity. During stagnant summer season DO is consumed in the hypolimnion and replenished in winter by turbulent mixing and potentially by plunging rivers (Figure 3.2c).

Occasional Complete Overturn – The term "complete overturn" will be used in the following to describe deep convective winter mixing which consequently leads to a density destratification of the whole water column. Lake Ohrid is an oligomictic lake, with complete overturn occurring roughly every seventh winter (Hadzisce 1966). In the absence of such an event a gradual increase in temperature of ~0.025 °C yr^{-1} was observed in the hypolimnion from 2001 to 2004 (Figure 3.6a), corresponding to an average heat input of ~0.12 W m^{-2}. This warming is the result of turbulent vertical heat flux, geothermal heat flux and subaquatic springs.

From March to November 2003 the temperature gradient at 200 m depth was almost zero and consequently the turbulent vertical heat flux was negligible. Still, temperatures in the deep water during this period continued to increase, as confirmed by thermistors and CTD profiles. Based on the two thermistors in Figure 3.6a the heat input of 0.069 W m^{-2} was found. Geophysical surveys for the area of Ohrid indicate a geothermal heat input of ~0.035 W m^{-2} (Cermak & Hurtig 1979; Hurtig et al. 1992).

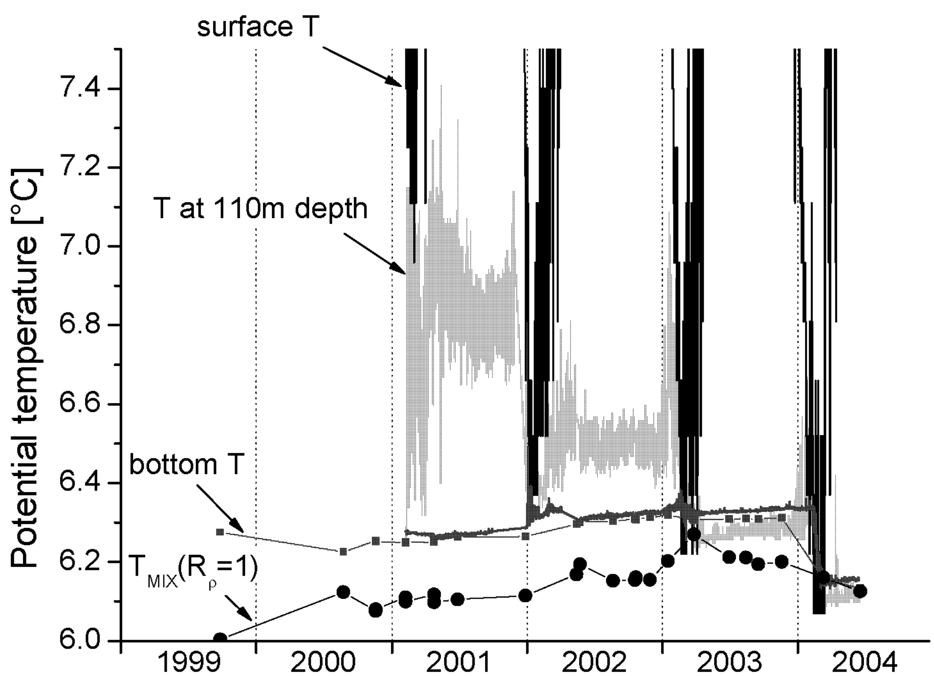

Figure 3.7: Water temperature 1999 - 2004. Temperature at surface and at 110 m depth was measured with Vemco temperature loggers, bottom temperature is based on RBR TR1050 temperature loggers at 200 and 272 m depth (line) and regular Seabird CTD profiles (squares). T_{MIX} (R_ρ = 1) is calculated using equation (2) for one layer above and one below 200 m depth.

Using T ≈ 10 °C in surface springs, the remaining increase of 0.034 W m^{-2} could be explained by subaquatic spring inflows below 200 m depth of 0.3 m^3 s^{-1}.

In March 2004 a complete overturn was observed (Figures 3.2 and 3.6), triggered by a two-week period when lake surface temperatures dropped by ~0.1 °C to 6.1 °C (Figure 3.7). Nevertheless the surface temperature in the two preceding winter seasons was only 0.1 °C higher than in March 2004 (Figure 3.7), illustrating the sensitivity of the lake to surface cooling. To better understand the phenomenon we divided the vertical structure into two layers. A layer MIX above ~200 m, which is mixed with the surface water annually and a bottom layer BOT, which is only mixed during irregular complete overturn. Such a complete overturn can occur if the density in MIX is higher than the density in BOT. In the case of Lake Ohrid the salinity-induced stability must be overcome by negative buoyancy from cooler surface temperatures. This is the case if the so-called mixing ratio

$$0 < R_\rho = \frac{\beta \cdot (S_{BOT} - S_{MIX})}{\alpha \cdot (T_{BOT} - T_{MIX})} < 1 \tag{3.1}$$

where β = 0.805 · 10^{-3} [g kg^{-1}]$^{-1}$ is the coefficient of haline contraction based on Lake Ohrid salt composition, α [°C^{-1}] is the thermal expansivity, S [g kg^{-1}] is salinity, T [°C] is potential temperature. Complete overturn can occur if T_{MIX} becomes less than T_{MIX} at R_ρ = 1. From equation (3.1) we find this theoretical mixing temperature

$$T_{MIX}(R_\rho = 1) = T_{BOT} - \frac{\beta}{\alpha} \cdot (S_{BOT} - S_{MIX}) \tag{3.2}$$

Applying equation (3.2) to data from CTD-profiles and thermistors from 1999 to 2004, yields the function plotted in Figure 3.7. Surface temperatures stayed higher than T_{MIX} (R_ρ = 1) until winter 2003/2004, when a complete overturn took place. T_{BOT} increased at a rate of ~0.025 °C yr^{-1} until complete overturn.

The bottom layer salinity S_{BOT} depends on the intensity of mineralization, as well as the erosion of salinity stratification in winter (Figure 3.6b). The two processes are individually assessed for the period 2000 - 2004 in Figure 3.8, excluding the complete overturn in 2004. Annual net mineralization is very similar over the time of observations (Figure 3.8a). The net erosion of salinity during winter shows more fluctuations, as it depends on turbulent mixing by surface cooling and wind forcing (Figure 3.8b). Linear regressions in Figure 3.8 show that on average the expected annual salt increase from mineralization within BOT is about 75% or 4 metric tons higher than the turbulence-induced decrease in winter. While this relation can be reversed in particular winters (e.g., 2003; Figure 3.6b), on average the salinity-induced stability increases between events of complete overturn. Equation (3.2) shows that the increase in S_{BOT} is equivalent to a decrease in T_{MIX} (R_ρ = 1) by 0.017 °C yr^{-1}. Nevertheless, taking into account both the observed temporal changes in T_{BOT} and S_{BOT} we find a net annual increase in T_{MIX} (R_ρ = 1) of 0.025 – 0.017 = 0.008 °C yr^{-1}. Thus, every year without complete overturn, the probability for a complete overturn will increase. If we assume a similar future temperature and salinity increase as observed from 2000 to 2003 it would take ~8.0 and 6.6 years, respectively for the deep-water to achieve the T and S levels of December 2003.

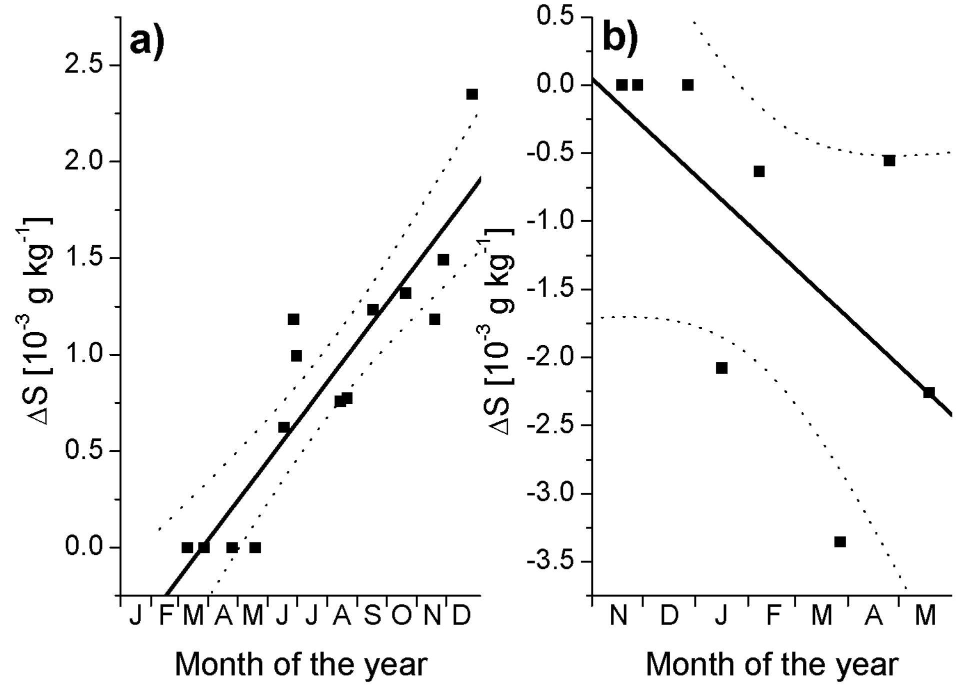

Figure 3.8: Salinity dynamics below 200 m depth from CTD profiles covering the period 2000 - 2004, excluding the deep convection in March 2004. Bold lines are linear regressions; dotted lines are respective 95% confidence limits. (a) Average seasonal salt increase from mineralization during summer of $1.8 \cdot 10^{-3}$ g kg^{-1} or $8.44 \cdot 10^{3}$ metric tons (R = 0.90). (b) Average seasonal salt decrease from enhanced mixing during winter of $-1.05 \cdot 10^{-3}$ g kg^{-1} or $-4.92 \cdot 10^{3}$ metric tons (R = 0.61).

Probability of Complete Overturn – Whether a complete overturn occurs depends on the lowest water temperature in winter $T_{min,MIX}$. Based on temperature measurements in Lake Ohrid from 1979 to 2004 (1979-1999, Hydrobiological Institute, unpublished data), performed at least monthly, an average $T_{min,MIX}$ of 6.56 °C was found. The observed complete overturn in March 2004 (Figure 3.6a) occurred at $T_{min,MIX}$ = 6.10 °C, 0.45°C below average. Based on a histogram of the observed T fluctuations in Figure 3.9 occurrence probability for a specific T deviation can be assessed. As only negative T deviations (cold events) are important for the discussion of complete overturn we used a one-sided histogram for this assessment, assuming normal distribution. A negative deviation of 0.45 °C then leads to an expected occurrence probability of 13 %. This probability corresponds to a reoccurrence periodicity of complete overturn of 7.8 yr, which consistently supports the reported value of ~7 yr by Hadzisce (1966). Figure 3.9 compares the data series for $T_{min,MIX}$ with long-term data from Lake Geneva (V = 89 km^3, max. depth = 310 m) and Lake Constance (V = 48 km^3, max. depth = 254 m), two lakes of oligomictic nature with comparable volumes as Lake Ohrid. Surprisingly deviations from average $T_{min,MIX}$ differ by less than 20% among the three lakes (Figure 3.9).

Turbulent Mixing – During periods without complete overturn, vertical exchange between MIX and BOT is governed by stratified turbulent mixing. The heat budget method (Powell & Jassby 1974) was used to assess turbulent diffusion coefficients K_z (m^2 s^{-1}) for the summer periods 2002 and 2003. For this particular application of the method, we assumed that the observed heat input below a certain depth is the result

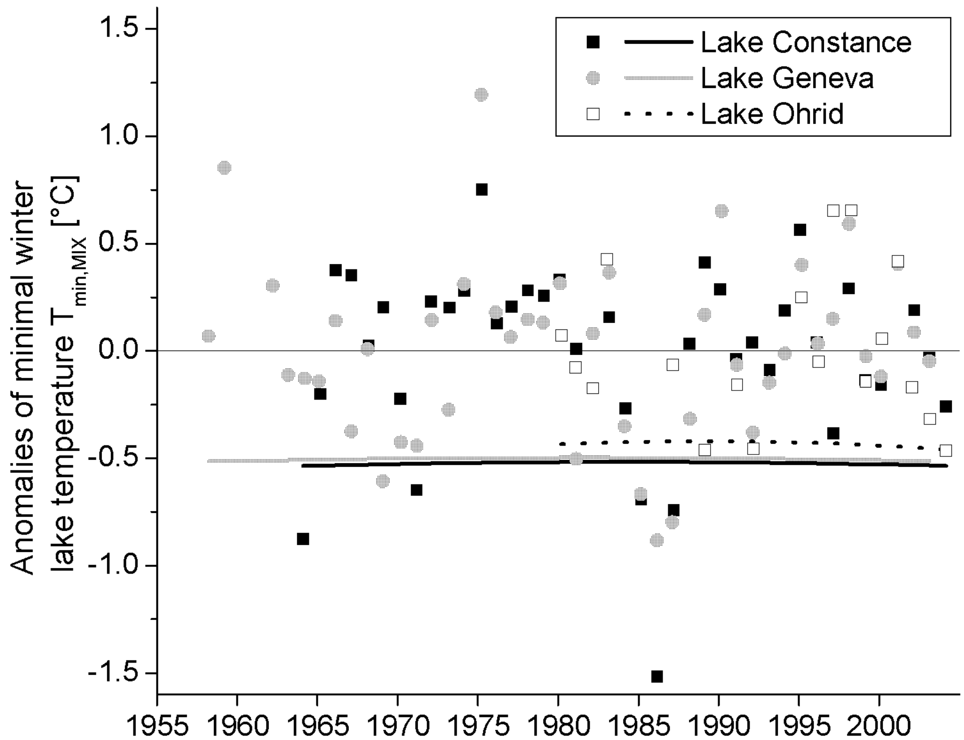

Figure 3.9: Variation in minimal lake surface temperatures $T_{min,MIX}$ in winter for Lake Ohrid, Lake Geneva and Lake Constance. Symbols are based on de-trended measurements from at least monthly temperature profiles and thermistors. Lines are one-sided prediction intervals at 87 % level. Data for Lake Constance are from IGKB (International Commission for the Protection of Lake Constance), Langenargen (GER); data for Lake Geneva are from CIPEL (International Commission for the Protection of Lake Geneva), Lausanne (CH).

of a heat flux from above and the combined geothermal and groundwater heat input of ~0.069 W m^{-2} (see above: occasional complete overturn). In summer an average $K_{z,sum}$ = $3.5 \cdot 10^{-5}$ m^2 s^{-1} was found at the interface between MIX and BOT in 200 m depth. Based on the net observed reduction in salinity in BOT (Figure 3.8b) and the input from River Sateska, the winter diffusivity $K_{z,win}$ was found to be $2.8 \cdot 10^{-4}$ m^2 s^{-1}, almost one order of magnitude greater than $K_{z,sum}$. The result of this increased turbulence can also be seen in temperature fluctuations at depths of 200 m and 277 m every winter (Figure 3.6a).

The observed mixing is also partly the result of buoyancy flux J_b [W kg^{-1}] caused by the subaquatic springs

$$J_b = \frac{g}{\rho} \cdot \left(\frac{Q \cdot \Delta\rho}{A} \right) \tag{3.3}$$

where g [m s^{-2}] is acceleration due to gravity, Q [m^3 s^{-1}] is the spring inflow, $\Delta\rho$ [kg m^{-3}] is the density difference from ambient lake water and A [m^2] is the cross sectional area at the considered depth. J_b can be transformed into K_z using the Osborn (1980) relation

$$K_z = \gamma_{mix} \cdot \frac{\varepsilon}{N^2} \sim \frac{\gamma_{\text{mix}}}{2} \cdot \frac{J_b}{N^2} \tag{3.4}$$

where ε [W kg^{-1}] is the dissipation of turbulent kinetic energy equal to ½ J_b for a buoyant plume (Wüest & Lorke 2003), N^2 [s^{-2}] is the stability frequency and γ_{mix} [-] is the mixing efficiency, set to 0.2, as typical for convective turbulence (Ivey & Imberger 1991). We applied equations (3.3) and (3.4) to the plume model of McGinnis et al. (2004), assuming 20 springs with inflows of 0.5 m^3 s^{-1} distributed between 200 and 50 m depth. The result shows that particularly below 100 m depth the contribution to basin-scale vertical diffusivity by subaquatic springs is of the same magnitude as observed $K_{z,sum}$.

Large-scale Wind-induced Mixing – An interesting phenomenon has been first observed by Stankovic & Hadzisce (1953), while profiling temperature on east-west transects in the northern part of the lake. On many instances they found that the isotherms are bent downwards approaching the shore forming a cone-shaped structure. These findings were verified with CTD transects both in N-S and E-W direction in June 2004 (Figure 3.10a). Remote sensing images from 2001 to 2004 indicate cooler surface water in the northern lake center on many instances, often extended over time periods of several days. Before and during the measurements presented in Figure 3.10a remote sensing images also confirm the persistency of the phenomenon (Figure 3.10b). Thus the cone-shaped structure cannot be the result of oscillating processes, such as internal seiching. The plausible explanation of such a persistent feature is provided by the so-called "Ekman pumping". In the bounded water body of a stratified lake, wind parallel to the shore causes downwelling/upwelling concentrated along the shore to within a distance, given by the baroclinic Rossby radius $a = c f^{-1}$, where c is the speed of long internal waves [m s^{-1}] and f [s^{-1}] is the Coriolis parameter (Gill 1982). For Lake Ohrid ~6.5 km is found in June 2004, well within the lake shoreline. The wind predominantly blows along the north-south axis and most often towards the south. Given the protection of the north-eastern lake side by hills, wind towards the south would be expected mainly along the western shore, whereas wind towards the north is probably equally distributed. In combination an anti-clockwise current seems to evolve along the northern lake shore with downwelling at the shore from Coriolis force and finally an up-welling in the centre. The downwelling velocity of Ekman pumping can be described in a two-layer approach by $c \cdot \tau \cdot (\rho \cdot g' \cdot H_1)^{-1}$ (Gill 1982), where τ [N m^{-2}] is the wind stress, g'[m s^{-2}] is the reduced gravity (= $g \cdot \Delta\rho/\rho$) and H_1 [m] is the depth of the surface layer. As wind speed exceeds 6 m s^{-1} only during ~3 % of the time in summer, downwelling velocities are small (wind data for station in Ohrid, US National Climatic Data Center, Station Nr. 135780). Using the above term it would take about 20 hours of a wind speed of 5 m s^{-1} to create a downwelling of surface water to 15 m depth as seen in Figure 3.10a. If indeed Ekman pumping is the reason for the observed temperature gradients both south and north winds have to contribute to an anti-clockwise current.

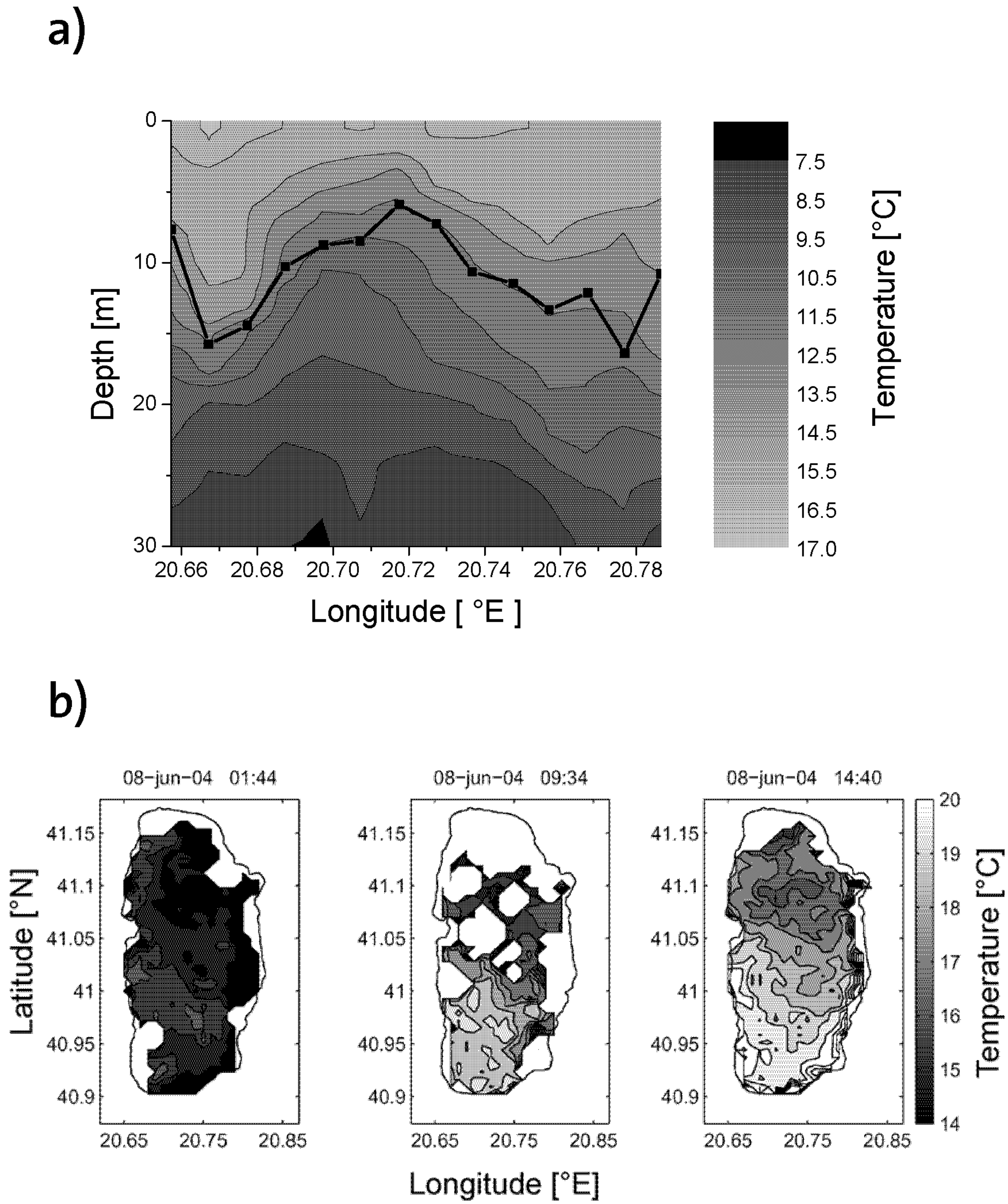

Figure 3.10: Upwelling in the northern part of Lake Ohrid on 8-June-2004. (a) Temperature contour from CTD transect at 41.105° N latitude from 10:30 to 14:30. Maximum pH indicates depth of major photosynthetic production (connected symbols). (b) Surface temperatures from remote sensing based on AVHRR measurements from NOAA satellites (detailed method in Oesch et al. 2005). White areas indicate missing values (e.g., because of clouds).

3.3.3 Oxygen and Salinity Balances

The processes described above influence the mass balances of DO and S (Table 3.3). As defined above, the water column is divided into an annually mixed section MIX above 200 m depth and the deep hypolimnion BOT. To close the balance for S_{MIX} we set $\Delta S_{MIX}/\Delta t = 0$, as the major fluxes are external and can be expected to be in equilibrium. The balance for S_{BOT} is based on observed net mineralization and net winter loss (Figures 3.8a and b). For DO_{BOT} it is assumed that the observed difference between MIX and BOT (Figure 3.6d) is built up during ~8 years of permanent stratification, resulting in an average annual net decrease $\Delta DO_{BOT}/\Delta t$ = -0.3 mg L^{-1} yr^{-1}. DO_{MIX} is returned to 89% saturation annually during winter convection.

Table 3.3: Oxygen and salt balance of Lake Ohrid in absence of complete deep convection

Process	Governing Equation [1]	Average annual rates of change		
		DO [mg L^{-1} yr^{-1}]	Salinity [g kg^{-1} yr^{-1}]	
		200 – 250 m (BOT)	0 – 200 m (MIX)	200 – 250 m (BOT)
River input (to BOT when plunging)	$\frac{Q_{riv,i}}{V_i} \cdot \left(X_{riv} - X_{LO,i}\right)$	0.05	$0.56 \cdot 10^{-3}$	$1.21 \cdot 10^{-3}$
Spring input (surface and subaquatic)	$\frac{Q_{spring,i}}{V_i} \cdot \left(X_{spring} - X_{LO,i}\right)$	0.006	$0.80 \cdot 10^{-3}$	$0.12 \cdot 10^{-3}$
Turbulent diffusion in winter (during 2 months)	$K_{z,win} \cdot \frac{\Delta X}{\Delta z} \cdot \frac{A_{200m}}{V_i}$	0.81	$0.21 \cdot 10^{-3}$	$-2.86 \cdot 10^{-3}$
Turbulent diffusion in summer (during 10 months)	$K_{z,sum} \cdot \frac{\Delta X}{\Delta z} \cdot \frac{A_{200m}}{V_i}$	0.50	$0.13 \cdot 10^{-3}$	$-1.40 \cdot 10^{-3}$
Evaporation (salinity increase at lake surface)	$\frac{Q_{evap}}{V_{MIX}} \cdot S_{MIX}$	-	$1.49 \cdot 10^{-3}$	-
Salt loss to sedimentation	constant	-	$-0.34 \cdot 10^{-3}$	-
Outflow	$-\frac{Q_{out}}{V_{MIX}} \cdot S_{MIX}$	-	$-2.84 \cdot 10^{-3}$	-
Mineralization	constant	-1.67	-	$3.09 \cdot 10^{-3}$
Total inputs		1.37	$3.21 \cdot 10^{-3}$	$4.41 \cdot 10^{-3}$
Total outputs		-1.67	$-3.21 \cdot 10^{-3}$	$-3.66 \cdot 10^{-3}$
Annual change		-0.30	0	$0.75 \cdot 10^{-3}$

[1] Rates of change are in [g kg^{-1} s^{-1}] for S and [mg L^{-1} s^{-1}] for DO, X is either DO [mg L^{-1}] or S [g kg^{-1}], subscript i refers either to compartment MIX or BOT, subscript LO stands for Lake Ohrid, Q [m^3 s^{-1}] are average annual flow rates, V [m^3] is compartment volume ($V_{MIX} = 50.3 \cdot 10^9$ m^3 and $V_{BOT} = 4.7 \cdot 10^9$ m^3), $A_{200m} = 149.8 \cdot 10^6$ m^2 is cross-sectional lake area at 200 m depth, $K_{z,win} = 2.8 \cdot 10^{-4}$ m^2 s^{-1} and $K_{z,sum} = 3.5 \cdot 10^{-5}$ m^2 s^{-1} are turbulent diffusion coefficients, Δz = 125 m is depth difference between MIX and BOT.

S_{BOT} and DO_{BOT} are governed by diffusive exchange with MIX and mineralization of settled material. Plunging of River Sateska contributes about one fourth of the salinity input to BOT, whereas it is almost negligible for DO input for the current situation. However its effect will increase with growing difference to the ambient lake concentration (equation in Table 3.3). For S_{BOT} the calculations are based on observations in Figure 3.8a and b. For DO_{BOT} the calculated summer net decrease of 1.16 mg L^{-1} yr^{-1} (Table 3.3) is in agreement with observed 0.79 ± 0.28 mg L^{-1} yr^{-1} from 2000 - 2004. If the annual fluxes (except diffusion) are constant DO_{BOT} will approach a level, where diffusion equals consumption. Under the above assumptions this "equilibrium" DO level is found at 6.7 mg L^{-1}. Indeed the observed volume averaged DO between 200 and 250 m is above this value (Figure 3.6d). Nevertheless the DO level directly above the lake bottom can be up to 4 mg L^{-1} lower than the average DO_{BOT} (Figure 3.6d). Evaporation and the main outflow are dominating the salt balance for S_{MIX}, followed by spring input. Net salt loss from MIX through adsorption to settling particles is comparably small. Given the numbers in Table 3.3, 80% of the salt leaving MIX with settling particles is expected to be mineralized in BOT, significantly more than the 13 and 60 %, found by Ramisch et al. (1999) for two eutrophic lakes. This difference can be explained by the relatively high contents in dissolved $CaCO_3$ in the lakes observed by Ramisch et al. (1999). Still it seems likely that we underestimate the salt loss to sedimentation. Such an underestimation could be explained by the high sensitivity of the salt loss term to errors in the balance: e.g., even an increase of 10% in evaporation would increase the sedimentation term to the expected range (~60% re-dissolution). However, given the small effect of the unknown error in the salt balance in MIX, no adaptation has been made.

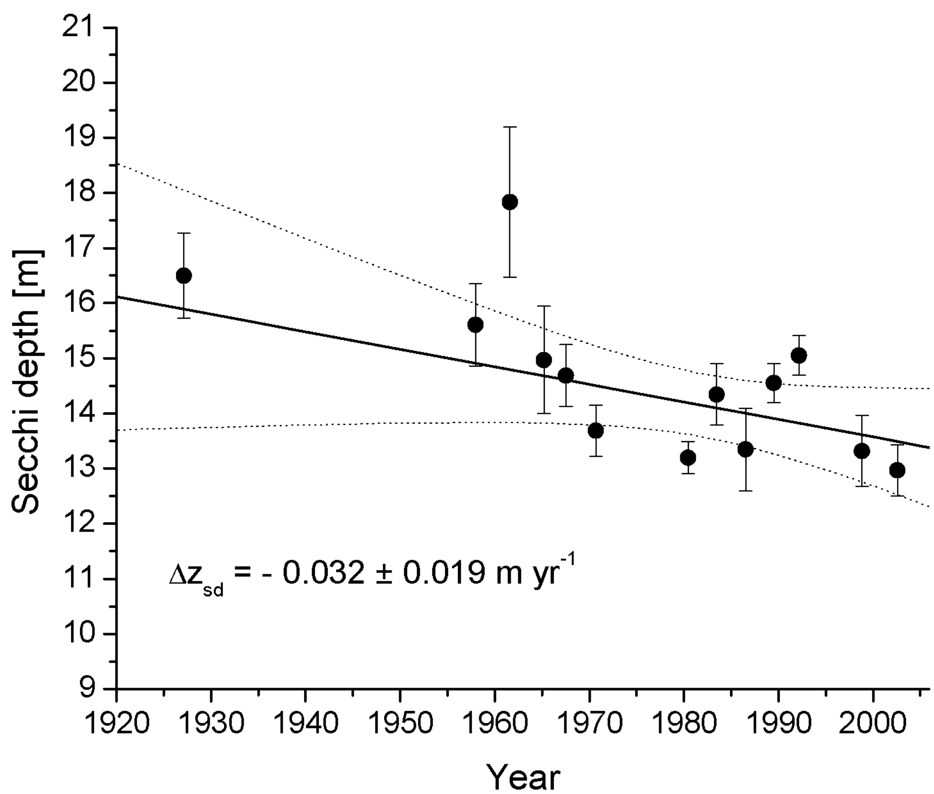

Figure 3.11: Pelagial Secchi depth measurements. Circles indicate 3-year averages from productive season (Mar - Oct), error bars represent error of mean. The bold line shows linear regression using $(errors)^{-2}$ as weight, dotted lines are 95% confidence limits. Data are taken from Stankovic (1960), Ocevski (1974b), unpublished data from Hydrobiological Institute (1979 - 1995), Patceva (2001) and Patceva (2005).

3.3.4 Water Clarity

Lake Ohrid is an exceptionally clear lake with an average Secchi depth z_{sd} of ~14 m (Figure 3.11). The clarity was also confirmed in a PAR-profile (400 to 700 nm) taken in June 2004 using a pair of spherical LI-COR sensors. In the profile the visible light penetrated below 90 m with a 1‰-compensation depth (Kalff 2002) between 50 and 60 m and an average extinction coefficient of $\eta = 0.13 \pm 0.004\ m^{-1}$. This value is comparable to values by Kalff (2002) from clear and oligotrophic lakes, such as Lake Perry (0.2 m^{-1}) and Crater Lake (0.05 m^{-1}). The empirical relation from Wetzel (2002)

$$\eta = 1.58 \cdot z_{sd}^{-1} \tag{3.5}$$

yields $\eta = 0.11\ m^{-1}$, so that the Secchi and PAR estimates of η are in agreement.

The clear water reflects oligotrophic conditions, with a production maximum often below z_{sd} and with generally low phytoplankton concentrations (average chlorophyll A (ChlA) during the productive periods 2001 - 2003 of ~1.3 ± 0.1 mg m^{-3} (Patceva 2005). Moreover the "filtered" spring water and the relatively small surface runoff lead to exceptionally small loads of suspended particles.

3.4 Definition of Human Impacts

In the following observations of anthropogenic changes in the physical boundary conditions are presented and their future potential discussed. For the impact assessment the changes are grouped into four scenarios:

(A) the historic situation, about one century ago, before major human impacts occurred,
(B) the observed status quo,
(C 1) expected future development during global warming process and
(C 2) at a new, warmer temperature equilibrium.

The main assumptions for each scenario are summarized in Table 3.4.

Table 3.4: Human impacts under historic, current and future scenario

Human impact	Unit	Historic situation[1]	Status quo[1]	Future development[1]	
		A	B	C1	C2
Eutrophication:					
TP concentration in Lake Ohrid	mg P m^{-3}	2.25	4.5	9	
Changes in water balance:[1]					
Annual inflow					
- surface springs	$m^3\ s^{-1}$	13.6	10.3	2.9	
- subaquatic springs	$m^3\ s^{-1}$	10.8	9.9	6.6	
- River Sateska	$m^3\ s^{-1}$	-	5.5	3.6	
- other tributaries	$m^3\ s^{-1}$	3.9	3.4	3.2	
Total change from historic (A)	$m^3\ s^{-1}$	-	+0.8	-12	
Global warming:					
Increase of lake surface T	°C yr^{-1}	0	0.006	0.04	0
Increase in total lake temperature (new equilibrium)	°C	-	-	-	4

[1] detailed account of the scenarios is given in the text

[2] status quo assumes 2 $m^3\ s^{-1}$ use of spring water and 1 $m^3\ s^{-1}$ from other tributaries during summer season. Runoff measurements are based on own measurements (2002-2003) and data from Ivanova (1974), the Macedonian Hydrometeorological Institute (unpublished data) and the Hydrobiological Institute Ohrid (unpublished data).

3.4.1 Eutrophication

As mentioned above, phosphorus (P) is the key eutrophication factor in Lake Ohrid. This eutrophication is mainly the result of increased human population in the catchment area and probably progressive use of P-containing washing agents and of fertilizer in agriculture. However, eutrophication is hard to track, because of the low P-concentrations (Total phosphorus (TP) ~4.5 mg m^{-3}) and the short history of precise TP-measurements. Still, clear changes are visible in the close vicinity of polluted inflows, with increased nutrient concentration, shifts in species composition and appearance of coliforms (Watzin et al. 2002; Patceva et al. 2004; Lokoska and Jordanoski 2004). Combining the observed increase in sedimentary P (Figure 3.12) with inflow measurements in a simple linear phosphorus model (Vollenweider 1969), revealed a ~3 fold increase in TP-concentration since historic times and a doubling over the past ~100 yr (Matzinger et al. 2004). In the following assessment we assume 50 % reduced TP concentration in the lake for scenario (A) and a further doubling in the future for (C) (Table 3.4). The assumption of slow increase in scenario (C) seems plausible, given the potential ongoing population increase, intensified agriculture but concurrent improvement of sewage treatment (Ernst Basler and Partners 1995).

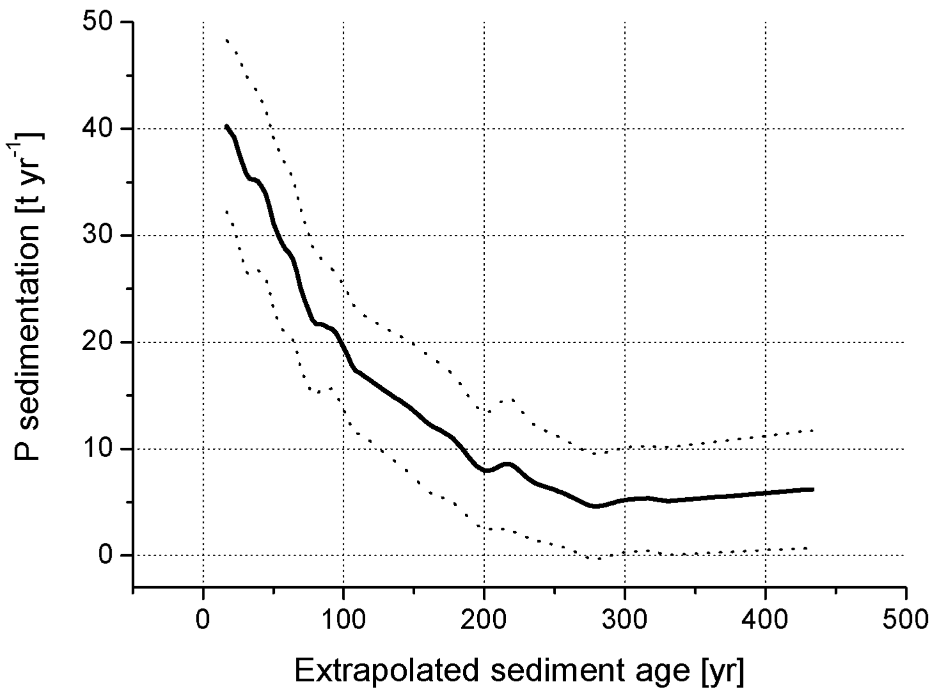

Figure 3.12: Bold line is average sedimentation of TP, dotted lines indicate error of mean. Results are based on three sediment cores taken along the north-south axis at 122, 202 and 282 m depth in 2002 and 2003. Sediment age is based on ^{210}Pb measurements and sediment trap results from 2001 - 2004.

3.4.2 Changes in Water Balance

As the area is rather dry with hot summers, water is generally scarce, particularly for use in agriculture. During the summer season, water is diverted from all tributaries of both lakes for irrigation, reducing their flow rate (e.g., ΔQ ~2.5 $m^3\ s^{-1}$ for River Sateska) or setting them completely dry. Moreover there are groundwater pumps and systems to take water directly from upstream Lake Prespa. The growing need for irrigation water has its most obvious effect on Lake Prespa, where the water level has dropped by ~6 m over the past decade (Matzinger et al. 2006a). The water use today, reduces the annual natural water inflows by 4-5 $m^3\ s^{-1}$, excluding River Sateska. This estimated reduction is composed of 40 % due to irrigation in the Lake Prespa catchment (Matzinger et al. 2006a), 40% due to use of surface spring water (mainly for water supply but also for irrigation in Albania) and 20% by irrigation in the north of Lake Ohrid. The historic scenario (A) is simplified by assuming that then no significant water use was present.

Apart from the reduction in water inflow, another major change in status quo (B) compared to (A) is the artificial diversion of River Sateska in 1962, which contributes about 5.5 $m^3 s^{-1}$ to the total river inflow today and thus compensates for the recent water losses described above (Table 3.2, Figure 3.1).

In a worst case future scenario (C) the tributaries and the surface spring inflows could be used completely during the summer season from May to October. Moreover Lake Prespa could basically be set dry. Taking into account the summer runoff of the rivers and the surface springs from Lake Prespa (Matzinger et al. 2006a), the annual water balance would be reduced by 13 $m^3 s^{-1}$ compared to status quo. During the summer season the surface springs and the tributaries would be used to their full capacity of 10.3 and 4 $m^3\ s^{-1}$, respectively and the subaquatic springs would be reduced by their current Lake Prespa share of 3.3 $m^3 s^{-1}$ (Table 3.4).

3.4.3 Global Warming

CO_2-levels in the atmosphere are very likely to increase in the next century (IPCC 2001). The potential effects of two IPCC scenarios A2 and B2 (CO_2 level increase from the base level of 353 ppm to 822 and 1143 ppm, respectively, in the year 2100) on European climate have been simulated with coupled global and regional circulation models by Räisänen et al. (2004) and Giorgi et al. (2004) who predict a significant increase in air temperature for the southern Balkan Peninsula from 1961 - 1990 to 2071 - 2100 of 4 ± 2 °C and 3.6 ± 0.9 °C, respectively, substantially larger than the 0.6 °C increase observed over the 20th century (IPCC 2001). It was shown that recent changes in air temperatures are also well reflected in average lake temperatures, whereas a smaller temperature increase is expected in the lakes' deep water, due to limited exchange with the main water body (cf., Livingstone 2003; O'Reilly et al. 2003). Observations for the deep water of Lake Ohrid since the 1920s show an increase of 0.005 ± 0.001 °C yr^{-1} ($R = 0.37$, $p<0.0001$) (Figure 3.13).

In the following discussion we will use the expected increase in air temperature of 0.04°C yr^{-1} for scenario (C), 0.006 °C yr^{-1} for status quo (B) and no increase for (A). We further assume that the potential increase in air temperature is directly transferable to surface water. This assumption is confirmed by observations (Livingstone & Lotter 1998) and the coupling of global circulation models with hydrological models (Mortsch & Quinn 1996; Blenckner et al. 2002). Based on the model results of Giorgi et al. (2004), the magnitude of interannual variation in winter air temperatures is assumed to be unaffected by climate change. Changes in other climate parameters, such as wind speed or precipitation, are not considered in our analysis, because their simulated changes were often in the range of expected variability (Räisänen et al. 2004; Giorgi et al. 2004).

In assessing the impact of an increase in lake surface temperatures we differentiate between (i) the ramp-up period of ongoing warming as expected over the next century and (ii) a new equilibrium with sub-scenarios (C1) and (C2) (Table 3.4).

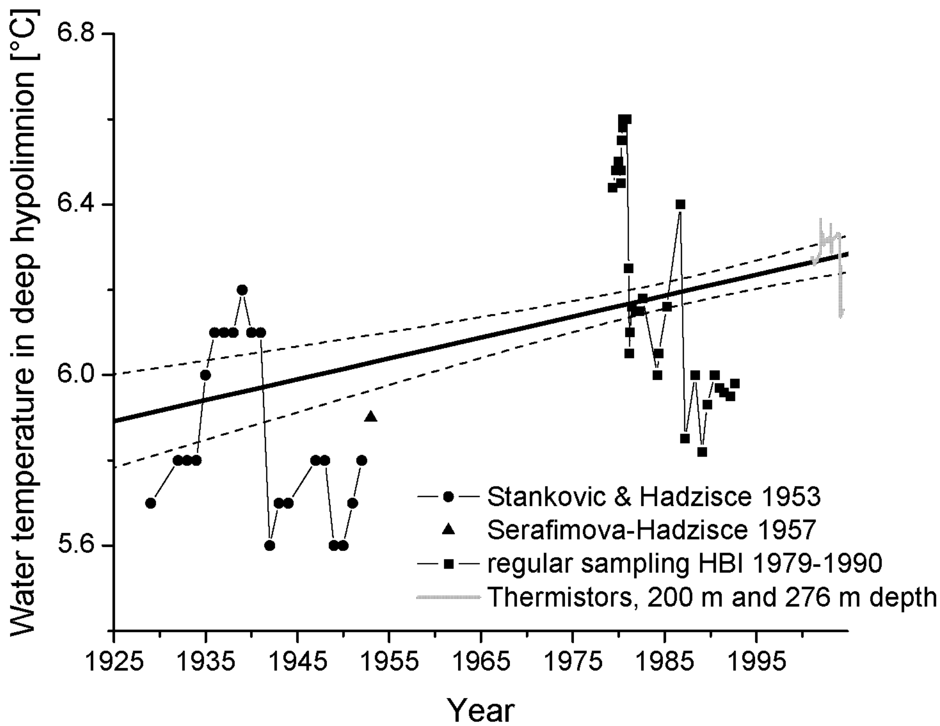

Figure 3.13: $T_{in\text{-}situ}$ measurements from different sources in the deep water between 200 and 285 m depth. The bold line is linear regression; dotted lines are 95% confidence limits.

3.5 Discussion - Analysis of Expected Changes

3.5.1 SRP Supply between 20 and 150 m Depth

As has been shown in the approach endemic phytoplankton species are dependent on nutrient supply, particularly bio-available SRP, to depths between 20 and 150 m. This nutrient supply is based on the input of springs, rivers and mineralization of organic matter at the lake bottom, as well as Ekman pumping in the upper water column. It can be expected that the water circulation of the top 20 m by Ekman pumping would remain unchanged under our assumption of constant wind speeds in all scenarios.

However inflow and mineralization are affected by changes in the water balance and eutrophication.

The main source of nutrients comes from surface and subaquatic springs, which continuously supply SRP to the 20 – 150 m depth interval throughout the year (Figure 3.4). Based on the assessment of actual and historic P concentrations in Lake Prespa and precipitation-fed springs by Matzinger et al. (2006a) the spring P-loads for the three scenarios can be calculated. It was found that the input during six summer months increased from scenario (A) to scenario (B) by merely 2 % to the current load of ~$1.8 \cdot 10^3$ kg P. This increase, despite a reduction in spring inflow (Table 3.4), is the result of observed ongoing eutrophication of Lake Prespa (Matzinger et al. 2006a). However, under scenario (C), without inflow from Lake Prespa and no surface spring inflow in summer, the spring P-input would be reduced to ~$0.3 \cdot 10^3$ kg P, ~16 % of the current value. This dramatic reduction contrasts the overall eutrophication from (A) to (C) (Table 3.4).

TP-sedimentation in the past has developed in parallel to lake P-concentration (Figure 3.12). This proportionality is likely to be true into the future, both for TP-sedimentation and P-concentration of River Sateska (P-concentrations taken from Jordanoski 1999). Based on SRP measurements in the water column it is assumed that ~50% of the settled TP is released as SRP and makes its way upwards through the water column (Matzinger et al. 2004). Based on these assumptions a maximal annual SRP-flux from the lake bottom of 10, 20 and $40 \cdot 10^3$ kg P yr^{-1} is expected for scenarios (A), (B) and (C), respectively, much higher than contributions from springs. Summarizing the above we found that

(i) there is a significant increase in phosphorus availability from scenario (A) through (C) close to the lake surface and towards the bottom from anthropogenic eutrophication, but
(ii) up to an 80% decrease in continuous SRP supply between 20 and 150 m depth during the summer season must be expected from reduced spring inflow.

3.5.2 Water Clarity in the Trophogenic Zone

Water clarity is certainly a crucial factor for the endemic phytoplankton species, given their deep habitat with light availabilities around 1 ‰ of surface radiation. Given the very low inorganic particle concentrations in Lake Ohrid, water transparency depends mainly on phytoplankton itself. Contrary to the natural nutrient input, which stems predominantly from the karst aquifers and plunges into the thermocline or below during the productive season, the bulk of the anthropogenic nutrient sources enter the surface layer of the lake through rivers, sewage channels and diffuse sources. In summer, when the rivers are almost dry from upstream irrigation, they still serve as sewage channels from the major towns and villages close to the lake shore (Veljanoska-Sarafiloska 2002). Thus eutrophication would mainly trigger phytoplankton production in the surface layer and reduce light availability at greater depths. For the change from historic (A) to status quo (B), Secchi depth observations indeed show a significant reduction by ~0.05 m yr^{-1} ($p = 1.04 \cdot 10^{-4}$) since 1926 for the summer periods (Figure 3.11). For an estimate of the effect of increased productivity we used the empirical relationship from Kalff (2002)

$$\mathrm{ChlA} = 0.407 \cdot \mathrm{TP}^{0.874} \tag{3.6}$$

where ChlA and TP are to be inserted in [mg m^{-3}]. Equation (3.6) resulted in a historic ChlA concentration of 0.8 mg m^{-3}, compared to the present of 1.5 mg m^{-3}, which is very close to the measured value of 1.3 mg m^{-3}. The effect of higher ChlA on light extinction coefficient η is $\Delta\eta \approx 0.019$ m^{-1} (mg-ChlA $m^{-3})^{-1}$ (average of values from Smith & Baker 1978 and Megard et al. 1980). In our case, we then find an increase in η from 0.117 to 0.13 m^{-1} since 1926, signifying a decrease of the theoretical 1 ‰-compensation depth by ~6 m. In both cases the major share of light extinction is from pure water. Using equation (3.5) we expect a concurrent reduction in Secchi depth of 1.4 m, which is about 60% of what is observed (Figure 3.11). For scenario (C) with doubled TP-level (Table 3.4) we expect a ChlA level of 2.8 mg m^{-3}, a coefficient η of 0.154 m^{-1} and finally a further decrease in the 1‰-compensation depth from today by ~8 m. In total a decrease in theoretical 1 ‰-compensation depth of ~14 m is expected from scenario (A) to (C).

Given the predominance of green algae and cyanobacteria in the top 20 m of the lake (Patceva 2005), these species would also be the main beneficiaries of additional nutrient inputs at the surface. Thus while the overall lake productivity increases, the habitat of deep living, endemic phytoplankton species would be reduced. The negative impact on these communities might even be amplified by their reduced competitiveness under nutrient-rich conditions, as observed in the vicinity of polluted inflows (Watzin et al. 2002).

3.5.3 Oxygen Supply to the Hypolimnion

Availability of DO was identified as one main factor for preserving the endemic bottom fauna. The DO content in the irregularly mixed bottom layer BOT is replenished by (i) the plunging River Sateska, (ii) input from subaquatic springs, (iii) diffusive input from MIX and (iv) complete overturn. On the other hand it is consumed by mineralization of settled organic material (Table 3.3). Given this multitude of influences each scenario (past, present, future) is assessed separately. Our general approach starts at the observed situation after the complete overturn in June 2004 (Figure 3.6), again dividing the lake into two layers MIX and BOT. For each year without complete overturn changes are evaluated iteratively, based on equations in Table 3.3. Turbulent diffusion coefficients $K_{z,win}$ and $K_{z,sum}$ are re-calculated after each time step based on changes in stability N^2 using equation (3.4). Finally, changes in initial values allow re-calculation of the theoretical mixing temperature T_{MIX} ($R\rho = 1$) and its deviation from T_{MIX}. Occurrence probability p_{co} of this T deviation can be calculated as discussed above, based here on the combined histogram of observed fluctuations in the minimal winter temperatures $T_{min,MIX}$ from Lake Ohrid, Lake Constance and Lake Geneva (Figure 3.9). In other words, p_{co} is the probability that T_{MIX} ($R\rho = 1$) is reached and thus complete overturn takes place. While the evaluation of S_{MIX}, S_{BOT}, and DO_{BOT} for successive time steps is straight forward, DO_{MIX}, T_{MIX} and T_{BOT} require a different approach. During winter mixing DO_{MIX} and T_{MIX} adapt to conditions at the surface. In the iterative evaluation T_{MIX} is assumed to reach an average minimal water temperature $T_{min,MIX}$ every winter. This average is set constant at $T_{min,MIX}$ = 6.56 °C in the absence of global warming. Based on observations DO_{MIX} is set to 89 % of saturation at T_{MIX} every year. Change in T_{BOT} can then be estimated based on observed increase from geothermal and subaquatic springs (~0.014 °C yr^{-1}) and turbulent diffusive heat flux from MIX. For the years 2001-2004 this heat flux from MIX to BOT was governed by local positive and negative vertical temperature gradients, which are often observed to be alternating below 150 m depth (Figures 3.2a and 3.6a). As a result our approach using global vertical temperature gradients between T_{MIX} and T_{BOT} overestimates the heat flux by a factor of ~4 during the years of observation. For the model it was assumed that this factor remains the same for varying temperature differences between T_{MIX} and T_{BOT}. Figure 3.14 shows the results of this iterative approach.

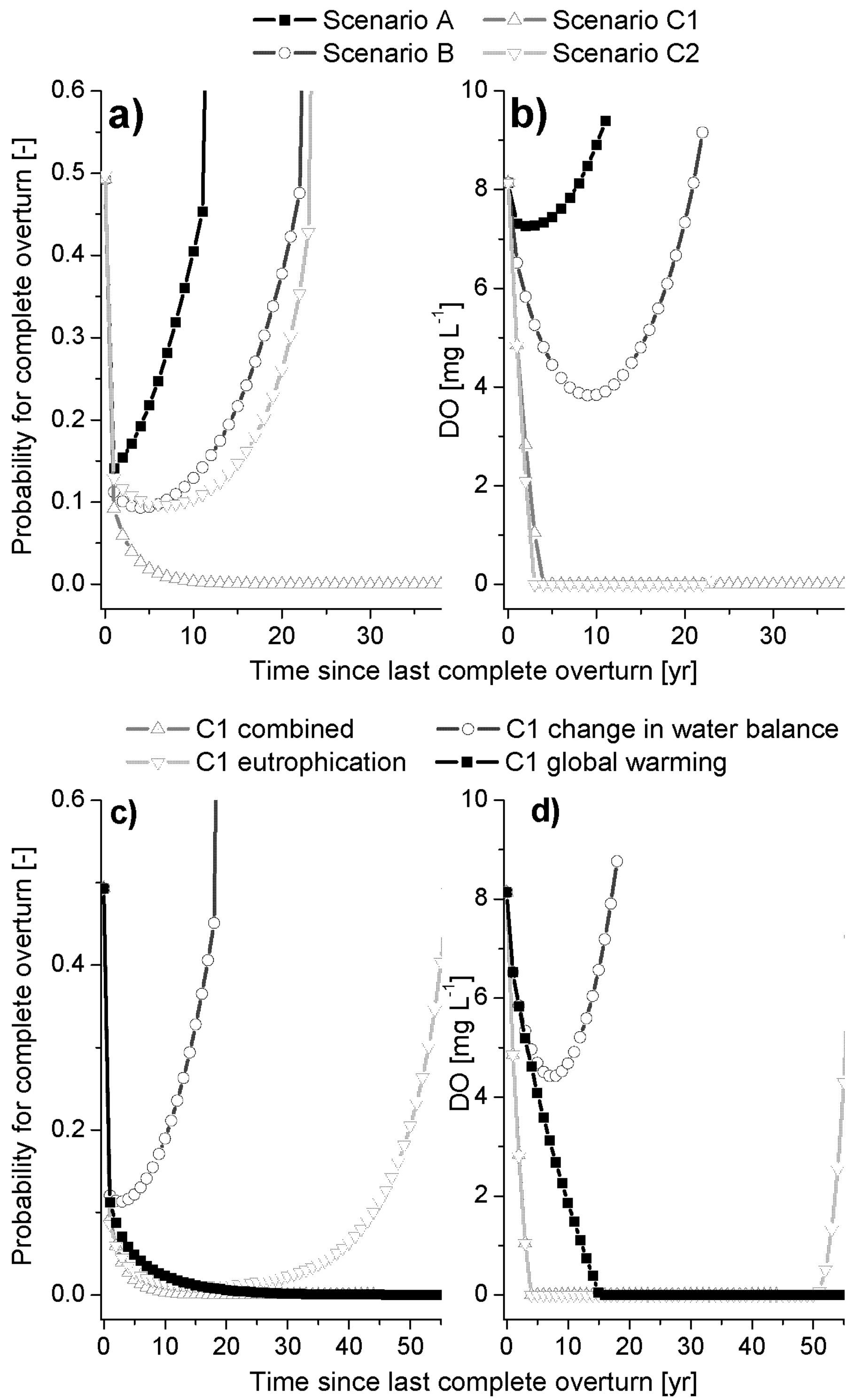

Figure 3.14: Probability of complete overturn and DO concentration in bottom layer BOT. (a) and (b) Effect of the different scenarios in Table 3.4. (c) and (d) Comparison of combined and single effects of anticipated future changes in eutrophication, water use and air temperature. Note the different time scale in (c) and (d).

Scenario (B), Present Status Quo – After winter 2003/2004 p_{co} decreases, as a result of increase in S_{BOT} (Figure 3.14a). After five yr, rising diffusive salt losses and smaller temperature gradient lead to a decrease in stability and p_{co}. Then, with T_{BOT} getting closer to T_{MIX} complete overturn gets more and more probable until after 23 yr it would occur even at average $T_{min,MIX}$. Similar to salinity, DO consumption from mineralization is outweighed by diffusive input after nine yr (Figure 3.14b). Thus no anoxia is expected under the status quo, even if several warm winters follow each other.

Scenario (A), Historic Conditions – Under historic conditions (Table 3.4) it was assumed that the 50% reduction in TP concentration is directly transferable to DO consumption, S depletion in MIX and S accumulation in BOT. Compared to status quo the salinity gradient between S_{BOT} and S_{MIX} stays much lower, because of the lack in plunging River Sateska and the smaller input from mineralization. As a result complete overturn would be expected to have occurred at much shorter intervals 100 yr ago (Figure 3.14a). With increasing diffusive flux and reduced DO consumption, DO_{BOT} is only decreasing during the first two years after complete overturn (Figure 3.14b).

Scenario (C1), Future Conditions during Global Warming – As in scenario (A) the doubling in TP was transferred into a doubling in DO consumption, S depletion in the epilimnion and S accumulation in the hypolimnion. The raised salt input to BOT from eutrophication together with an annual increase in T_{MIX} from global warming higher than in T_{BOT} from geothermal sources and diffusion dramatically increases the stability of BOT. If no complete overturn occurs in the first few years BOT will basically turn permanently stratified, as long as the situation stays as outlined in Table 3.4 (Figure 3.14a). Because of increased DO consumption DO_{BOT} would turn anoxic after four years (Figure 3.14b). Figures 3.14c and 3.14d show the effect of the three human impacts separately. Changes in water balance had almost no effect on mixing periodicity and DO availability, as the inflows are small relative to the large lake volume (Table 3.1). A larger impact on DO_{BOT} and S_{BOT} could only be expected from plunging winter inflow of River Sateska, which remains unchanged from scenario (B) to (C). In a separate run for scenario (B) it was found that the diversion of River Sateska reduces p_{co} by ~7 %, because of increased S_{BOT}. Despite its input of DO to BOT the stabilizing effect dominates and leads to a reduction of DO_{BOT} by ~1 mg L^{-1}.

Eutrophication is the main reason for smaller DO_{BOT} in scenario (B) and for fast anoxia observed in scenario (C1) (Figure 3.14d). Moreover, it supports stratification of the water column through increased mineralization, constraining complete overturn for 35 yr (Figure 3.14c). Because of the low stability after 50 yr, BOT could even turn oxic again in the absence of complete overturn (Figures 3.14c and d). However accumulation of organic matter, which is not considered in our simplified model, would further delay this process.

Global warming is the process that could ultimately lead to meromictic conditions due to rapid increase in stability of the water column. Even under current DO mineralization BOT would still turn anoxic after 16 years, because of reduced turbulent diffusive DO supply from MIX (Figure 3.14d). Under increasingly stable conditions the anoxic layer BOT is also expected to grow in thickness.

Scenario (C2), Future Conditions at Higher Temperature – While the expected warming over the next century would reduce the mixing of BOT dramatically it is interesting to note that the situation is completely different if temperature is assumed to be in a new quasi-steady-state equilibrium, 4 °C higher than today (Table 3.4).

Although higher eutrophication supports stratification, p_{co} in (C2) is very similar to (B) (Figure 3.14a) and clearly reduces the effect of increased eutrophication (Figure 3.14c). The reason lies in the thermal expansivity α, which increases by a factor of 2.3 under an increase in water temperatures from 6 to 10 °C at 200 m depth for Lake Ohrid salinity. An increase in α leads in turn to higher T_{MIX} ($R\rho = 1$) (Equation 3.2) and thus renders complete overturn easier. Still anoxia would be reached fastest of all scenarios because of smaller DO solubility with increased temperature in MIX and in plunging River Sateska.

3.6 Conclusions

Current and potential future human activities will have a major impact on the physical characteristics of Lake Ohrid, which in turn will likely affect the renowned endemism of this unique ecosystem.

3.6.1 Endemic Phytoplankton Species

Deep-growing endemic phytoplankton species require high water clarity and sufficient nutrient supply between 20 and 150 m depth. Future water abstraction from surface spring inflows and Lake Prespa would reduce the continuous nutrient input to this depth interval to ~16 % of today's value. However expected eutrophication could compensate for this loss through generally increased phosphorus (P) level and raised release of soluble reactive P from the lake bottom. A much more serious concern is the reduction in water clarity in the surface layers, which is already apparent in trends of Secchi depth measurements. Further eutrophication could lead to a significant decrease in compensation depth by ~14 m relative to historic conditions.

3.6.2 Bottom Dwellers

Endemic bottom dwellers depend upon adequate availability of dissolved oxygen (DO) and thus reach very deep today. While water abstraction has little effect on the overall DO balance, the diversion of River Sateska into Lake Ohrid does affect the lake by the mechanism of deep plunging in winter. However, the resulting salinity input decreases the downward diffusive DO flux into the bottom layer to a larger extent than it replenishes DO by direct transport.

Eutrophication stabilizes the bottom layer through mineralization and increases DO consumption at the sediment. Even a comparably low total P (TP) concentration of 9 mg m^{-3} would extend periods without complete overturn and lead to anoxic conditions in the bottom layer.

A regular increase in temperature from global warming by 0.04 °C yr^{-1} would create an almost completely secluded, permanently stratified bottom layer. This layer will turn anoxic as the exchange with the regularly mixed part of the lake decreases. While this tendency towards lower mixing is in line with findings of other authors the situation at a new 4 °C higher temperature equilibrium is not. Because of an increase in thermal expansivity with temperature even a more frequent complete overturn would be expected.

3.6.3 Local versus Global Human Impacts

Our results show that both local eutrophication and global warming are jeopardizing the endemic species of Lake Ohrid. Indeed, the effects of eutrophication is already apparent. Moreover, the increase to 9 mg TP m^{-3} is far below the concentrations of most central European lakes, and thus it is a conservative estimate. Since eutrophication can be controlled at a local level it is certainly a priority task for lake management, particularly since its negative effects would be intensified by potential global warming.

However, a more detailed, bio-geochemical assessment is necessary to quantify sustainable P loads under different climate scenarios. The results of this study form an excellent basis for such an assessment regarding the physical lake properties and the sensitivity of Lake Ohrid to human impacts.

Acknowledgments

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Chapter 4

Eutrophication of Ancient Lake Ohrid - Global Warming Amplifies Detrimental Effects of Increased Nutrient Inputs

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Abstract

Lake Ohrid in south-eastern Europe is one of the few ancient, long-lived lakes of the world, and contains more than 200 endemic species. Based on integrated monitoring of internal and external nutrient fluxes, a progressing eutrophication was detected that led to a ~3.5-fold increase in phosphorus (P) concentration in the lake over the past century. The lake is fortunately still oligotrophic, with high concentrations of dissolved oxygen (DO) in the deep water that are requisite for the unique endemic bottom fauna. Coupling a physical with a biogeochemical lake model it was found that the hypolimnetic DO is not only very sensitive (i) to changes in anthropogenic P load - via mineralization of organic material - but also (ii) to global warming via decrease of vertical mixing and less frequent complete deep convection. Moreover these two human impacts were shown to amplify each other. In order to keep DO from falling

below currently observed minimal levels - given the predicted atmospheric warming of 0.04 °C yr^{-1} - the P load must be decreased by 50% over the next decades. However, even at such a reduction in P load, anoxia is still expected towards the end of the century if the rate of warming follows predictions.

4.1 Introduction

There are but a few ancient lakes which have provided favorable living conditions for freshwater organisms over time periods far beyond the usual. These ancient systems allowed the persistence and speciation of fauna and flora, acting as literal centers of evolution (Martens et al. 1994). Compared to short-lived lakes, they often developed extensive species endemism (Table 4.1). Most of the lakes in Table 4.1 are potentially threatened by cultural eutrophication (Cohen 1994; Beeton 2002); for Lake Victoria and Lake Biwa even possible species extinction has been reported (Hecky et al. 1994; Seehausen et al. 1997; Tsugeki et al. 2003). Moreover, shifts towards non-endemic species have been observed in the vicinity of polluted inflows to oligotrophic Lake Ohrid (Watzin et al. 2002). To prevent future irreversible losses, it is therefore important to detect trends towards eutrophication in ancient lakes as early as possible to react in time (e.g., Bootsma & Hecky 1993).

Table 4.1: Comparison of Ancient Lakes

Lake	Age	Endemic species described	Residence time[3]	Max. depth	TP	Sources
Unit	10^6 yr	#	yr	m	mg m^{-3}	
Baikal	25-30	982	350	1636	~8[2]	Shimaraev et al. 1994; Martin 1994; Goldman et al. 1996
Tanganyika	~20	632	7000 (1600*)	1470	1.9[1]	Coulter 1994; Järvinen et al. 1999; Bootsma & Hecky 2003
Malawi	>2	~620	650 (450*)	700	9.3[1]	Ribbink 1994; Guildford & Hecky 2000; Bootsma & Hecky 2003
Victoria	0.015-0.75[4]	~240	140	79	77.5	Greenwood 1994; Guildford & Hecky 2000; Bootsma & Hecky 2003
Titicaca	~3	61	660	284	24[2]	Wurtsbaugh et al. 1992; Dejoux 1994; Grove et al. 2003
Biwa	0.4-1	54	5.5	104	9	Nakajima & Nakai 1994
Ohrid	2-3	210	70	289	4.6	Stankovic 1960; updated for selected groups by Jerkovic 1972; Kenk 1978; Gilbert & Hadzisce 1984; Salemaa & Kamaltinov 1994; Michel 1994; Decraemer & Coomans 1994

[1] upper, oxygenated layer
[2] SRP during main mixing in July (L. Baikal) and September (L. Titicaca)
[3] defined as volume per outflow
[4] large range due to different opinions regarding extent of past desiccation events

Once the extent of cultural eutrophication is known, the choice of mitigation measures may be complicated by interactions with other impacts. Of particular concern is the predicted global warming, which will affect a wide range of lakes and may aggravate effects of eutrophication (Matzinger et al. 2006b; Schindler 2006). In summary, appropriate lake management needs to answer the following three questions:

(i) To what extent has eutrophication occurred so far?

(ii) What are the time scales of past changes and of the potential steady-state?

(iii) What are acceptable P loads and are they sensitive to global warming or other expected changes?

However, eutrophication is difficult to identify for most of the unique lakes listed in Table 4.1, due to (a) low phosphorus (P) concentrations with variations in the range of measurement errors and (b) excessively long residence times. Thus it would take decades of high quality measurements to detect eutrophication in the water column. Alternatively, nutrient input can be monitored, however, sensible input assessment again requires an enormous effort and may still be unreliable (Richards & Holloway 1987; Moosmann et al. 2005). Finally, the analysis of sediments provides an option to look into the history of eutrophication, but results can only be interpreted in combination with additional information, such as input measurements or historic pollution records (Schelske & Hodell 1995; Wüest et al., 2006). The scientific challenge contrasts the often limited research resources of the local institutions investigating those lakes of Table 4.1.

In this article we present an example of such an assessment for ancient Lake Ohrid (Macedonia / Albania; Figure 4.1), using results from P monitoring and model simulations. In a preliminary assessment Matzinger et al. (2006b) estimated sensitivities of Lake Ohrid to different human impacts by reviewing their potential effects on physical processes. They found specifically that dissolved oxygen (DO) in the deep water may be not only highly sensitive to eutrophication but also to global warming. DO availability is a key parameter for the predominantly bottom-dwelling endemic organisms of Lake Ohrid (Stankovic 1960). As a result, anoxic conditions would reduce the habitat for many species, in particular the ones which are limited to large depths (e.g., several endemic forms of amphipods, ostracods and oligochaets; Stankovic 1960). Moreover higher trophic state is expected to worsen oxygen conditions in the sediment throughout the lake (Sweerts et al. 1991). The present paper advances by the next two logical steps to quantify the potential severity of worsened DO conditions in Lake Ohrid.

In the first part of the paper the extent and dynamics of eutrophication is assessed. It is based on P, the limiting element in Lake Ohrid, with total phosphorus (TP) concentrations ~4.6 mg-P m^{-3} and molar ratios of total nitrogen TN:TP > 50 (Guildford & Hecky 2000). P measurements in inflows, water column and sediments are used to establish the contemporary lake-internal P balance. In a second step, time scales of past changes and a potential steady state are discussed based on a linear P model to answer questions (i) and (ii). While the results of the monitoring are presented in the paper, details are given in appendix 1 for future reference to a lake system of global importance.

In the second part of the paper, a dynamical model is established based on the P balance (a) to quantify DO availability for different future eutrophication and global warming scenarios, as well as their interactions and (b) to define allowable P loads depending on these human impacts and DO requirements (question iii).

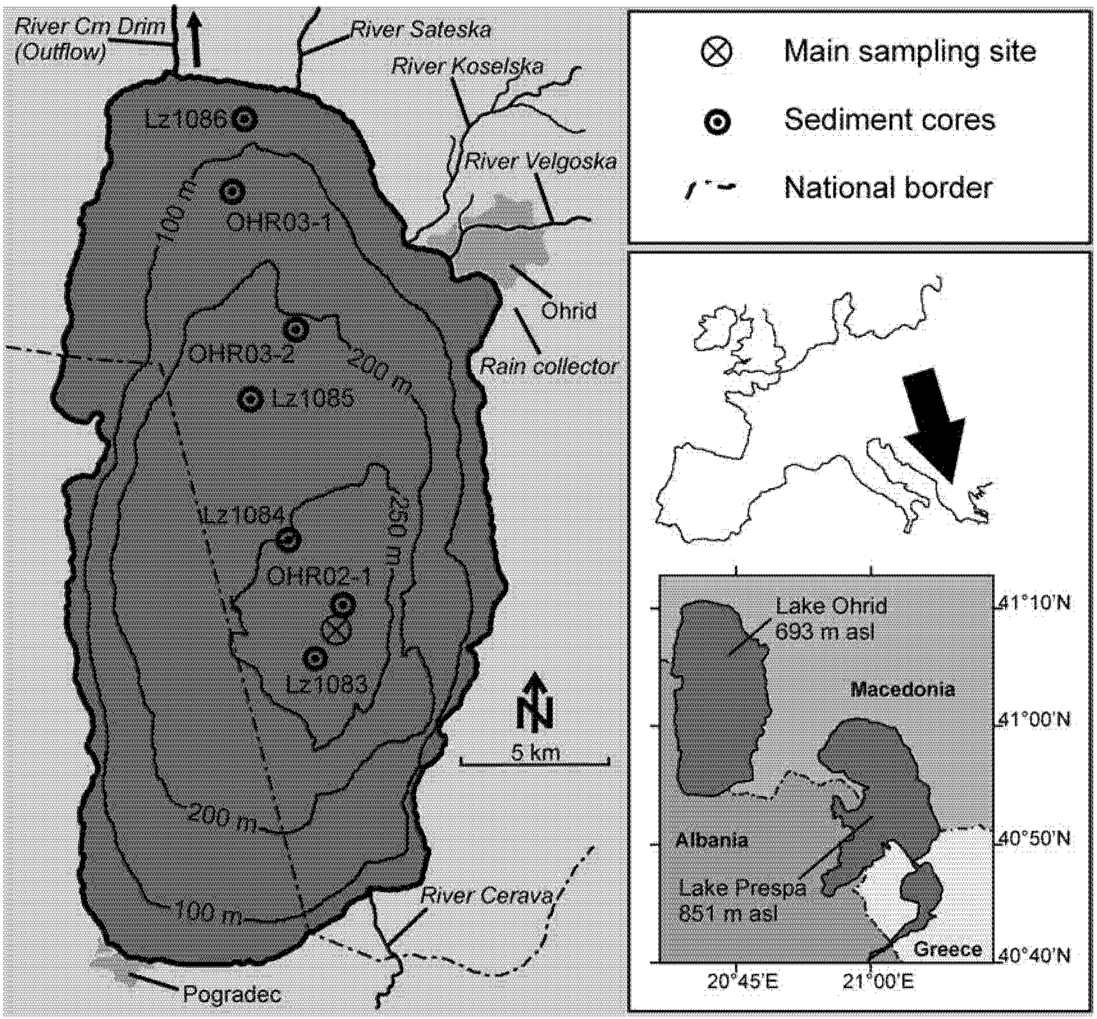

Figure 4.1: Geographical overview. The inset maps indicate the location of Lake Ohrid within Europe and the Macedonian – Albanian – Greek triangle.

4.2 Lake Ohrid

Lake Ohrid is a transboundary lake shared by Macedonia and Albania (Figure 4.1), situated between mountain ranges to the east and west. It is oligotrophic, deep (max. depth ~ 289 m), large (surface area ~ 358 km^2) and one of the most voluminous lakes (~ 55 km^3) in Europe (Table 4.2). The water balance is dominated by inflow from karst aquifers (~50%) with smaller shares from runoff and direct precipitation (Matzinger et al. 2006b). The fraction of river runoff had even been less than 10% before 1962, when River Sateska was deliberately diverted into the lake to reduce siltation in downstream reservoirs (Figure 4.1). The karst aquifers are charged from mountain range precipitation and from Lake Prespa, which has no surface outflow (Figure 4.1; Anovski et al. 1992; Eftimi & Zoto 1997; Matzinger et al. 2006a). Via this underground connection to Lake Prespa the lake catchment extends also to Greece.

Table 4.2: Lake Ohrid Characteristics

Property	Unit	Value
Latitude	° N	41.1
Longitude	° E	20.7
Altitude	m asl	690
Catchment area (including Lake Prespa)	km^2	2600
Surface area	km^2	358
Volume	km^3	54.9
Maximal depth	m	288.7
Average depth	m	155
Average annual inflow	$m^3\ s^{-1}$	38
Average annual outflow	$m^3\ s^{-1}$	25
Mean water residence time	yr	70
Average phosphorus concentration	mg-P m^{-3}	4.6

The top 150 to 200 m water column of Lake Ohrid follows the usual thermal stratification seasonality of deep, temperate lakes, whereas the lower hypolimnion is stably stratified by salinity (Figure 4.2a). The stability due to the salinity gradient – although very weak – allows complete, deep convective mixing (in the following referred to as "complete overturn") only roughly once every seven years during cold winters (Hadzisce, 1966; Matzinger et al. 2006b). Figure 4.3 shows the change in different water properties for such an event in winter 2003/04. In the absence of cold winters, geothermal heat input and turbulent vertical exchange steadily raise hypolimnion temperature, making complete overturn more probable as time progresses (Figure 4.3b). In turn salinity (S) and DO show a seasonal trend because of summer productivity and increased turbulence in winter (Figures 4.3c and 4.3d). DO stays always high throughout the water column with a maximum between 20 and 40 m depth as a result of phytoplankton production (Figure 4.2b).

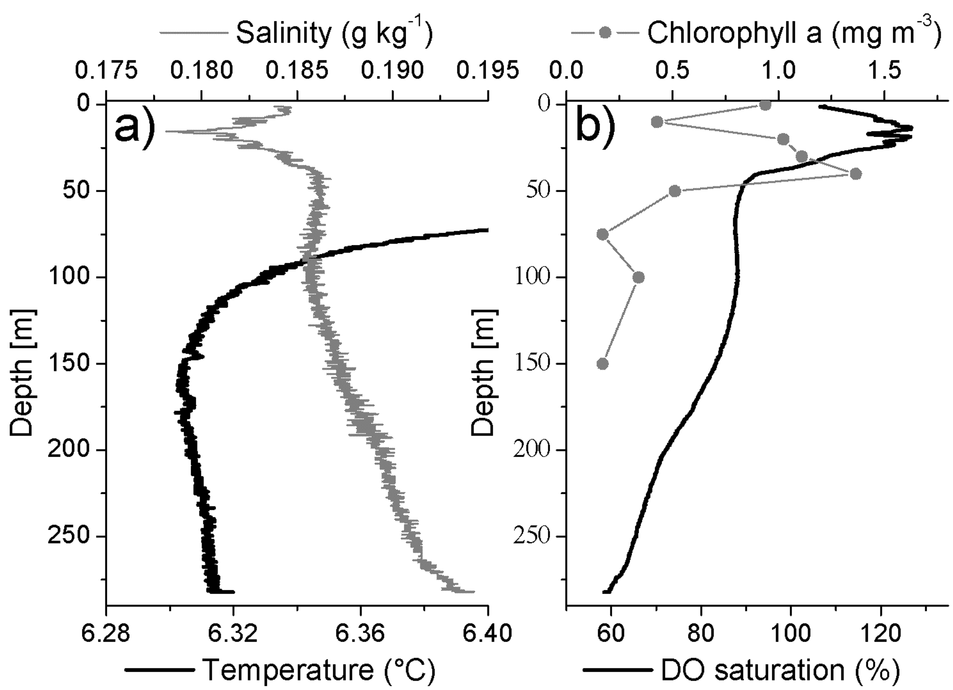

Figure 4.2: Typical summer profiles of salinity, temperature, chlorophyll a and DO saturation from Lake Ohrid (30-Jun-2003)

Clearly the most spectacular quality of the lake is its impressive endemism. Similar to Lake Baikal, it harbors endemic species covering the whole food-chain, from phytoplankton (e.g., *Cyclotella fottii*) over zooplankton (e.g., *Cyclops ochridanus*), cyprinid fish (e.g., *Pachychilon pictus*), to predatory fish (e.g., *Salmo letnica*) and finally

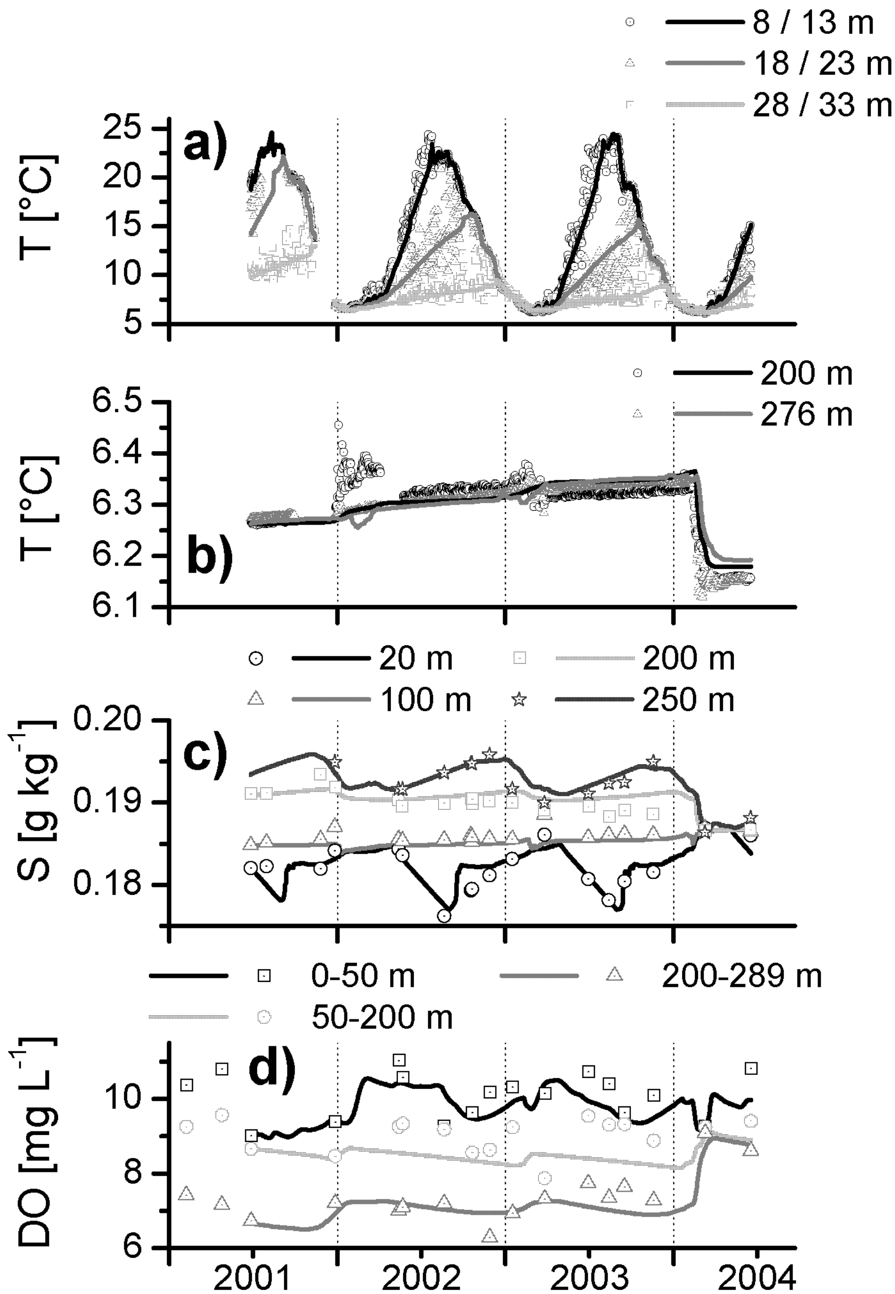

Figure 4.3: Temperature (a and b), salinity (c) and dissolved oxygen (d) for different depths (see legend) over the period of observation. Symbols are measurements (from thermistors for T and from CTD profiles for S and DO), lines are simulated results. The reason for the high temperature signal in 200 m depth in (b) at the beginning of 2002 is unknown; most probably it is connected to subaquatic spring inflows.

its diverse endemic bottom fauna (e.g., *Ochridagammarus solidus*) (review in Salemaa 1994). Whereas most of the endemic species descriptions are based on morphological and ecological characteristics, some recent applications of molecular genetic techniques underline the difference of the Ohrid fauna from common European taxa (Korniushin et al. 2000; Sywula et al. 2003; Sell & Spirkovski 2004).

4.3 Materials and Methods

4.3.1 Monitoring

From 2001 to 2004 water samples were collected from Lake Ohrid, its tributaries and rain and analyzed by standard methods. Plankton biomass estimates were established using chlorophyll a measurements for algae and bio-volume for zooplankton. In addition lake profiles were taken regularly using a Seabird SBE 19 CTD. Also the tributaries water levels were read regularly and occasionally calibrated with discharge measurements from a SEBA Universal Current Meter F1. Sediments were collected from a series of traps from 2001 to 2004, as well as from seven cores, taken in 2002, 2003 and 2004 (Figure 4.1). Analysis involved basic nutrients, as well as ^{137}Cs and ^{210}Pb for dating. A more detailed account of the sampling sites and methodology used is given in appendix 1.

4.3.2 Linear Phosphorus Model

To check the calculated P budget and estimate historic concentrations, the linear model by Vollenweider (1969) was used:

$$\frac{\partial\langle TP\rangle}{\partial t} = \frac{1}{V} \times P_{inp} - \sigma \times \langle TP\rangle - \frac{\beta}{\tau} \times \langle TP\rangle \qquad (4.1)$$

where <TP> [mg m^{-3}] is the volume-averaged concentration of TP, ∂<TP>/∂t [mg m^{-3} yr^{-1}] is the rate of change of <TP>, V [m^3] is the lake volume, τ [yr] is the average water residence time, P_{inp} [mg yr^{-1}] is the annual phosphorus input, net sedimentation is assumed proportional to the total phosphorus content with the sedimentation rate σ [yr^{-1}] and the outflow of TP is expressed by the discharge (V/τ) times the average surface concentration $\beta \times$ <TP> (where $\beta = TP_{surface}$/<TP>). Following a change in P_{inp}, current TP concentration <TP> (t = 0) will approach a new equilibrium <TP> (t = ∞) with <TP> (t) = <TP> (∞) + (<TP> (0) - <TP> (∞))$\times e^{-t \times (\beta/\tau + \sigma)}$. The residence time of P in the lake is then $\tau_P^* = ((\beta/\tau) + \sigma)^{-1}$ and <TP> (∞) = $\tau_P^* \times P_{inp}/V$.

4.3.3 Dynamic Modeling

For the evaluation of different future eutrophication and global warming scenarios the physical lake model by Goudsmit et al. (2002) was combined with an adapted version of the biogeochemical lake model by Omlin et al. (2001), implemented with the simulation software AQUASIM (Reichert 1994). The model includes temperature (T), S, P, DO, total phytoplankton, total zooplankton and dead organic matter as model variables. In terms of biogeochemical processes, primary production, growth of zooplankton, respiration, aerobic mineralization in the water column and at the sediment, as well as death and P-adsorption to settling particles are taken into account. The model was calibrated for observed T and S, measured balances of P, DO and organic matter, as well as historic data for primary production (Figure 4.3). A detailed account on model equations and calibration are given in appendix 2.

4.4 Contemporary Phosphorus Balance

A summarized overview of the following P balance is given in Table 4.3; for more details please refer to appendix 1.

Table 4.3: Phosphorus Balance

Parameter	Method	Total P[1]	Potential bio-available P[1]
		[t-P yr^{-1}]	[t-P yr^{-1}]
External balance			
External P loads from rivers, rain, groundwater	C-Q measurements	32 ± 5	27 ± 5
Internal balance			
P input	from balance	99 ± 15	47 ± 15
P outflow	Surface TP × outflow	3.0 ± 1.8	3.0 ± 1.8
P gross sedimentation	Sediment traps	110 ± 19	58 ± 19
	Sediment core 0-2 cm	128 ± 11	76 ± 11
P net sedimentation	Sediment core 2-4 cm	96 ± 15	44 ± 15
P release from sediment	SRP increase in lake	25 ± 4	25 ± 4
	Sediment core	32 ± 16	32 ± 16

[1] indicated errors are standard deviations among different profiles, cores, sediment trap periods or P load estimates (appendix 1).

4.4.1 External Phosphorus Loads

From regular monitoring of tributaries, groundwater and rain an integrated SRP input of ~27 t-P yr^{-1} was found. Most of the SRP stems from untreated point sources, which is indicated by the dilution effect for increasing runoff (Gächter et al. 2004). The dominance of household point sources is further underlined by the presence of many open sewage channels discharging into lake tributaries. In contrast local, non-intensive agriculture is of minor importance. This is exemplified by River Sateska, bringing significant particulate P loads (~5 t-P yr^{-1}) but negligible SRP.

4.4.2 Phosphorus Outputs

The P balance can be alternatively calculated via its loss terms. P output via the only outflow, the River Crn Drim (Figure 4.1), amounts to ~3 t-P yr^{-1}, calculated from surface concentrations and flow rate. In comparison P loss to the sediment requires a more elaborate analysis.

Sediment traps (Figure 4.4), which were placed in Lake Ohrid over two full annual cycles showed a gross P sedimentation of 110 t-P yr^{-1}. Surprisingly almost 60 % of this amount was found during the less productive winter season (Figure 4.4e), when phytoplankton density in the lake drops to about 10% of its summer maximum (Patceva 2005) and total inorganic carbon (TIC) amounts to 1/3 of the sedimentation in summer (Figure 4.4a). Thus a significant share of the observed P must stem from allochthonous material, particularly during the winter months, when 70% of annual precipitation occurs (Watzin et al. 2002). The low corresponding molar ratios of total organic carbon TOC:TP and TN:TP (Figures 4.4g and 4.4h) indicate as well that a large share of the allochthonous P input is of inorganic, apatite-bound form (Downing & McCauley 1992).

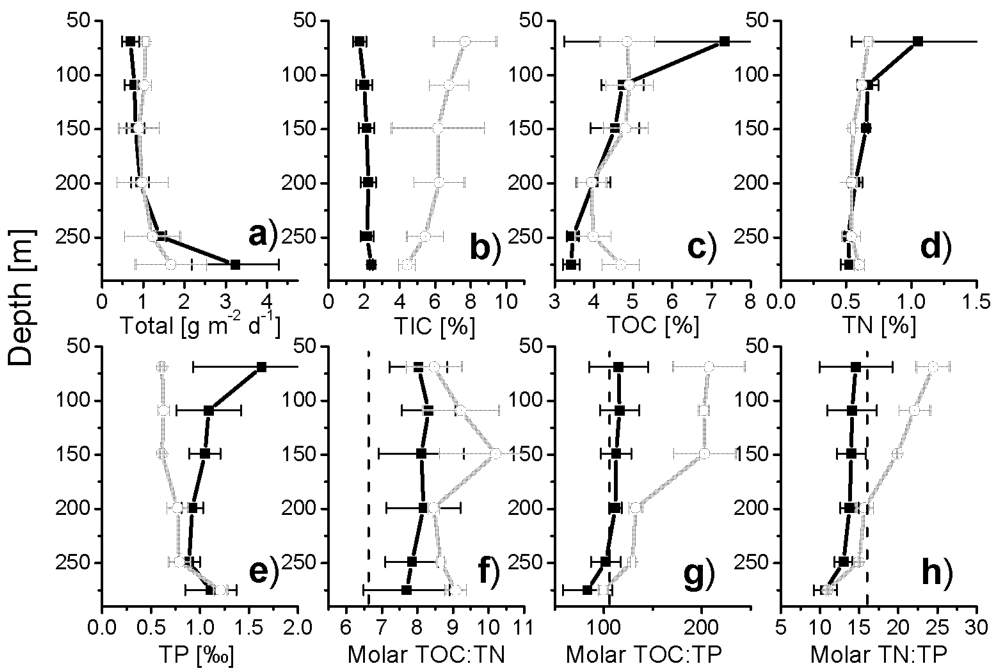

Figure 4.4: Data from sediment traps. Squares for winter season (three periods: 7-Feb-2001 to 25-Apr-2001, 27-Dec-2001 to 14-May-2002, 21-Oct-2002 to 27-Mar-2003), circles for summer season (two periods: 23-May-2002 to 21-Oct-2002, 27-Mar-2003 to 16-Sep-2003). Error bars show standard deviations of averaged periods. (a) shows sedimentation as dry mass per area and time; (b) to (h) describe contents of trap material. Dashed lines in (f) to (h) indicate Redfield ratio (C:N:P = 106:16:1).

Compared with trap results, cores provide an averaged picture of sedimentation, both spacially (Figure 4.5) and temporally. Moreover the sedimentation rate of ~0.09 ± 0.02 cm yr^{-1} from ^{210}Pb dating is in line with the 0.08 cm yr^{-1} reported by Roelofs & Kilham (1983) for cores taken in 1973, indicating that overall sedimentation has not changed during the last century. Thus core results were used to define average fluxes in the following analysis.

They show background P sedimentation of ~66 t-P yr^{-1} below 20 cm core depth (> 220 yr old), far higher than estimated current SRP input (Table 4.3) and therefore mostly consisting of inorganic P. For the organic P cycle, core data was corrected in Figure 4.5e to an estimated natural, bio-available P input of 14 t-P yr^{-1} for a roughly ten times smaller population (appendix 1).

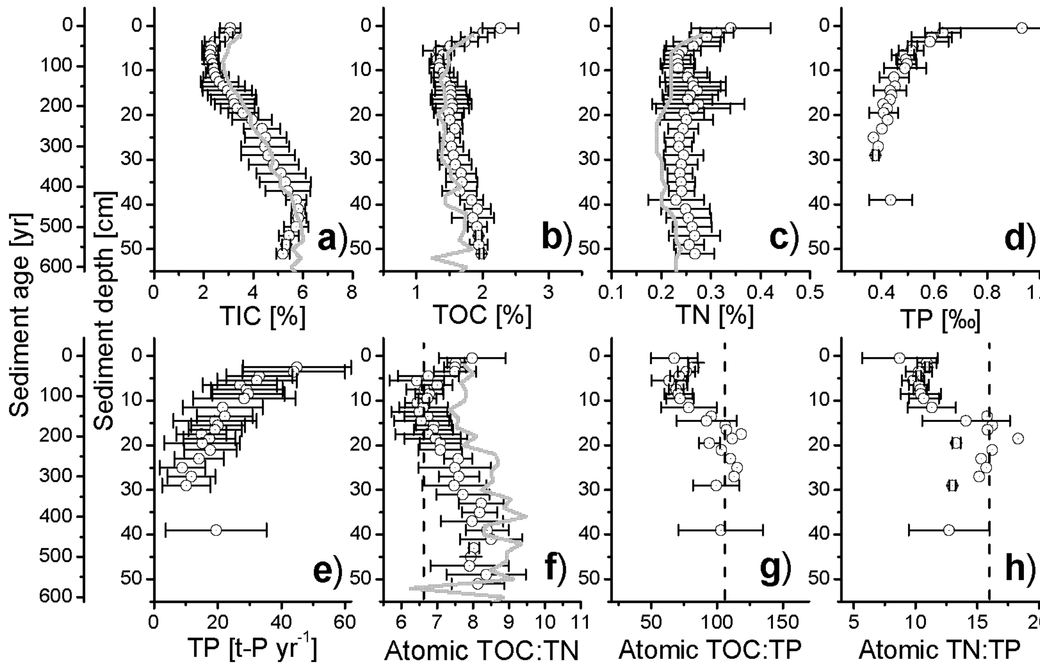

Figure 4.5: Results from sediment cores. Black circles are mean values of three cores taken in 2002 and 2003 (OHR02-1, OHR03-1, OHR03-2, Figure 4.1). Error bars are standard deviations of the three cores, including dating error for (e). Bold grey lines are mean results from four cores taken and analyzed in 2004 (Lz1083 - Lz1086, Figure 4.1; Wagner et al. 2006). (a) to (d) show relative sediment contents, (e) is P accumulation, (f) to (h) are molar ratios. Dashed lines in (f) to (h) are Redfield ratio (C:N:P = 106:16:1). Sediment age is extrapolated from ^{210}Pb dating.

Based on this correction we find a recent gross sedimentation of non-inert P of ~76 t-P yr^{-1} in the top two cm of the cores. Below this top layer between two and four cm sediment depth or ~30 yr before present, when major early diagenetic processes are terminated (Lotter et al. 1997; Urban et al. 1997), recent net P sedimentation is ~44 t-P yr^{-1}. During these three decades about 76 – 44 = 32 t-P yr^{-1} are released to the water column.

This 42% release rate is in the range of observations by Hupfer et al. (1995) and Moosmann et al. (2006). The high background of inorganic P - typical for mountainous, oligotrophic regions - is imported by surface runoff following heavy rain events (Wüest et al. 2006), and by dry deposition (Herut et al. 1999).

4.4.3 P dynamics in the Water Column

Over the period of observation average lake concentrations TP ≈ 4.6 ± 0.8 mg-P m^{-3} and soluble reactive P (SRP) ≈ 2.1 ± 0.5 mg-P m^{-3} were found. Average concentrations and molar fractions in the euphotic zone of TN:TP ≈ 54 and dissolved inorganic N:P ≈ 220 indicate an oligotrophic, P-limited situation (Vollenweider 1968; Guildford & Hecky 2000). It is important to mention that the euphotic zone extends down to 150 m depth due to the exceptionally high water clarity of the lake (Figure 4.2b; Ocevski & Allen 1977; Patceva 2001). Because of the relatively low P concentrations in Lake Ohrid measurement errors are large with 41 and 29 % for TP and SRP, respectively, hence short-term P content variations can not be interpreted.

TP and SRP concentrations in the deep hypolimnion below 150 m are consistently higher than in the euphotic layer above, indicating SRP release from settling particles despite the aerobic water column, as has been observed by Hayes & Phillips (1958) or more recently by Gächter & Müller (2003) and Moosmann et al. (2006). The role of mineralization and consequent SRP release is evidenced by parallel maximum SRP and minimum DO concentrations close to the lake bottom (Figure 4.6). Annual hypolimnetic SRP increase can be estimated from observations to ~21 t-P yr^{-1} and ~28 t-P yr^{-1} for the years 2002 and 2003, respectively.

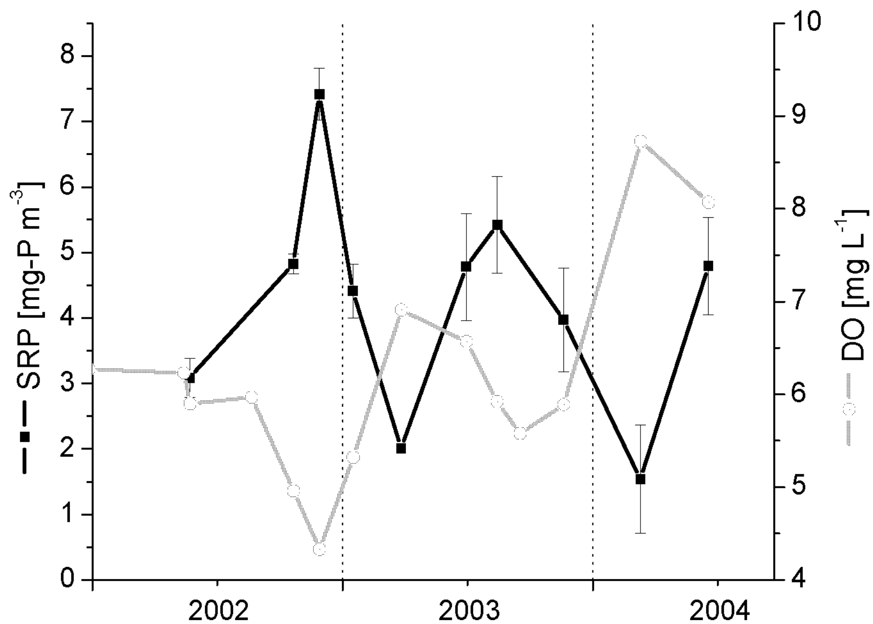

Figure 4.6: Concentrations of SRP and dissolved oxygen (DO) above the lake bottom by ~5 m and ~1 m, respectively. Error bars are measurement errors. Note the different scales for SRP and DO.

4.4.4 Overall Phosphorus Budget

The overall budget (Table 4.3) is consistent regarding gross sedimentation (top of sediment cores versus sediment traps) and P release during early diagenesis (sediment cores versus SRP in water column).

Adding up the P net sedimentation and the P outflow amounts to a annual input of bio-available phosphorus of 44 + 3 = 47 t-P yr^{-1}. The difference of ~20 t-P yr^{-1} from the measured SRP input in Table 4.3 could be explained by "diffusive" point sources. According to Foy et al. (1995) a P load of 20 t-P yr^{-1} is equivalent to the urban P discharge of about 20,000 people. In 1995 the households of 9,000 people living in villages directly at the lake shore and up to 100,000 people throughout the catchment were without connection to any sewage treatment (Ernst Basler and Partners 1995). The same is valid for several hotels and camp sites. Moreover sewage pumps were reported to overflow regularly during heavy rain storms. Thus 20 t-P yr^{-1} of additional P sources seem more than plausible.

4.5 Eutrophication Assessment

4.5.1 Detection of Eutrophication

Population in the Lake Ohrid catchment has more than doubled (+ 100,000 inhabitants) since the late 1940s (Albania: Watzin et al. 2002; Macedonia: Macedonian State Institute for Statistics, personal communication 2004). Moreover up to 50,000 tourists visit the area annually (Ernst Basler and Partners 1995). Thus changes in the nutrient balance of the lake could be expected despite the installation of a limited sewer and treatment system in Macedonia in the 1970s and its improvement since 1995 (Ernst Basler and Partners 1995).

Concurrent increase in TOC, TN and TP in the top sediment (Figures 4.5b to 4.5d) is a clear sign of on-going eutrophication (Edmondson 1991; Schelske & Hodell 1995). The increase can be split into a period of slow eutrophication, which started about 150-200 yr ago and an accelerated phase over the past ~50 yr (Figure 4.5e).

4.5.2 Time Scales

Based on P monitoring and the net sedimentation of organic and bio-available P in Table 4.3, a linear P model can be set up according to equation (4.1). It was assumed that the current TP concentration of ~4.6 mg-P m^{-3} is at quasi-steady-state. Based on the parameters β (≈ 0.84) and σ (≈ 0.18 yr^{-1}) derived from the observations, $\tau_P^* \approx 5.3$ yr was found, which is the average time that bio-available P will remain in the water column before being buried in the sediment or leaving via outflow. After an increase in P input, new equilibrium concentration will be reached to 95% after $3 \times \tau_P^* \approx 16$ yr and will thus lag behind significantly.

Having estimated historic P loads of ~14 t-P yr^{-1} (see *P outputs* above), equation (4.1) can be used to calculate corresponding P concentrations. Using current parameters β and σ we find equilibrium TP concentration of ~1.3 mg-P m^{-3}, 3.5 times less than today.

4.5.3 Answers to Questions (i) and (ii)

Based on the assessment above the two first introductory questions can be answered:

(i) Lake Ohrid is clearly in the process of eutrophication, given the sediment records. Based on a linear model the P concentration in the lake may have increased by a factor of ~3.5, from historic ~1.3 mg-P m^{-3} to current ~4.6 mg-P m^{-3}. The most probable reason for eutrophication is the increase in domestic sources due to growing population.

(ii) Different time scales are important. Eutrophication is relatively slow but ongoing since ~150 to 200 yr. Since the late 1940s its speed seems to have accelerated. Average P residence time is ~5.3 yr. As a result it takes many years before increased P inputs can be detected.

4.6 Definition of Sustainable Phosphorus Load

4.6.1 Scenario Development

Given its importance for the endemic fauna of Lake Ohrid, hypolimnetic DO is applied for the definition of sustainable P load. However, it is unknown which level of DO might be critical for the endemic species of Lake Ohrid. During the period of observation average DO below 200 m, referred to as DO_{hypo} in the following, never dropped below ~6.2 mg L^{-1} (Figure 4.3d), which seems to be sufficient for profundal bottom fauna and deep-living fish species (pers. comm. S. Trajanovski and Z. Spirkovski). Directly above the lake bottom DO levels down to 4.3 mg L^{-1} have occurred over short time periods (Figure 4.6). However the following analysis concentrates on DO_{hypo}, because the phenomenon concerns only a small volume of the lake.

Table 4.4: Simulated Scenarios and their Effects on Lake Ohrid

Scenario		Results				
Global warming	P load	z_{mix}[1]	K_z at 200 m depth [2]	z above which DO > 6.2 mg L^{-1} [2,5]	average TP[2,3]	Gross primary production[2]
[°C yr^{-1}]	[% of status quo]	[m]	[cm^2 s^{-1}]	[m]	[mg-P m^{-3}]	[% of status quo]
0[4]	50	289	2.6	289	1.0 / 2.2	70
	100[4]	**289**	**2.2**	**288**	**1.8 / 4.6**	**100**
	200	204	1.5	198	5.6 / 13.9	123
0.01	50	289	2.1	289	1.0 / 2.5	71
	100	247	1.8	257	1.8 / 5.2	101
	200	97	1.1	102	4.6 / 17.3	129
0.02	50	125	1.3	289	1.0 / 2.9	71
	100	89	1.1	114	1.7 / 7.1	99
	200	76	0.8	84	3.5 / 21	130
0.04	50	71	0.7	92	0.9 / 5.4	67
	100	64	0.6	76	1.5 / 11.3	94
	200	58	0.5	69	2.7 / 23.4	129

1 maximum convective mixing depth in 2067
2 averaged from 2053-2067
3 first value: 0-50 m, second value: 0-289 m
4 bold numbers are status quo
5 6.2 mg L^{-1} is the minimal observed (2001 – 2004) mean DO level below 200 m depth

DO_{hypo} in Lake Ohrid has been shown to be sensitive to changes in lake stratification due to global warming (Matzinger et al. 2006b). Thus several scenarios were used for the next decades from no warming to the expected 0.04 °C yr^{-1} increase in air temperatures for the Balkan Peninsula (Table 4.4), based on a two- to three-fold increase in atmospheric CO_2 level over the next century (Giorgi et al. 2004, Räisänen et al. 2004). Following model results by Mortsch & Quinn (1996) and observations by Livingstone & Lotter (1998) increase in air temperature was directly transferred to the surface of Lake Ohrid and its tributaries. Global warming is expected to affect lake mixing through enhanced stratification during the warming process and, in contrast, stronger convective mixing because of increasing thermal expansivity with water temperature (Matzinger et al. 2006b). Reduced vertical mixing will lead to a decrease in upward transport of SRP and thus lake productivity, as was reported for Lake Tanganyika by O'Reilly et al. (2003). Consequently DO_{hypo} would be enhanced by lower lake productivity and sedimentation of organic matter but limited by reduced downward flux of DO. Finally, higher temperatures will positively affect biological processes, such as algal growth and decomposition of organic matter but reduce DO solubility in water.

Unlike global warming, lake eutrophication can be controlled by local measures. Thus both increased, as well as decreased anthropogenic P loads, relative to the current situation, are considered in separate scenarios (Table 4.4). Changes in P input are expected to affect the entire biogeochemical cycle and thus DO production in the trophogenic layer as well as DO consumption at the lake bottom. Moreover increased salt transfer to the hypolimnion is anticipated from eutrophication via mineralization of settled organic matter and calcite. This salt transfer was assumed to be changing linearly with gross P-sedimentation.

It is not clear, a-priori, which of the competing processes above dominate under different boundary conditions. To test their relative importance the above scenarios are discussed in the following, based on the results of a coupled physical, biogeochemical lake model, calibrated to observations (Figure 4.3; appendix 2). The term "status quo" is used to refer to current P loads and no warming, although an atmospheric warming of ~0.006 °C yr^{-1} has been observed over the past decades (IPCC 2001).

4.6.2 Vertical Mixing and Stratification

Vertical exchange through occasional complete overturns and the exchange between the annually mixed layer – stretching as deep as 200 m – with the lower, stratified, hypolimnion are crucial for biogeochemical cycling. Vertical mixing at the 200 m boundary is about one order of magnitude larger from December to March (Diffusion coefficient $K_{z,win} \approx 2.5$ cm^2 s^{-1}) compared to strong stratification from April to November ($K_{z,sum} \approx 0.25$ cm^2 s^{-1}), in agreement with indirect measurements by Matzinger et al. (2006b). Under status quo complete overturn is occurring roughly once every seven years (Hadzisce 1966; Matzinger et al. 2006b). Consequently the most probable situation was simulated with regular complete overturns at seven year intervals, based on observed meteorological forcing (Figure 4.7).

The vertical mixing pattern changes significantly for the different scenarios in Table 4.4. At an atmospheric warming of $T_t \approx 0.04$ °C yr^{-1} temperature and salinity gradients increase with time rendering stratification more and more stable (Figure 4.7b). As a result the water body below ~64 m is basically secluded from the regularly-mixed top layer after 60 yr of simulation, even in comparably cool winters. The exchange between this upper layer and the hypolimnion – important for transport of nutrients and DO – is on average more than three times lower compared to status quo (Table 4.4). It is interesting that hypolimnetic mixing does not decrease linearly with increasing atmospheric warming rates T_t. The strongest change occurs between $T_t = 0.01$ and 0.02 °C yr^{-1}, where K_z is reduced by 50% (Table 4.4) and maximal convective mixing depth decreases from ~250 m to merely ~90 m. Up to a certain warming rate, obviously between 0.01 and 0.02 °C yr^{-1} for Lake Ohrid, bottom water temperature can cope with the pace through geothermal heating and vertical exchange. Once T_t goes beyond that threshold, bottom temperature lags behind leading to an increase in stratification with time. Thus the warming rate rather than the absolute temperature increase is crucial for the extent of deep water isolation.

Under predicted T_t, eutrophication is of secondary importance for vertical mixing. However at T_t = 0.01 °C yr^{-1}, eutrophication-induced salt input from mineralization would lead to a deep water isolation of similar extent as an increase to T_t = 0.02 °C yr^{-1}.

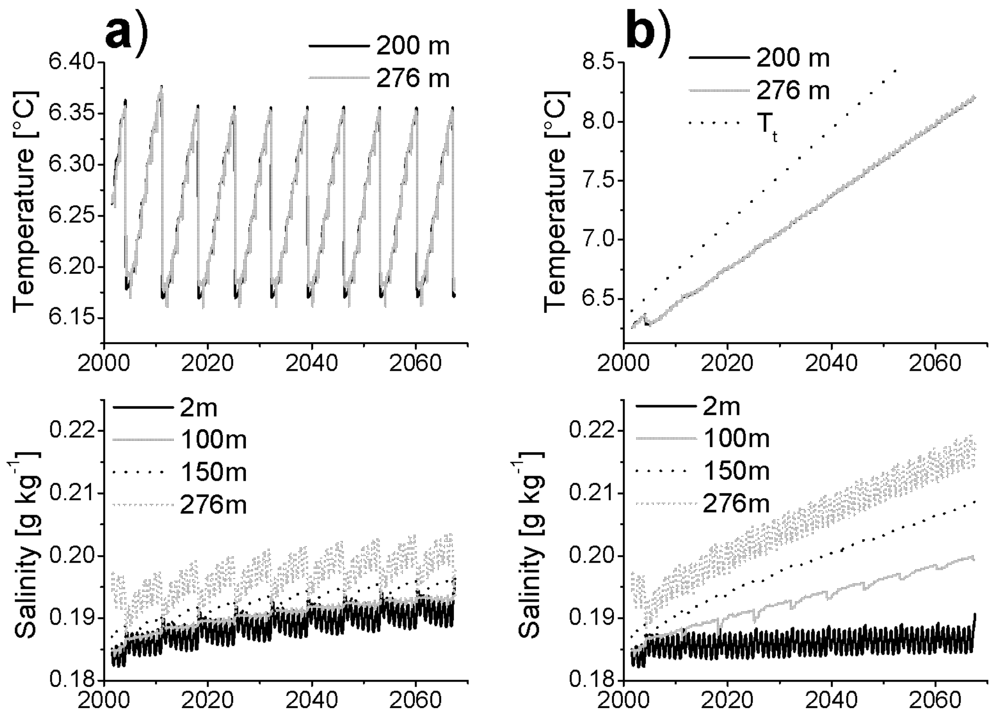

Figure 4.7: Bottom temperature and salinity simulated by k-ε model for different depths (see legend). (a) Status quo. (b) Temperature increase of T_t = 0.04 °C yr^{-1} and status quo for P load; T_t is shown for comparison with simulated *in-situ* temperatures.

4.6.3 In-situ Biogeochemical Processes

As for the physical parameters an excellent agreement of the model was found with observed P and TOC balances, primary productivity and DO concentrations (Figure 4.3; appendix 2). In the years without complete overturn DO_{hypo} continuously decreases, while total dissolved phosphorus (TDP) is increasing due to release during mineralization (Figures 4.8a and 4.8b). After complete overturn both DO and TDP are distributed almost homogeneously throughout the water column. As a result productivity in the trophogenic layer increases, which consequently leads to high sedimentation of organic particles and consumption of DO in the hypolimnion. It is interesting to note that raised primary productivity after complete overturns can be seen very well in simulated DO_{hypo}, sedimentation of organic matter or zooplankton biomass (Figures 4.8a and 4.8d). However TP concentration shows but minor fluctuations and phytoplankton abundance is not a reliable indicator at all (Figures 4.8b and 4.8c).

The results of the different scenarios are shown in Table 4.4, based on average simulation results from 2053-2067. Whereas the influence of P load on average lake TP

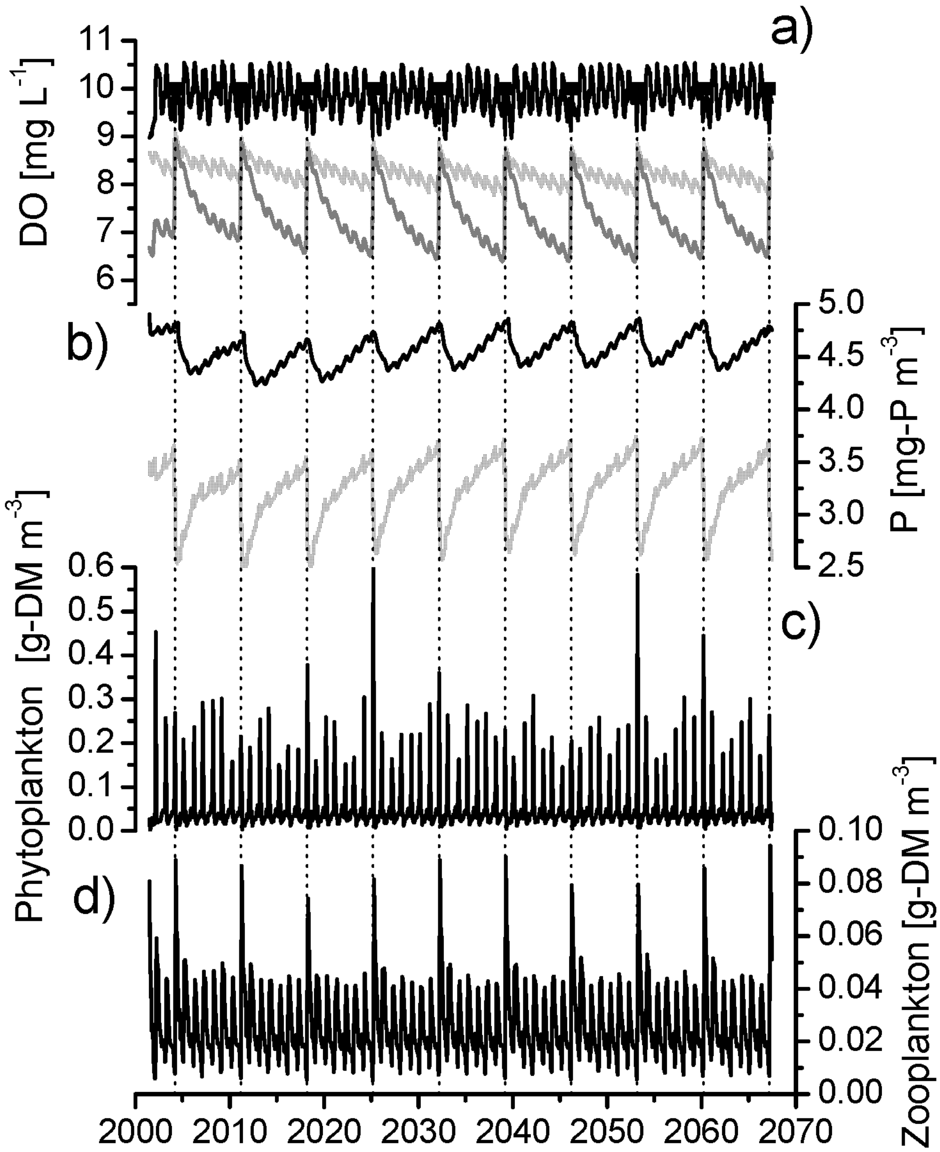

Figure 4.8: Simulation results for status quo over a 66-year period. (a) Averaged DO concentration, 0-50 m (black line), 50-200 m (light grey line), 200-289 m (dark grey line). (b) Averaged TP concentration (black line) and total dissolved phosphorus (light grey line). (c and d) plankton biomass. Dotted, vertical lines indicate years of complete overturn.

is straightforward, the simulated TP increase with global warming is less obvious. It was found to be mainly the result of reduced mixing (Table 4.4), which leads to TDP accumulation in the hypolimnion. Despite this P accumulation, TDP is depleted in the top 50 m with increasing temperature, because of reduced winter P flux from below (Table 4.4). As a result smaller primary productivity would be expected in P-limited Lake Ohrid due to global warming. However Table 4.4 shows that primary productivity is almost exclusively depending on the P load, irrespective of the extent of warming. The reason lies in the temperature dependence of phytoplankton growth and microbiological P recycling, which in the case of Lake Ohrid, are almost exactly compensating the effect of reduced mixing.

DO_{hypo}, the goal variable for the endemic bottom fauna, is particularly interesting, as it is influenced by net system productivity via settling organic matter, as well as by vertical exchange through mixing. Thus both P loads and global warming are expected to be key boundary conditions. Indeed the depth $z_{DO>DOref}$, above which DO is higher than DO_{ref} = 6.2 mg L^{-1}, was found to decrease with increasing extent of both eutrophication and warming (Table 4.4). However $z_{DO>DOref}$ does not decrease linearly but switches between two main states. A first, where basically the whole lake is above DO_{ref} ($z_{DO>DOref}$ = 250-289 m) and a second, where only the top third of the water column ($z_{DO>DOref}$ = 70-114 m) fulfils this requirement. Under status quo P load, DO switches states between T_t of 0.01 °C yr^{-1} and 0.02 °C yr^{-1}. However a similar switch in DO occurs between T_t of 0.02 °C yr^{-1} and 0.04 °C yr^{-1} for a 50% reduction of P loads but already at 0 or 0.01 °C yr^{-1} for doubled P loads. The worst case scenario – an air temperature increase of 0.04 °C yr^{-1} coupled with a doubling of anthropogenic P input – would lead to DO < DO_{ref} below 69 m depth and practically anoxic conditions below 110 m. In that case the sediment area for which DO > DO_{ref} would be reduced by ~280 km^2 or 80 %. The dependency of DO availability on external P loads and global atmospheric warming is summarized in Figure 4.9. The DO-isolines are clearly not equidistant, which further underlines non-linearity with forcing.

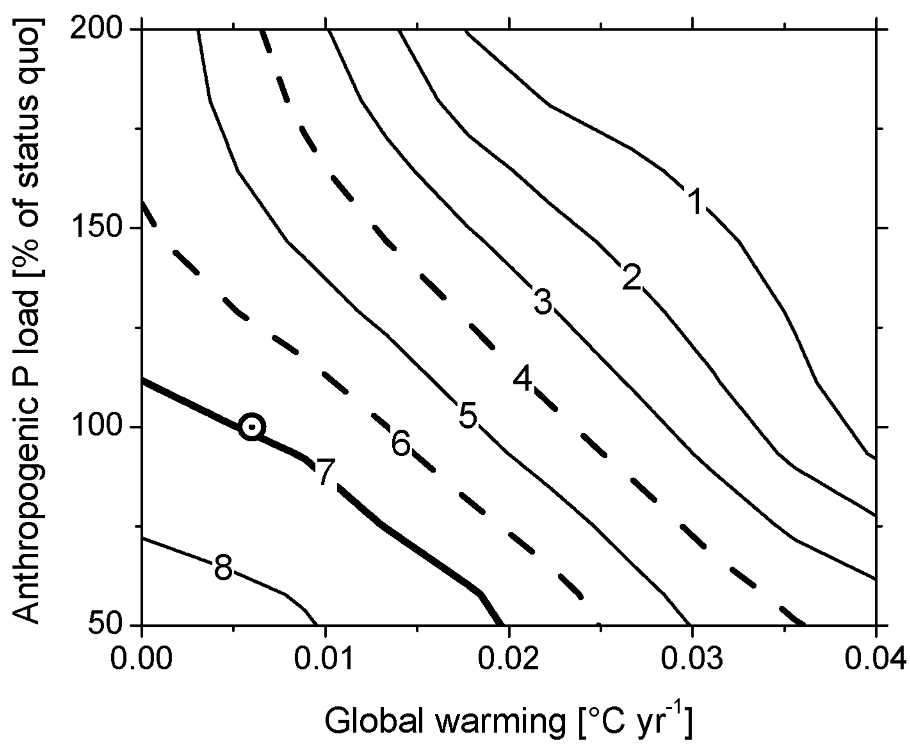

Figure 4.9: Dependence of DO availability on anthropogenic P load and global (atmospheric) warming. DO [mg L^{-1}] values (numbers in plot) are volume-weighted averages below 200 m depth for the simulated period 2053-2067. Bold line indicates current mean DO (~7 mg L^{-1}), dashed lines are minimal observed mean DO (~6 mg L^{-1}) and minimal overall DO, measured directly over lake bottom (~4 mg L^{-1}), all from 2001-2004. Circle represents current situation with observed temperature increase ~0.006 °C yr^{-1} from the past decades (IPCC 2001).

From a lake management point of view the findings imply that P loads must be adapted to the extent of global warming in order to maintain oxic conditions in the hypolimnion. If air warming exceeds 0.01 °C yr^{-1} over the next decades, a reduction in anthropogenic P load is necessary to keep DO_{hypo} above the observed minimum of 6.2 mg L^{-1}. However with predicted T_t = 0.04 °C yr^{-1}, anthropogenic P load has to be reduced by at least 50 % to find DO_{hypo} just around the minimum observed bottom concentration of ~4 mg L^{-1} after ~60 yr (Figure 4.9).

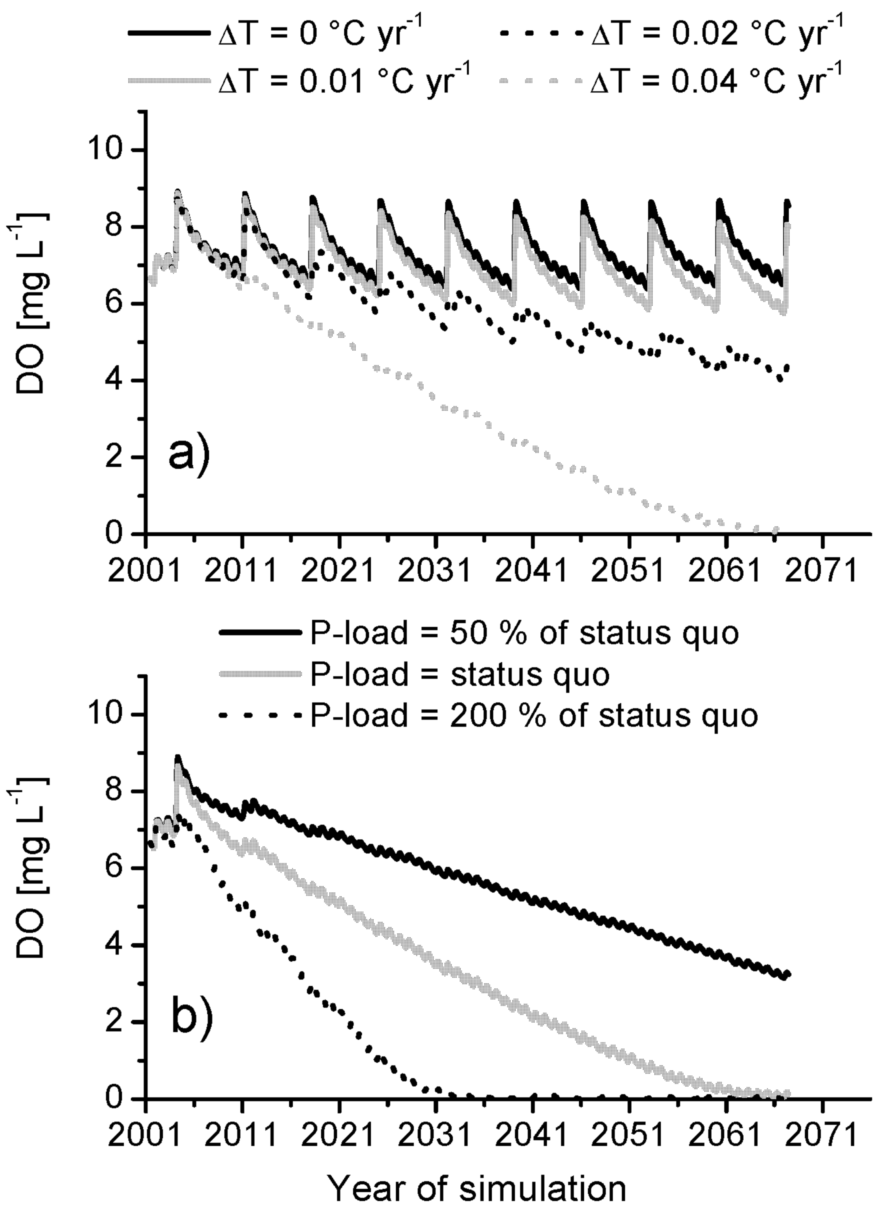

Figure 4.10: Simulated hypolimnetic DO (200 – 289 m) versus time. (a) Global warming scenarios at status quo P load. (b) Different P loads with 0.04 °C yr^{-1} atmospheric warming.

Another important aspect, not shown in Figure 4.9, is the temporal dynamics of DO_{hypo}. While DO_{hypo} approaches equilibrium under status quo and for moderate warming scenarios, it simply decreases for T_t = 0.04 °C yr^{-1} (Figure 4.10a). By reducing the P load only the speed of this decrease can be influenced (Figure 4.10b); however increasing hypolimnion stratification (Figure 4.7b) causes anoxia eventually. In the long run anoxia can only be prevented if global warming remains below the projected rate (Figure 4.10b).

4.6.4 Answer to Question (iii)

Lake Ohrid is highly sensitive to increased P loads, as well as global warming. In particular DO_{hypo} is reduced both by increased productivity from higher P loads, as well as reduced mixing due to global warming. As a result sustainable P load cannot be defined by a constant but has to be expressed as a function of global warming. A reduction of current anthropogenic P loads by 50% must be achieved to keep most of the lake above 4 mg L^{-1} for the next decades.

4.7 Discussion

4.7.1 Lake Ohrid

The analysis of Lake Ohrid nutrient balance clearly points to an eutrophication process, which has led to more than a three-fold increase in average phosphorus (P) concentration over the past century. Although Lake Ohrid is a slow-reacting (P-residence time ~5 yr) and oligotrophic system, the ongoing eutrophication was traced and quantified with low-cost monitoring, thanks to a combination of information on river inputs, lake concentrations, sediment cores and population development in the catchment.

Based on simulation results, increase in P loads to Lake Ohrid leads to higher lake productivity and consequently higher sediment fluxes and mineralization. As a secondary effect water column stratification is stabilized, because of an increase in salt transfer to the hypolimnion by mineralization. However mixing is reduced to a much larger extent by anticipated global warming, since the temperature of the deep water lags behind, increasing the density difference to the surface layer.

Both processes – higher mineralization at the sediment and reduced mixing of the water column – lead to a decrease in dissolved oxygen (DO) in the deep water. If air temperatures increase by 4 °C over the next century as predicted, current anthropogenic P load would have to be reduced by at least 50 % to maintain sufficient oxygen conditions for the endemic bottom fauna for the next decades.

Lake management is challenged to reduce P loads to Lake Ohrid and keep them on a low level, as global warming seems to be already occurring and is likely to accelerate. Good steps in that direction have been taken in Macedonia over the past 5 years by extending the sewerage system and limiting the P content in washing agents (Ernst Basler and Partners 1995; G. Traub, personal communication).

Time scales of expected changes are relatively long, on the order of decades: e.g., even at worst case scenario of 0.04 °C yr^{-1} atmospheric warming and doubling of anthropogenic P-input, P concentration in Lake Ohrid would increase at a rate of only 0.4 mg-P m^{-3} yr^{-1}. As a result long-term, regular monitoring of basic parameters is a necessity to track such slow changes and plan or evaluate protection measures. Still, P makes a sensible monitoring parameter as it reacts primarily to higher P loads.

Temperature (T) on the other hand allows the assessment of the effect of global warming both in terms of absolute T, as well as the potential isolation of deep layers. Finally DO is the most complete parameter, as it is most sensitive to both global warming and higher P input. For Lake Ohrid a long-term monitoring of the three parameters DO, P and T, would allow detection of changes, as well as the evaluation of their main causes. While the monitoring of those "simple" parameters is suggested, one should not forget that the reaction of the endemic species, the main treasure of this unique lake, cannot be anticipated and thus their observation should not be neglected.

4.7.2 General Implications

While local human pressures may differ, global atmospheric warming would affect most deep lakes by reducing vertical exchange. Deep water isolation would be expressed most severely in lakes without alternative deep water renewal, such as turbidity-driven, plunging rivers. Regarding biogeochemical cycling we found that lakes are most sensitive to the rate of warming via vertical mixing, whereas higher temperature showed only secondary effects via biological processes. This phenomenon is very important when discussing temporal stability of the ancient lakes of Table 4.1. These lakes have certainly experienced warmer, as well as cooler periods than today over their long history; however, predicted global warming rates for the next century may be unprecedented regarding the past million years (Petit et al. 1999; Bintanja et al. 2005).

Many special lakes, such as the ones in Table 4.1, are equally clean as Lake Ohrid and no urgent eutrophication control measures seem necessary. Nevertheless our analysis shows that it is important to react in time, (a) when dealing with slow reacting systems and (b) because anticipated global atmospheric changes will amplify effects of eutrophication. The presented, analysis is a first important step to assess this interaction. It is suggested that similar monitoring programs are established for other precious ancient lakes.

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Appendix 1: Monitoring Results

The following appendix gives a detailed account of sampling sites, methodology and results of the monitoring program carried out on Lake Ohrid from 2001 to 2004. The presented data aims to support future work on this globally unique lake system.

A1.1 Materials and Methods

Water Analysis – Regular water samples were taken from Lake Ohrid, its tributaries and collected rain water from 2001/02 to 2004 (Figure 4.1, Table A1.1). Samples were stored in new or acid-rinsed plastic bottles and cooled for transport. Total phosphorus (TP), total nitrogen (TN), soluble reactive phosphorus (SRP) and nitrate (NO_3^-) were analyzed colorimetrically using standard analytical methods (DEW, 1996). Mean measurement errors were 1.9 mg-P m^{-3} for TP, 28 mg-N m^{-3} for TN, 0.6 mg-P m^{-3} for SRP and 4 mg-N m^{-3} for NO_3^-.

In addition, profiles were taken in Lake Ohrid with a Seabird SBE 19 CTD. Parameters included water temperature, in-situ conductivity κ_T, pH and dissolved oxygen (DO). Based on an analysis of major ionic composition of Lake Ohrid water (IC 690 equipped with a Super-Sep column for cations; IC 733 with 753 suppression module for anions, all Metrohm, Switzerland; methods in Weiss 2004), κ_T values were transformed to conductivity at 20°C and to salinity (Wüest et al. 1996). CTD DO values were calibrated using the Winkler method.

Plankton Sampling – Plankton samples for quantitative analysis were taken with Niskin bottles covering the whole water column of Lake Ohrid down to 150 m (Table A1.1). Species were identified and counted under the microscope. Phytoplankton biomass was calculated from photometric chlorophyll-a analyses, using the relationship proposed by Marshall & Peters (1989) (detailed methods in Patceva 2005). Zooplankton biomass was based on detailed assessment of numbers and biovolumes of all major zooplankton species and their different stages (*Eudiaptomus gracilis*, *Arctodiaptomus steindachneri*, *Cyclops ochridanus*, *Mesocyclops leuckarti*, *Daphnia pulicaria*) for the year 2000 (detailed methods in Guseska 2003).

Sediment Analysis – Three sediment cores OHR03-1, OHR03-2 and OHR02-1 were retrieved from 122, 202 and 280 m depth, along the north-south axis of the lake (Figure 4.1, Table A1.1), using a gravity corer (Kelts et al. 1986) and subsequently sectioned to 0.5 to 2 cm long vertical segments. For each segment, the water content was measured by weight loss after freeze-drying. TP was measured photometrically after digestion with $K_2S_2O_8$ in an autoclave for 2 hours at 120 °C (DEW 1996). Total carbon (TC) and total nitrogen (TN) were analyzed with a combustion CNS-Analyzer (EuroVector Elemental Analyzer). Total inorganic carbon (TIC) was measured by infrared absorption of CO_2 after acidifying the sample with 3M HCl (Skoog et al. 1996). Total organic carbon (TOC) was calculated as TOC = TC – TIC.

For consistency check, four additional cores Lz1083 to Lz1086 were taken from 270, 250, 232 and 85 m depth in 2004 (Figure 4.1, Table A1.1). Vertical 2 cm segments were equally freeze-dried and analyzed using a VARIO elemental CNS analyzer for TC and TN. TOC content was measured with a Metalyt CS 1000S (ELTRA Corp.) analyzer, after pretreating the sediment with 10% HCl at 80 °C in order to remove carbonate. Finally TIC was calculated indirectly from TOC and TC.

For the dating of the core OHR02-1 ^{137}Cs and ^{210}Pb activities were established from gamma-counting in Ge-Li borehole detectors (Hakanson & Jansson, 1983). As no clear peaks could be identified in the ^{137}Cs profile, only ^{210}Pb was used. In the top 2 cm of the core, the ^{210}Pb activity was practically constant, probably because of bioturbation by benthic organisms. Below the homogeneous layer, the ^{210}Pb signal decreased exponentially to background activity.

Sediment mass flux was calculated using:

$$S_M = (1 - POR) \times sed \times \rho_{sed} \tag{A.1}$$

where S_M [kg m^{-2} yr^{-1}] is sedimentation of dry matter, POR [-] is porosity calculated from the water content, sed [m yr^{-1}] is the sediment accumulation rate from ^{210}Pb dating and ρ_{sed} = 2600 kg m^{-3} is the sediment density from Ohrid cores established by pycnometer. The average S_M in the dated section of the core (top 8 cm, ~100 yr) was used to calculate sedimentation of TP, TN, TOC and TIC, by multiplication with their respective measured mass fractions.

Table A1.1: Monitoring of Lake Ohrid

Site	Sampling period	Number of samples	Parameters
Lake water column (0, 10, 20, 30, 40, 50, 75, 100, 150, 200, 250, 275 m)	2/2001-6/2004	118 23 8 1 40	SRP, TP, DO NO_3^- TN Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-} CTD profiles (~ 0.2 m depth resolution)
Phytoplankton samples (0, 10, 20, 30, 40, 50, 75, 100, 150 m)	3/2001-11/2003	155	Abundance, chlorophyll-a
Zooplankton samples (0, 10, 20, 30, 40, 50, 75, 100, 150 m)	1/2000-12/2000	108	Abundance, bio-volume
Sediment cores OHR02-1, OHR03-1, OHR03-2 (Figure 4.1)	5/2002 & 4/2003	140 76 13	TC, TIC, TN, water content TP ^{137}Cs, ^{210}Pb
Sediment cores Lz1083, Lz1084, Lz1085, Lz1086 (Figure 4.1)	6/2004	151	TC, TOC, TN, water content
Sediment traps at main sampling site (69, 109, 149, 199, 249, 275 m)	2/2001-9/2003	30	TC, TIC, TN, TP
River Velgoska (Figure 4.1)	10/2002-11/2003	3 7 9 304	TP SRP Discharge Water level
River Koselska (Figure 4.1)	10/2002-11/2003	2 6 9 360	TP SRP Discharge Water level
River Sateska (Figure 4.1)	10/2002-5/2004	25 8 25	TP, SRP Discharge Water level
Eight Albanian tributaries	10/2002	8	TP, SRP, Discharge
Rain samples	10/2002-11/2003	24 5	SRP TP

Apart from cores, sediment was collected by six sediment traps at the main sampling site (Figure 4.1, Table A1.1). Traps were emptied twice per year from 2001 to

2003. Before chemical analysis, as for the OHR-cores, sediment mass was determined after drying the samples at 30 °C.

Assessment of Phosphorus Loads **–** For the three main tributaries, Rivers Velgoska, Koselska and Sateska (Figure 4.1), discharge measurements have been performed with a SEBA Universal Current Meter F1. To reach a higher resolution of river discharge, water levels were read regularly and calibrated with occasional discharge measurements (Table A1.1). For the calculation of TP-loads, rating-curves were established using a linear fit for River Velgoska, which is influenced mainly by point sources, and the concentration (C) - discharge (Q) relation $C = k_1/Q + k_2 \times \ln(Q + k_3)$ (Moosmann et al. 2005) for Rivers Koselska and Sateska, which are influenced both by point and diffuse sources. Based on the rating-curves P-loads were calculated and integrated using daily flow readings for River Velgoska and Koselska and average monthly discharge for River Sateska from Ivanova (1974) and the Macedonian Hydrometeorological Institute (unpublished data). For a second estimate of P-loads, measured C-Q pairs since 1996 (from this monitoring, Naumoski (2000) and Veljanoska-Sarafiloska (2002)) were combined and integrated over one year.

Eight Albanian tributaries were sampled on one excursion after several rainy days. Flow was estimated through water speed and cross-sectional river area.

Finally rain samples were collected on the roof of the Hydrobiological Institute in Ohrid using a funneled plastic collector, which was emptied after every rain event (Figure 4.1).

A1.2 Results of Phosphorus Monitoring

External Phosphorus Loads **–** P was monitored in the catchment (a) to quantify P inputs and (b) to identify the main contributors. Inputs are expected from direct precipitation, dry deposition, groundwater inflows, tributaries and diffusive sources, such as agricultural activities and settlements close to the lake shore. The calculated P-loads are summarized in Table A1.2.

In rain samples collected close to the town of Ohrid over one year (Figure 4.1), average SRP of ~8.4 ± 1.6 mg-P m^{-3} was found, which is in the range of sites with minor anthropogenic influence, such as Lake Malawi or north-eastern Crete (Bootsma & Hecky 1993; Markaki et al. 2003). The measured value also contains SRP that may have leached from dry deposition, given the time between rain event and actual sample collection, which was typically several hours (Herut et al. 1999). Few samples on which also TP was measured (though without prior stirring) indicate that the load of inert particulate P is at least one order of magnitude larger than for SRP.

The contribution of groundwater depends on the amount of water stemming from Lake Prespa and the P retention capacity of the underground karst connection. Matzinger et al. (2006a) found SRP ~10.9 ± 2.6 mg-P m^{-3} for the 7.8 $m^3 s^{-1}$ flow from Lake Prespa and SRP ~4.0 ± 0.9 mg-P m^{-3} for the precipitation-fed remainder, which add up to a total groundwater load of ~4.2 t-P yr^{-1}.

Table A1.2: External P-loads to Lake Ohrid

Source	Status quo		Historic situation (> 200 yr ago)	
	Mean annual inflow[1] [$m^3 s^{-1}$]	SRP-load[2] [$t\text{-}P\ yr^{-1}$]	Mean annual inflow[1] [$m^3 s^{-1}$]	P-load [$t\text{-}P\ yr^{-1}$]
Precipitation on lake surface	8.8	2.3 ± 0.4	8.8	2.3
Groundwater inflow	20.2	4.2 ± 0.7[3]	~24.4	4.0 ± 0.6[3]
Tributaries:				
River Velgoska	0.4	5.8 ± 0.4		
River Koselska	1.3	1.0 ± 0.1		
River Sateska	5.5	0.6 ± 0.6 (5.8 ± 0.6 TP)		
River Cerava	0.2	0.3 ± 0.15[4]		
Small creeks from Macedonia	1.0	0.9 ± 0.9		
Albanian tributaries	0.5	12.1 ± 4.0		
Total tributaries	8.9	25.9	3.9	7.2[5]
Total	37.9	32.4	37.1	13.5

[1] from Matzinger et al. (2006b)
[2] errors represent standard deviation among measurements for precipitation and groundwater, and deviations among methodologies for tributaries
[3] from Matzinger et al. (2006a)
[4] based on Naumoski (2000)
[5] including point sources

P-load of River Velgoska is dominated by point sources, as can be observed in P dilution with discharge (Figure A1.1), leading to an extreme increase in SRP over the mere five kilometers from source to mouth (Figure A1.2). In contrast, River Sateska – by far the largest tributary – shows an increase in P with higher flow in Figure A1.1, indicating that leaching and erosion from agricultural soils are important (Gächter et al. 2004). Finally River Koselska is influenced both by agriculture and point sources with increased concentrations at low and high discharge (Figure A1.1). The expected moderate SRP increase along the river is diluted effectively by more pristine tributaries (Figure A1.2).

For all three rivers, differences between P-loads from direct integration and rating-curve estimates were smaller than 10%. These small deviations may be attributed to a reduced influence from agriculture-based pollution compared to central European streams examined by Moosmann et al. (2005). For the remaining Macedonian tributaries, rough estimates are given using average Sateska P concentration and based on measurements for River Cerava (Naumoski 2000). The Albanian catchment consists mainly of small creeks with large seasonal flow variations, some of which are severely polluted. As a result two creeks at the town of Pogradec contribute ~90% of the SRP input from Albanian tributaries. Given this dominance of permanent point sources on the Albanian side, it can be assumed that the measurements after several days of rain reflect the daily household pollution quite accurately.

Gross Sedimentation **–** Settling material has been collected with sediment traps to estimate gross P-sedimentation and to better understand internal lake cycling (Figure 4.4). Sedimentation is surprisingly constant throughout the seasons as well as inter-annually (Figure 4.4a). As the top trap is still in the trophogenic zone and the lowest is subject to focusing, the intermediate four traps were used for the estimation of lake-wide vertical fluxes.

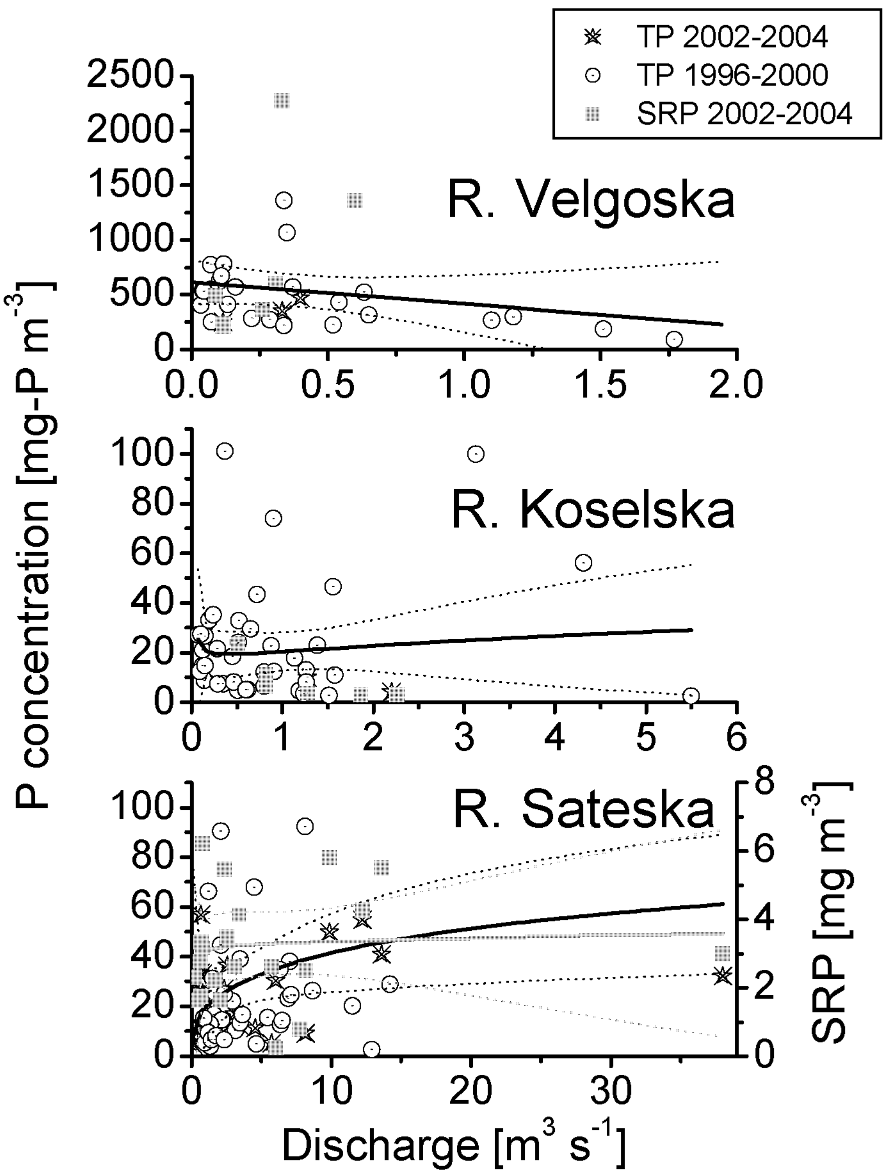

Figure A1.1: Measured TP and SRP concentrations versus discharge of three tributaries (Figure 4.1). Data from 1996 to 2000 are from Naumoski (2000) and Veljanoska-Sarafiloska (2002). For River Sateska SRP concentrations are on a separate axis (right). Bold lines are least-square fitted rating-curves using the concentration (C) - discharge (Q) relation $C = k_1/Q + k_2 \times \ln(Q + k_3)$ and a linear relationship for River Velgoska. Dotted lines are 95% confidence limits.

The three times higher TIC values during summer season (Figure 4.4b) can be explained through increased temperature and photosynthetically-driven calcite precipitation, as described by Hodell et al. (1998). The increased summer productivity is verified by phytoplankton counts and chlorophyll-a measurements; however phytoplankton never drops below 10% of its maximum density (Patceva 2005). This comparably high standing crop can also partly explain the similar summer and winter sedimentation of TOC and TN (Figures 4.4c and 4.4d). However higher TP-sedimentation in winter (Figure 4.4e) is most likely the result of allochthonous material over the winter months, when about 70% of annual precipitation is occurring (Watzin et al. 2002). Organic allochthonous material would increase the TN:TP ratio in the sediment (Downing & McCauley 1992). However, while anticipated P-deficient algal growth is observed in sediment traps in summer with TOC:TP and TN:TP ratios above Redfield, these ratios are significantly lower in winter (Figures 4.4g and 4.4h). These low ratios strongly indicate that the excess P-sedimentation in winter consists mainly of inorganic particles (Downing & McCauley 1992). Such an input of inorganic material also explains the seasonally similar overall sedimentation, despite the higher TIC values in summer (Figures 4.4a and 4.4b). Based on the sediment trap results gross P-sedimentation ~110 t-P yr^{-1} is found, of which 43% settle in summer and 57% in winter.

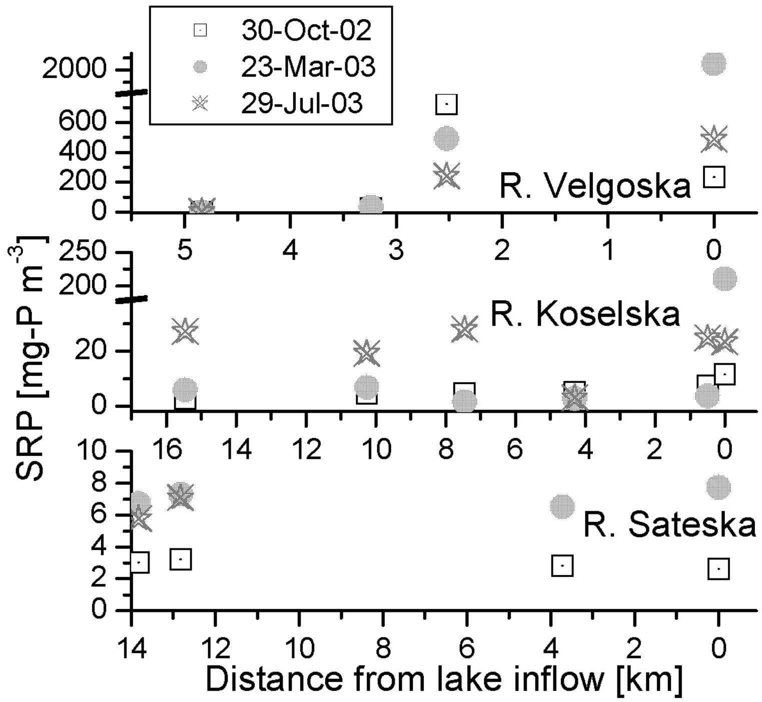

Figure A1.2: Change in SRP concentrations of three tributaries as a function of distance from their mouth (Figure 4.1). Note breaks on y-axis in top two graphs. Missing symbols for River Sateska in July 2003 indicate that river bed was dry at the sampling point.

Net Sedimentation – Net sedimentation estimates are based on the three OHR sediment cores, which have been taken along the north-south axis of Lake Ohrid (Figure 4.1). The results for TIC, TOC and TN contents are in excellent agreement with four additional cores taken and analyzed in 2005 (Figure 4.5), despite the numerous thrusts and folds that have been observed at the lake bottom in a seismic survey (Wagner et al., 2006). As a result the three OHR cores can be assumed to give a representative picture of lake sedimentation.

^{210}Pb dating of core OHR02-1 provided an average sediment accumulation of sed ~0.09 ± 0.02 cm yr^{-1}. This result is in line with the 0.08 cm yr^{-1} reported by Roelofs & Kilham (1983) for cores taken in 1973. Thus 0.09 cm yr^{-1} should be reliable covering at least the topmost 8 cm (~100 yr) of the core, if we assume that ^{210}Pb dating is appropriate for the past ~70 yr (≈ three times half life of ^{210}Pb). Using ^{14}C dating Roelofs & Kilham (1983) found 0.045 cm yr^{-1} for sediment layers beyond 10,000 yr of age, which indicates that no dramatic changes in sed occurred even over long time spans. Gross sedimentation from sediment traps ~1.0 g m^{-2} d^{-1} (Figure 4.4a) would lead to a sed ~0.07 cm yr^{-1}, assuming a water content of ~60 % as found in the top 8 cm of the cores. Thus sediment trap results are consistent with the analysis of the cores.

Total sedimentation (equation A.1) is ~26 % higher in the cores than for the sediment traps consistent with 0.02 cm yr^{-1} difference in sed. In contrast, individual material sedimentation in the top 2 cm of the cores were found on average 30 % lower than in the sediment traps for TIC, TOC and TN, and 16 % higher for TP. The results imply that – in contrast to TOC and TN – release of TP is relatively weak during settling and takes place mainly at the sediment (Figures 4.4c to 4.4e). This is also indicated by the sharp TP decrease within the top sediment layer (Figure 4.5d). The P release during early diagenesis was estimated to be ~32 t-P yr^{-1}, by comparing sedimentation from 0-2 cm with 2-4 cm depth. The 32 t-P yr^{-1} represent a maximum estimate, as it also contains potential eutrophication signal of the past two decades.

Molar TOC:TN:TP ratio stayed almost constant over the past 100 – 200 yr (Figures 4.5f and 4.5g). The abrupt change in TOC:TP and TN:TP at 15-20 cm sediment depth is the result of the increase in TOC and TN with depth of the core (Figures 4.5b and 4.5c). This increase might stem from a change in allochthonous organic material input, but cannot be interpreted based on the available data. However TOC:TP and TN:TP are below Redfield, contrary to expectations for a P-limited lake (Figures 4.5g and 4.5h). The low ratios can be explained by a significant fraction of non-decomposable, possibly apatite-bound P.

P in the Water Column – As expected from sediment results, TP and SRP concentrations in the deep hypolimnion below 150 m are consistently higher than in the euphotic layer above (Figures A1.3b and A1.3c), indicating SRP release from settling particles and the sediment. Indeed, SRP makes up ~60 % of the average difference $TP_{150\text{-}270m} - TP_{0\text{-}150m} \approx 1.7$ mg-P m^{-3}. Annual hypolimnetic SRP increase can be estimated from observations with ~21 t-P yr^{-1} and ~28 t-P yr^{-1} for the years 2002 and 2003, respectively (Figure A1.3b).

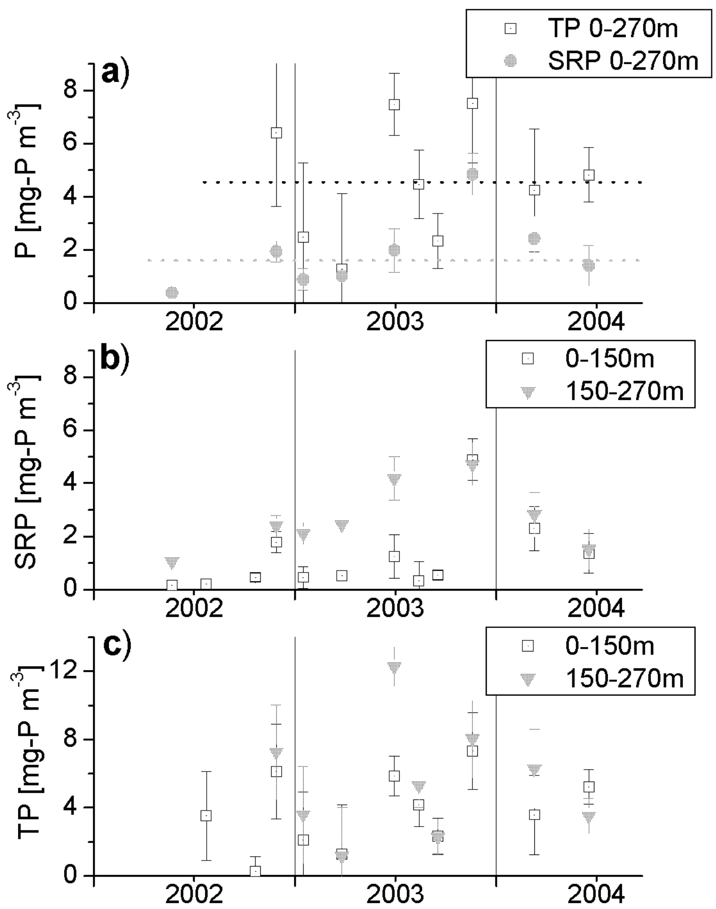

Figure A1.3: Volume weighted mean total phosphorus (TP) and soluble reactive phosphorus (SRP) concentrations for the whole water column in (a), and split up into top 150 m and the hypolimnion in (b) and (c). Dotted lines in (a) show mean concentrations TP ≈ 4.6 ± 0.8 mg m^{-3} and SRP ≈ 2.1 ± 0.5 mg m^{-3}. Error bars are measurement errors.

Appendix 2: Numerical Model Setup

In the following appendix the model approach and calibration is described. Moreover detailed model equations are listed, which allow the use of the same model for interested readers.

A2.1 Approach

A two step procedure was chosen for the evaluation of different future eutrophication and global warming scenarios. Vertical mixing was calculated as turbulent diffusivities K_z [m^2 s^{-1}] based on a physical lake model. The resulting K_z profiles were then used as input for a bio-geochemical model in order to simulate DO concentrations.

For the simulation of physical lake processes the k-ε turbulence model by Goudsmit et al. (2002) was applied. The basic idea behind the k-ε-approach, the most popular two-equation turbulence model, is the combination of the budget of turbulent kinetic energy (k), representing the source of turbulent mixing, with the budget of the dissipation rate (ε). Apart from the classical k sources, shear and buoyancy, Goudsmit et al. (2002) introduced an additional term to account for boundary mixing from internal seiches (Wüest & Lorke 2003). The vertical diffusivities are then calculated (details in Goudsmit et al. 2002) from k, ε, and stratification (N^2). In order to improve model performance during complete lake overturn the model by Goudsmit et al. (2002) was extended with the stability functions proposed by Galperin et al. (1988), which lead to an increase in mixing at unstable stratification. A measured CTD profile from June 2001 served as initial condition. Considered model inputs were water and salt balance (Matzinger et al. 2006b), wind data measured at Ohrid station (US National Climatic Data Center, WMO station 135780), solar radiation (NOAA NCEP-NCAR CDAS-1 reanalysis project described in Kalnay et al. (1996)), water absorption, measured surface temperature and geothermal heat flux (Matzinger et al. 2006b).

For the simulation of bio-geochemical processes, a one-dimensional reaction-advection-diffusion model, implemented in the software package AQUASIM (Reichert 1994) was used. The biogeochemical lake model for Lake Ohrid is a trimmed version of the model established by Omlin et al. (2001) for mesotrophic Lake Zurich. In particular no nitrogen cycle has been included and processes at the sediment were simplified by assuming that a fixed ratio of settling organic material is mineralized. Moreover the light dependency of phytoplankton growth was adapted for the exceptionally clear Lake Ohrid based on photosynthesis measurements in Lake Tahoe (Tilzer and Goldman 1978), including a photoinhibition term simplified from Cullen et al. (1992). The model includes P, DO, total phytoplankton, total zooplankton and dead organic matter as model variables. In terms of processes, primary production, growth of zooplankton, respiration, aerobic mineralization in the water column and at the sediment, death and P-adsorption are taken into account. The simulation period was again 2001 to 2004. Accordingly, water balance, solar radiation, surface temperatures, mixing, as well as the initial condition were directly adapted from the k-ε-approach. The P cycle was implemented using the monitoring data presented in this work.

A2.2 Calibration

The physical lake model was calibrated using temperature and salinity measurements from 2001 to 2004. Measured salt and heat budgets are reproduced very well (Figures 4.3a and 4.3c). Moreover the observed complete overturn in winter 2003/04 is predicted accurately (Figure 4.3b). At the 200 m boundary, k-ε simulation resulted in a relatively low mixing $K_{z,sum} \approx 0.25$ cm² s⁻¹ from April to November and enhanced $K_{z,win} \approx 2.5$ cm² s¹ from December to March. These turbulent diffusion coefficients are in good agreement with estimated $K_{z,sum} \approx 0.35$ cm² s⁻¹ and $K_{z,win} \approx 2.8$ cm² s¹ from measured heat and salt budgets at the same depth (Matzinger et al. 2006b). In Lake Ohrid complete overturn is occurring roughly once every seven years (Hadzisce 1966; Matzinger et al. 2006b). For realistic evaluation of model performance, runs were extended to 2067 to cover several seven year cycles. Regular complete overturn was simulated using seven year intervals of meteorological forcing, each containing three sets of 2001-2003 measurements and one period of 2003/04. Figure 4.7a shows that the model is highly stable over the whole 66-year period, predicting a complete overturn every seven years.

The bio-geochemical model was calibrated to measured balances of P, DO and organic matter, as well as historic data for primary production. For the adjusting of the model an iterative approach was used by calculating and comparing the simulated balances after each simulation run. Adjusted parameters were average settling velocity of dead organic matter, mineralization rate of settling material, the adsorption rate of dissolved phosphorus, as well as phytoplankton and zooplankton growth rates. Most of the adapted parameters had to be changed by less than 50 % from the original version by Omlin et al. (2001), which underlines its applicability for a broad range of lakes (see model equations below). However growth rates of phytoplankton and zooplankton had to be increased by factors 1.7 and 11.6, respectively, probably owing to the oligotrophic nature of Lake Ohrid. Whereas zooplankton growth rates from the literature vary significantly, which makes comparison difficult, the calibrated phytoplankton growth rate of 1.88 d⁻¹ is still well in the expected range (Kalff 2002). The settling velocity $v_{sed,org}$ is the third parameter, which was adapted significantly. It was set to 10 m d⁻¹ by Omlin et al. (2001), a settling velocity which is possible for large particles, e.g., if extensive coagulation with organic and inorganic particles occurs (Kalff 2002). For oligotrophic and clear Lake Ohrid the adaptation to low average $v_{sed,org} \approx 0.46$ m d⁻¹, 2.3 times higher than the settling velocity of living phytoplankton, is reasonable (Kalff 2002).

Table A2.1: Mass balance simulation versus measurements

Parameter	**Unit**	**Simulation**	**Measurement**[1]
TP in water column	mg-P m⁻³	4.56	4.55 ± 0.76
P input	t-P yr⁻¹	47.2	47 ± ?
P outflow	t-P yr⁻¹	1.9	3.0 ± 1.8
Net P-sedimentation	t-P yr⁻¹	46	44 ± 15
Gross sedimentation of organic material	t-TOC yr⁻¹	6300[2]	5600 ± 1000
Primary gross production	g-C m⁻² yr⁻¹	62[3,2]	77[4]

1 Mean and standard deviations of available measurements

2 assuming Redfield stochiometry

3 from direct integration over top 100 m to be comparable with measurement method; simulated, volume weighted production is ~55 g-C m⁻² yr⁻¹

4 measured in 1972 by Ocevski & Allen (1977)

Table A2.1 shows that the model simulates both the P and TOC budget excellently, well within the range of measurements. The modeled results are also in surprisingly good agreement with measured primary productivity, given the complexity of the parameter (Table A2.1). The good reproduction of the mass balances is also reflected in DO concentrations (Figure 4.3d). Finally, phytoplankton and zooplankton biomass are in the right order of magnitude showing reasonable seasonal succession (Figure A2.1). The model fails to reproduce the often high densities of phytoplankton at depths exceeding 100 m (Patceva 2005), although productivity there is comparably small (Ocevski & Allen 1977). From a classical standpoint only few algae would be expected beyond the 1 ‰ compensation depth (Kalff 2002), which is at ~55 m in Lake Ohrid (Matzinger et al. 2006b). Thus the phenomenon of these deep-living, endemic species is not yet understood. As for the k-ε approach, model stability was tested in a simulation covering 66 years.

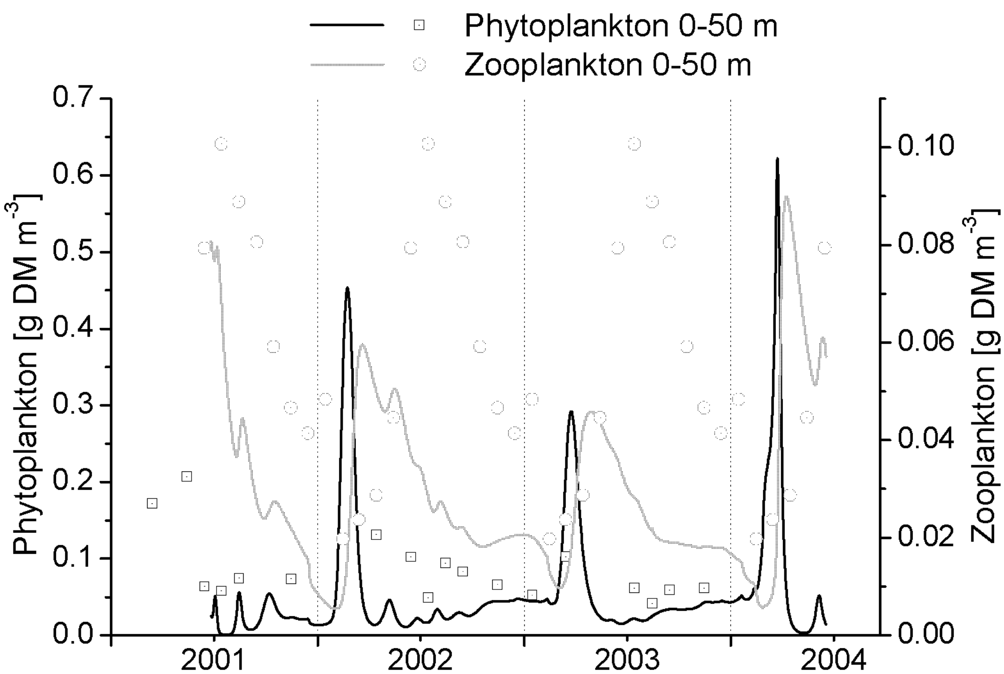

Figure A2.1: Results from bio-geochemical model for plankton: Simulated (lines) versus measured (symbols) data, volume-integrated for different 0-50 m depth section. Measured zooplankton biomass was only available for the year 2000 and is therefore shown in repetition.

A2.3 Model Equations

In the following setup and equations for the bio-geochemical model part are listed. Changes from the model by Omlin et al. (2001) are indicated in *italics*.

Compartments: Water
The original model of Omlin et al. (2001) had two sediment layers which were replaced by an overall sediment balance (process mineral_sed) since no data were collected to calibrate processes in the sediment.

State variables (in alphabetical order):

State variable	Unit	Description
S_HPO4	[g-P m^{-3}]	Dissolved, bio-available phosphorus (mainly ortho-phosphates)
S_O2	[g DO m^{-3}]	Dissolved oxygen (DO)
T	[°C]	Temperature
X	[g-DM m^{-3}]	(Degradable) dead organic material
X_ALG	[g-DM m^{-3}]	Algal biomass
X_P	[g-P m^{-3}]	Phosphorus incorporated in dead organic material
X_PI	[g-P m^{-3}]	Ortho-phosphates adsorbed to dead organic material
X_P_ALG	[g-P m^{-3}]	Phosphorus incorporated in algae
X_ZOO	[g-P m^{-3}]	Zooplankton biomass

Compared to the original model (Omlin et al. 2001) several variables describing the nitrogen cycle have been removed as it does not play an important role in Lake Ohrid. Moreover no difference was made between degradable and inert organic matter, as we use a fixed share of mineralization at the sediment. Finally the algal species "Planktothrix rubescens" was not modelled, as it does not occur in the lake.

Process rates (in order of appearance):

State variable	Unit	Description
growth_ALG	[g-DM m^{-3} d^{-1}]	Growth of algae
resp_ALG	[g-DM m^{-3} d^{-1}]	Respiration of algae
death_ALG	[g-DM m^{-3} d^{-1}]	Death of algae
growth_ZOO	[g-DM m^{-3} d^{-1}]	Growth of zooplankton
resp_ZOO	[g-DM m^{-3} d^{-1}]	Respiration of zooplankton
death_ZOO	[g-DM m^{-3} d^{-1}]	Death of zooplankton
P_uptake	[g-P m^{-3} d^{-1}]	Adsorption of ortho-phosphates to organic particles
mineral_aero	[g-DM m^{-3} d^{-1}]	Aerobic mineralization in water column
mineral_sed	*[g-DM m^{-3} d^{-1}]*	*Aerobic mineralization at sediment surface*
T_relax	[°C d^{-1}]	Heat exchange by surface temperatures as boundary condition
Sol_rad	*[°C d^{-1}]*	*Heat input from solar radiation (added to simulate heatflux due to solar radiation as T_relax is limited to the upper two model gridpoints (5.7 m)*

Processes of nitrification and anoxic mineralization were omitted compared to Omlin et al. (2001). The aerobic mineralization at the sediment has been simplified by the process rate mineral_sed, as no sediment compartments are considered.

Inputs: Rivers (Sateska, Kosleska, Velgoska, Cerava, combined Albanian rivers, temporary inflows), springs (divided among surface and subaquatic springs), diffusive inputs (e.g., P from unconnected households), precipitation
Outputs: Evaporation, outflow
Other fluxes: O_2-gas exchange at surface

The following tables describe each of the processes mentioned above. The top cell indicates the respective process rate. The contribution of a process to the transformation rate of state variables is calculated by multiplying the rate with the corresponding stoichiometric coefficient. Formulas for stoichiometric and process rate variables are indicated in the sub-table “equations”. Finally constant variables are given at the end of each table.

If not indicated differently, numbers and equations are based on Omlin et al. (2001).

Process: Growth of algae

growth_ALG = k_gro_ALG × I_ALG_matrix × monod_HPO4_ALG × X_ALG			
Bio-geochemical conversion processes			
State Variables		**Stocheometric coefficients**	**Description**
X_ALG		1	Algal growth
X_P_ALG		b_P	Algal uptake of orthophosphate (Stochiometry dependent on S_HPO4)
S_O2		1.24	According to the Redfield-ratio
S_HPO4		-b_P	Reduction of orthophosphate in water
Equations			**Description**
k_gro_ALG = k_gro_ALG_20 × exp (β_ALG × (T-20))			Temperature dependence of growth [d^{-1}]
*I_ALG_matrix**			*Effect of light intensity (I) on growth rate: Photosynthetic activity from short-wave radiation was adapted to data from Lake Tahoe (Tilzer & Goldmann 1978) to account for deep-living species of Lake Ohrid [-]*
monod_HPO4_ALG = S_HPO4 / (K_HPO4_ALG + S_HPO4)			Effect of dissolved orthophosphate concentrations on grow rate [-]
I = I_0 × exp (-k_extinct × z)			In-situ light intensity [W m^{-2}]
I_0 = 220 – 125 × cos(t_year / 365.25 × 2 × π) × (1-rs)			*Average global radiation [W m^{-2}] for Lake Ohrid latitude (NOAA NCEP-NCAR CDAS-1 reanalysis project described in Kalnay et al. (1996))*
k_extinct = k_1+k_2 × w_ALG × X_ORG_withoutzoo			Light attenuation coefficent [m^{-1}]
X_ORG_withoutzoo = X_ALG+X			Organic particles without zooplankton [g-DM m^{-3}]
b_P = (b_P_min+b_P_max)/2+(b_P_max-b_P_min)/2 × tanh((S_HPO4-S_HPO4_crit)/DeltaS_HPO4)			Dependence of phosphorus incorporation as a function of phosphate concentration [g-P g-DM^{-1}]
Constant variables	**Unit**	**Value**	**Description**
k_gro_ALG_20	*[d^{-1}]*	*1.88*	*Max. specific growth rate at 20°C; fitted parameter*
β_ALG	[°C^{-1}]	0.046	Temperature dependency coefficient
K_HPO4_ALG	[g-P m^{-3}]	0.0019	Monod coefficient for orthophosphate
rs	*[-]*	*0.2*	*Reflection coefficient of short-wave radiance, as in k-ε model by Goudsmit et al. (2002). rs was not considered separately by Omlin et al. 2001.*
b_P_min b_P_max	[g-P g-DM^{-1}]	0.0014 0.0087	Minimum and maximum phosphorus content of newly produced algae
k_1	*[m^{-1}]*	*0.1*	*Light extinction in the absence of particles, based on PAR measurements (Matzinger et al. 2006b)*
k_2	[m^2 g-WM^{-1}]	0.026	Coefficient for light extinction with particles
w_ALG	[g-WM g-DM^{-1}]	5	factor for converting dry mass to wet mass
S_HPO4_crit	[g-P m^{-3}]	0.004	Orthophosphate concentration at which algal growth switches to reduced P content
DeltaS_HPO4	[g-P m^{-3}]	0.00125	Parameter for switching to production with reduced P content

* Algal growth was fitted to in-situ radiation Rad: *I_ALG_insitu = Rad / (29.34082 + Rad) × 1.89144 × 1 / (1 + 0.00488 × Rad). However the equation had to be evaluated for average daily radiation, which is given in the model. I_ALG_matrix represents average algal reaction for typical day-length and radiation level of each month of the year. Finally I_ALG_matrix was normalized between 0 and 1.*

Process: Respiration of algae

resp_ALG = k_resp_ALG × monod_O2_resp × X_ALG			
Bio-geochemical conversion processes			
State Variables		**Stocheometric coefficients**	**Description**
X_ALG		-1	"Feeding" of algae on their biomass
X_P_ALG		-a_P_ALG	P release to water column
S_O2		*-1.24*	DO consumption; according to Redfield ratio; differs from Omlin et al. (2001) as we do not consider nitrogen cycle.
S_HPO4		a_P_ALG	P release to water column
Equations			**Description**
k_resp_ALG = k_resp_ALG_20 × exp (β_ALG × (T-20))			Temperature dependence of respiration [d^{-1}]
monod_O2_resp = S_O2/(K_O2_resp+S_O2)			Effect of dissolved oxygen [-]
a_P_ALG = X_P_ALG / X_ALG			Average algal P-content [g-P g-DM^{-1}]
Constant variables	**Unit**	**Value**	**Description**
k_resp_ALG_20	[d^{-1}]	0.05	Maximum specific respiration rate at 20 °C
β_ALG	[$°C^{-1}$]	0.046	Temperature dependence coefficient for algae
K_O2_resp	[g DO m^{-3}]	0.5	Half saturation rate

Process: Death of algae

death_ALG = k_death_ALG × X_ALG			
Bio-geochemical conversion processes			
State Variables		**Stocheometric coefficients**	**Description**
X_ALG		-1	Algal death
X_P_ALG		-a_P_ALG	P transfer to dead organic matter
X		1	Increase in dead organic matter
X_P		a_P_ALG	P transfer to dead organic matter
Equations			
k_death_ALG = k_death_ALG_20 × exp (β_ALG × (T-20))			Temperature dependence of death rate [d^{-1}]
a_P_ALG = X_P_ALG / X_ALG			Average algal P-content [g-P g-DM^{-1}]
Constant variables	**Unit**	**Value**	**Description**
k_death_ALG_20	[d^{-1}]	0.03	Specific death rate at 20 °C
β_ALG	[$°C^{-1}$]	0.046	Temp. dependency coefficient

Process: Growth of zooplankton

growth_ZOO = k_gro_ZOO × X_ALG × X_ZOO × LimitAlgP

Bio-geochemical conversion processes

State Variables	Stocheometric coefficients	Description
X_ZOO	1	Zooplankton growth
X_ALG	-1 / Y_ZOO	Zooplankton feeding on Algae
X_P_ALG	-a_P_ALG / Y_ZOO	P transfer from phytoplankton to zooplankton
X	c_e × (1-Y_ZOO) / Y_ZOO	Increase of dead organic material due to fecal pellets
X_P	0	Fecal contains no P
S_O2	*-1.24* × (1-c_e) × (1 - Y_ZOO) / Y_ZOO	Oxidation of food, which is not excreted and not used for zoopl. biomass. Factor according to Redfield ratio; differs from Omlin et al. (2001) as we do not consider nitrogen cycle.
S_HPO4	a_P_ALG / Y_ZOO-a_P_red	P release due to inefficient zooplankton feeding

Equations

k_gro_ZOO = k_gro_ZOO_20 × exp (β_ZOO × (T-20))	Temperature dependence of growth [d^{-1}]
LimitAlgP = min(1,a_P_ALG / a_P_red)	The greater the algal P-content, the greater the growth; zooplankton is according to Redfield ratio [-]
a_P_ALG = X_P_ALG / X_ALG	Average algal P-content [g-P g-DM^{-1}]
Y_ZOO = Y_ZOO_max × min(1, a_P_ALG / a_P_red)	Yield for zooplankton growth; the smaller the algal P-content, the more algae must be eaten for zooplankton growth. [-]

Constant variables	Unit	Value	Description
k_gro_ZOO_20	*[(g-DM m^{-3})$^{-1}$d^{-1}]*	*3.47*	*Max. specific growth rate at 20°C; fitted parameter*
β_ZOO	[°C^{-1}]	0.08	Temp. dependency coefficient
c_e	[-]	0.7	Fraction of food excreted as fecal pellets (not used for zoopl. biomass)
a_P_red	[g-P g-DM^{-1}]	0.0087	Phosphorus content of organic material according to Redfield
Y_ZOO_max	[-]	0.5	Maximum yield for zooplankton growth

Process: Respiration of zooplankton

resp_ZOO = k_resp_ALG × monod_O2_resp × X_ALG

Bio-geochemical conversion processes

State Variables	Stocheometric coefficients	Description
X_ZOO	-1	"Feeding" on own biomass
S_O2	*-1.24*	DO consumption; according to Redfield ratio; differs from Omlin et al. (2001) as we do not consider nitrogen cycle.
S_HPO4	a_P_red	P release to water column

Equations

k_resp_ZOO = k_resp_ZOO_20 × exp (β_ZOO × (T-20))	Temperature dependence of respiration [d^{-1}]
monod_O2_resp = S_O2/(K_O2_resp+S_O2)	Effect of dissolved oxygen on respiration [-]

Constant variables	Unit	Value	Description
k_resp_ZOO_20	[d^{-1}]	0.003	Maximum specific respiration rate
β_ZOO	[°C^{-1}]	0.08	Temperature dependence coefficient for zooplankton
a_P_red	[g-P g-DM^{-1}]	0.0087	Phosphorus content of organic material according to Redfield
K_O2_resp	[g DO m^{-3}]	0.5	Half saturation rate

Process: Death of zooplankton

death_ZOO = k_death_ZOO × X_ZOO			
Bio-geochemical conversion processes			
State Variables	**Stocheometric coefficients**		**Description**
X_ZOO	-1		Zooplankton death
X	1		Increase in dead organic matter
X_P	a_P_red		P transfer to dead organic matter
Equations			
k_death_ZOO = k_death_ZOO_20 × exp (β_ZOO × (T-20))			Temperature dependence of death [d^{-1}]
Constant variables	**Unit**	**Value**	**Description**
k_death_ZOO_20	[d^{-1}]	0.029	Specific death rate at 20 degrees
β_ZOO	[°C^{-1}]	0.08	Temperature dependency coefficient
a_P_red	[g-P g-DM^{-1}]	0.0087	Phosphorus content of organic material according to Redfield

Process: Adsorption of orthophosphates to organic particles

P_uptake = abs(AreaGradient/Area) × k_upt × (a_P_max-a_PI) × monod_O2_ads × S_HPO4 × X			
Bio-geochemical conversion processes			
State Variables	**Stocheometric coefficients**		**Description**
X_PI	1		Adsorption of P to organic matter
S_HPO4	-1		Reduction in phosphates
Equations			
a_PI = X_PI / X			Mass of adsorbed phosphate per mass of X [g-P g-DM^{-1}]
monod_O2_ads = S_O2/(K_O2_ads+S_O2)			Effect of dissolved oxygen [-]
Constant variables	**Unit**	**Value**	**Description**
abs(AreaGradient/Area)	[m^{-1}]		Sediment area per lake volume
k_upt	[m^4 g-P^{-1} d^{-1}]	1200	Phosphate uptake rate constant
a_P_max	*[g-P g-DM^{-1}]*	*0.0052*	*Maximum mass fraction of phosphate adsorbed to org. matter; fitted parameter*
K_O2_ads	[g DO m^{-3}]	0.5	Half saturation rate

Process: Aerobic mineralisation of organic material in open water

mineral_aero = k_miner_aero × monod_O2_aero × X			
Bio-geochemical conversion processes			
State Variables	**Stocheometric coefficients**		**Description**
X	-1		Mineralization of organic matter
X_P	-a_P		Release to water column of incorporated P
X_PI	-a_PI		Release to water column of adsorbed P
S_O2	*-1.24*		Consumption of DO; according to Redfield ratio; differs from Omlin et al. (2001) as we do not consider nitrogen cycle.
S_HPO4	a_P+a_PI		P release to water column
Equations			
k_miner_aero = k_miner_aero_20 × exp (β_BAC × (T-20))			Temperature dependence of bacterial activity [d^{-1}]
monod_O2_aero = S_O2 / (K_O2_aero + S_O2)			Effect of dissolved oxygen [-]
a_P = X_P / X			Average P-content in organic matter [g-P $g\text{-}DM^{-1}$]
a_PI = X_PI / X			Average P adsorbed to organic matter [g-P $g\text{-}DM^{-1}$]
Constant variables	**Unit**	**Value**	**Description**
k_miner_aero_20	*[d^{-1}]*	*0.008*	*Aerobic specific mineralization rate at 20 °C in open water; fitted parameter*
β_BAC	*[$°C^{-1}$]*	*0.09*	*Temperature dependence coefficient for bacteria; fitted parameter*
K_O2_aero	[g DO m^{-3}]	0.2	Half saturation rate

Process: Aerobic mineralisation of organic material at sediment surface

mineral_sed *= X × v_sed_ORG × abs(AreaGradient/Area)**			
Bio-geochemical conversion processes			
State Variables	***Stocheometric coefficients***		***Description***
S_O2	*-1.24 × a_miner_sed × monod_O2_aero*		*Consumption of DO during mineralization*
S_HPO4	*(a_P+a_PI) × a_miner_sed*		*Production of bioavailable P during mineralization*
Equations			
monod_O2_aero = S_O2 / (K_O2_aero + S_O2)			*Effect of dissolved oxygen [-]*
a_P = X_P / X			*Average P-content in organic matter [g-P $g\text{-}DM^{-1}$]*
a_PI = X_PI / X			*Average HPO4 adsorbed to organic matter [g-P $g\text{-}DM^{-1}$]*
Constant variables	***Unit***	***Value***	***Description***
abs(AreaGradient /Area)	*[m^{-1}]*		*Sediment area per lake volume*
v_sed_org	*[m d^{-1}]*	*0.46*	*Settling velocity of organic material; fitted parameter*
K_O2_aero	*[g DO m^{-3}]*	*0.2*	*Half saturation rate (from Omlin et al. 2001)*
a_miner_sed	*[-]*	*0.42*	*Fraction of sedimented org. material, which is mineralised at sediment surface (1-a_miner_sed, enters sediment permanently), based on measured P budget*

** the removal of particulate matter (X, X_P and X_PI) by sedimentation is incorporated in the standard Aquasim lake compartment and thus not included under conversion processes (see sedimentation process below).*

Process: Heat exchange by surface temperatures as boundary condition

T_relax = if z<z_epi_meta_min then k_relax × (T_meas-T) else 0 endif			
Bio-geochemical conversion processes			
State Variables	**Stocheometric coefficients**		**Description**
T		1	Adaptation of T to measured surface temperature to mimic heat budget (except short-wave solar radiation)
Constant variables	**Unit**	**Value**	**Description**
z_epi_meta_min	*[m]*	*5.7*	*Depth adapted to surface measurements (chosen to include two topmost model gridpoints)*
k_relax	$[d^{-1}]$	20	Rate of T-adaptation
T_meas	*[°C]*		*Measured or extrapolated surface temperatures (as used in k- simulations)*

Heat input from solar radiation

Sol_rad *= if z>z_epi_meta_min then I × k_extinct / cp / 1000 × 86400 else 0 endif*			
Bio-geochemical conversion processes			
State Variables	***Stocheometric coefficients***		***Description***
T		*1*	*T increase as a result of absorbed short-wave solar radiation*
Equations			
k_extinct, I			*see definition under "Growth of algae"*
Constant variables	**Unit**	**Value**	**Description**
z_epi_meta_min	*[m]*	*5.7*	*Depth adapted to surface measurements (chosen to include two topmost gridpoints)*
cp	*$[J\ kg^{-1}\ K^{-1}]$*	*4180*	*Heat capacity of water*

Sedimentation Process*

Constant variables	**Unit**	**Value**	**Description**
v_sed_zoo	$[m\ d^{-1}]$	0	Zooplankton does not sediment
v_sed_alg	$[m\ d^{-1}]$	0.2	Settling velocity of algae X_ALG
v_sed_org	*$[m\ d^{-1}]$*	*0.46*	*Settling velocity of organic material; fitted parameter*

* the incorporated sedimentation process effects the settling of particulate variables, as well as their removal, when reaching the sediment surface.

Vertical mixing

monthly Kz-profiles from k-ε model (Goudsmit et al. 2002).

Chapter 5

Capacity Building

5.1 Settings

After independence of the Republic of Macedonia from Yugoslavia in 1991 Lake Ohrid became a focus of international developing agencies, being one of the few ancient, long-lived lakes in the world and the only one in Europe. The main concern was lake eutrophication, fostered by reports on severely polluted inflows by the Hydrobiological Institute in Ohrid (HBI). In 1995 a feasibility study was commissioned by the World Bank through the Global Environment Facility (GEF) to identify necessary efforts to protect Lake Ohrid (Ernst Basler and Partners 1995). In terms of infrastructural assistance, extension of the sewerage system, as well as technical support of a bi-lateral scientific monitoring program were suggested in the study. In the following a large-scale initiative, the GEF Lake Ohrid Conservation Program (LOCP), was set up headed by the World Bank and supported by the German KfW Developing Bank and the Swiss State Secretariat for Economic Affairs (seco).

A significant share of the seco investments within LOCP was allocated for the establishment of a scientific lake monitoring program, carried out by HBI, in order to identify threats to the lake ecosystem and to evaluate existing and potential future protection measures. In the course of the project technical training alone has proved inadequate for providing monitoring results using the modernized analytical infrastructure. As a result seco supported this PhD project to increase the sustainability of preceding seco-investments at HBI. The main aims of the three year Eawag-seco project were:

- scientific analysis of water quality threats,
- regular reporting on institutional background,
- transitional support in the use of the new analytical infrastructure,
- establishing conditions for (independent) future monitoring of Lake Ohrid.

5.2 Project Setup

The core element of the project was the scientific research performed within this PhD thesis. The findings should extend the knowledge on Lake Ohrid's nutrient cycle and be used for the proposition and evaluation of future and already planned environmental protection measures. The outcome of the scientific work was very much depending on regular local sampling and sample analysis. Thus a measurement campaign was to be planned commonly between Eawag and HBI.

A particular discrepancy within the LOCP had lain in the use of the Swiss-funded new sampling and analytical equipment. This discrepancy had its reason mainly in the lack of technical experience to handle the new equipment, but also of know-how to evaluate and interpret the achieved data. During the measurement campaign the sampling equipment was to be used jointly with HBI technicians and scientists so both partners would gain the necessary experience together. In terms of analytical measurements this direct collaboration was intended to cover the use and quality control of nutrient analyzers. Except for some analyses, which required specific equipment, all measurements were to be performed at HBI. The instruction of new equipment was planned to be further supported by visits of Eawag specialists in Ohrid.

The interpretation of data was to be supported by the close cooperation during my visits, in joint scientific publications and more generally by contact building, making available recent scientific literature and helping with English language.

In terms of laboratory work, the PhD monitoring was limited mainly to nutrient analysis. In order to trigger know-how transfer for further instruments a general scientific and student exchange was planned. Within this project part, HBI scientists would be given the opportunity to stay for an internship at Eawag and get an in-depth knowledge of analytical techniques and data analysis.

For clarity, the different project parts were organized in four modules. *Module 1* – the *Scientific Program* – contained the main scientific tasks linked to my PhD thesis, including three annual stays in Ohrid. Modules 2 to 4 were to further strengthen the collaboration between the two institutes and to support the outcome of the project. *Module 2 – Student Exchange* – included two highly motivated students. They were intended to work at Eawag on their own project or on an Eawag project for approximately four months. During their stays, they could gain experience on equipment, which exists at HBI as well. For direct scientific exchange, *Module 3 – Visits of Eawag Scientists at HBI* – and in the opposite direction *Module 4 – Visits of HBI Scientists at Eawag* – were planned.

It was my main task to link the four proposed project modules. With regular reports, administrational problems within Macedonia were to be reported and made transparent to *seco* and the Swiss Cooperation Office in Skopje. Moreover, I should help establishing contacts between Eawag and HBI scientists. In the set-up of the project, a particular emphasis was laid on triggering the interests of the main implementing institutions. Whereas I was depending on the amount and quality of collected data for my PhD, HBI had a strong interest in international exchange and to proof their ability to perform lake monitoring tasks within a scientific scheme. The bilateral framework, with two scientific partner institutions, was also intended to keep the administrative effort on a low level.

For the evaluation of the project achievements a detailed set of success indicators were defined between Eawag and seco at the beginning of the project and following each annual report.

5.3 Results

5.3.1 Scientific Monitoring

The measuring program for the project was established in several steps. During my first two stays in Ohrid, in spring and summer 2002, joint sampling tours were organized. The focus of the two stays lay in the adaptation of sampling techniques, testing of analytical equipment, quality control and the definition of suitable sampling sites. Based on these experiences, sampling regularity, parameters and sites were defined in a memorandum between HBI and Eawag. While the measuring campaign was followed within the agreed-on framework, details were adapted, depending on latest findings. Since the beginning of 2003 the sampling program was extended at the lake sites by HBI, collecting samples for chemical, microbiological and biological parameters with the new sampling devices. After the conclusion of the memorandum-based PhD monitoring campaign, the sampling of Lake Ohrid was continued independently. The collaboration with HBI within the measuring campaign was clearly above expectations, both in terms of reliability and scientific quality. Thanks to the motivation of HBI technicians and scientists the program could be adapted flexibly on many instances (e.g. weekly samples from Sateska River, sub-lacustrine spring search on Lake Ohrid, etc.). The results of the monitoring and successive data analysis formed the prerequisite of this PhD thesis.

However monitoring became an unexpected challenge. From 2003-2004 national financial support for HBI had been cut dramatically, as a result of the economic crisis in Macedonia. There was basically no funding for running costs, such as chemicals, petrol, heating oil or phone bills. Thanks to the flexibility of everybody involved this gap could be partly closed by CHF 15,000.-, allocated for consumables within the seco-Eawag project. As a result monitoring activities for my PhD but also for all the national Macedonian projects could be maintained. While the situation has improved since the incident underlined the importance of alternative financial sources for Macedonian institutions if they want to be able to plan ahead.

In short layman's reports I tried to summarize first scientific results in a generally understandable form during the project for distribution among all stakeholders known to me. In addition special contributions to current activities were made: e.g., commenting on the planned building of a treatment plant in Albania, participation at a donor's conference organized by seco, one detailed reporting at the Ministry of Environment and Physical Planning in Skopje. While these contributions were an honest try to transfer scientific knowledge to the applied side, my activities lacked the necessary persistence to have a great effect. The reason is simply that I was always working in parallel on several tasks and I was simply unable to follow everything with the same dynamic. Moreover responses to my efforts were mostly un-enthusiastic and it sometimes seemed to me that there is more energy in the depths of Macedonian administration to make each others life difficult than for actual activities. What was clearly neglected during the project was the Albanian lake side. Although I performed limited samplings in Albania only few, irregular contacts were held with Albanian scientists.

5.3.2 Use of Sampling and Analytical Equipment

In the course of the PhD monitoring campaign sampling and analytical methods were shared and developed together with HBI technicians, assisted also by Module 3. Today the nutrient analysis, as well as the lake sampling equipment is operated professionally by HBI staff. Moreover inter-laboratory quality control was organized between HBI and Eawag for all parameters used in my PhD.

In addition to the regular cooperation in the field, two student internships have been organized in order to allow the involved students the apt handling of new technical equipment at HBI. A first internship in 2003 was performed by Suzana Patceva in the field of phytoplankton and concentrated on the comparison of chlorophyll measurements and the use of flow cytometer for cell counting (Patceva 2003; Patceva 2005). During the internship she was supervised by Renata Behra, head of the "Environmental Toxicology of Algal Communities and Populations" group at Eawag. The second student exchange in 2004 was planned to be in the field of analytical chemistry, involving the new GC and GC-MS equipment at HBI. In accordance with seco, E. Veljanoska-Sarafiloska from HBI could work for four months at the University of Novi Sad (SCG), where she was supervised by Ivana Ivancev – Tumbas, professor at the department of chemistry (Veljanoska-Sarafiloska 2004).

Since the internship, there has been a regular exchange between Novi Sad and HBI. In the future the student will write her PhD on micro-pollutants in the three natural Macedonian Lakes, Ohrid, Prespa and Dojran. Particularly in the last two, current use and accumulation of pesticides could be a potential problem. For both internships, it can be said, that the techniques were learned and can be applied independently.

The learning of techniques was possible thanks to the students' initiative and their integration in the respective teams. However, one major question will be whether the involved students will be able to continue this work on the long-run. There are two main problems: (a) the new analytical equipment has very high running costs, which cannot be paid at the moment through national projects, which have allocated typically around € 1000.- per year. (b) There is no real company support for most instruments in Macedonia. However the availability of spare parts and problem diagnosis are crucial for the work with such instruments. Some problems have already materialized during this project. Whereas the use of the HPLC at HBI to measure chlorophyll-a seemed like a sensible idea, being the standard method at Eawag, it turned out that standards are extremely costly and basically unaffordable for HBI. For the measurement with GC-MS large stocks of spare parts and consumables had been delivered through the LOCP. However some minor parts were forgotten. The necessary € 600.- could be paid through this project, however one reagence was lacking again in 2005. At the moment it seems that the new equipment can only be run in the future with funding and technical support through international projects.

5.3.3 International Exchange

Although scientific exchange was not among the main aims of the original scheme, it was realized during the project that this point had not been given enough emphasis. Cooperation, i.e. the link with international scientific community was found crucial to achieve up-to-date scientific and applied results. Moreover running of high-tech scientific infrastructure is only possible with international financial support at the moment. Thus we tried to adapt the project in the last project year, by strengthening this point, as far as possible.

In addition to the two foreseen internships a third internship was organized for Dusica Ilic-Boeva within the Eawag department "Fish Ecology and Evolution", supervised by Ole Seehausen, professor at the University of Berne (Ilic-Boeva 2004). Rather than aiming at a mere learning of technical skills its objective was the fostering of a long-term exchange with Eawag.

In general the internships seem to have triggered the motivation of the involved young scientists. They all stayed in contact and plan to publish their work internationally. Given the fact that those people are involved in recent projects and will be part of the future generation at HBI the money spent on the three internships was certainly very well invested.

The idea of the visits of HBI scientists at Eawag was mainly the support and acceptance of the education received by the students within their department and generally at HBI. As a result visits were organized for the involved supervisors– Vasa Mitic, Momcula Jordanoski and Zoran Spirkovski – at Eawag or in Novi Sad. These visits have proofed to be very effective, both regarding the support of the new techniques, as well as intensive contact building. For the third internship the visit of the supervisor Z. Spirkovski was joined up with a visit of the director of HBI, Goce Kostoski. In a tense program, a particular focus was given to future cooperation between Eawag and HBI. In several meetings in November 2004, the submission of a joint project proposal for May 2005, the continuation of the collaboration with Eawag in the field of fish and potentially a future collaboration in the field of zooplankton were agreed on.

In addition to exchange generated directly within the project it was possible to help establish two promising international contacts in the fields of sedimentology (headed by the University of Leipzig) and freshwater molluscs (University of Giessen).

5.3.4 Outcome

The results of the three-year project above were judged very positive, both regarding general performance and future outlook. However the actual effect of a project can only be verified in the future. As the Eawag-seco project finished by the end of 2004, nearly one year ago, it is possible to get a first glimpse of realistic project performance.

As planned a project proposal was submitted in May 2005 to SCOPES, a joint program of the Swiss National Science Foundation and the Swiss Agency for Development and Cooperation to support scientific collaboration with institutions from South-Eastern Europe. In September 2005 the proposed joint project on the subaquatic springs of Lake Ohrid was granted, ensuring continuity of the Eawag-HBI cooperation for the next three years. Manuel Kunz, student of Environmental Sciences at ETH in Zürich, has already started his diploma thesis on Lake Ohrid as a project start-up. Apart from scientific cooperation the SCOPES project structure allows HBI to cover running costs, which are particularly high for some of the "new" equipment and have partly blocked their regular use in the past year, despite newly acquired skills. Finally a new Masters position in analytical chemistry (taking three to four years in Macedonia) can be established through the SCOPES program.

Dusica Ilic-Boeva has continued her work on genetic analysis of fish species in 2005 by two stays at Eawag. In addition a large project for mollusc investigations has been started from the Justus Liebig University Giessen (GER), headed by Tom Wilke. HBI has profited greatly from the collaboration receiving an equipped lab for DNA extraction. Moreover Saso Trajanoski, scientist at HBI, will probably be able to do a sabbatical in

Giessen. Finally a group of sedimentologists headed by Bernd Wagner from the University Leipzig (GER) have extended their activities to a full project, which includes a practical training with German, Macedonian and Albanian students.

Seeing the great increase in international, scientific links of HBI it has to be mentioned that the main drivers are a few people who are supported by several motivated young scientists and technicians. During the changes of the past years the difference between these initiative people and some who are almost excluded from international cooperation – because of a lack of interest or simple language problems – has increased.

5.4 Conclusions

The dynamics and motivation at HBI of the past years have a great potential to improve monitoring of Lake Ohrid continuously into the future. This PhD project merely supported the process, and other factors, such as the great, personal initiative of several HBI technicians and scientists were the key for any changes to occur. Based on the experience within this PhD project I have tried to outline the main lessons learnt. Moreover the sustainability of the project success is critically discussed. In the last section I try to evaluate my personal experiences with the combination of a PhD thesis with a small project for capacity building.

5.4.1 Lessons Learned

The whole project based strongly on a partnership between two institutions with scientists on either side. The similar background – in this case scientific – of both project partners has certainly helped collaboration, as there are joint interests, which make understanding across language barriers easy. A very successful way of knowledge "transfer" has proved to be simply joint work. Thanks to my own inexperience we maximized our learning through a trial-and-error approach, both on the field and in the lab. The work towards a clear scientific goal was also emphasized in the organized internships, where the students were integrated in a group working within a real project. Such learning-on-the-job seems to be much more successful regarding their long-term effect than expert trainings supplied by manufacturers of scientific equipment. Visits of experts can then be effective, when the instrument has already been applied on realistic samples for several months and lists of questions and problems were made.

Unfortunately an originally planned, parallel PhD thesis at HBI could not take place. As a result joint data analysis and publication of results was neglected to a certain extent. For similar future projects it is thus suggested to put more emphasis on the direct involvement of at least one person from both partners.

» *Given the above experience it can be concluded that technical know-how transfer, particularly when dealing with high-tech equipment, is most effective if training-on-the-job is possible to gain experience in a realistic setting. Short expert training sessions can support this learning process; however they are no alternative if sustainable, long-term operation of the instruments is aimed at.*

In the context of application of high-tech equipment and measurement output, large-scale structural assistance, such as the LOCP, seems sub-optimal compared to our small-scale scientific project.

(1) A scientific project setup is similar to new future tasks of a scientific institution. Thus experience gained from such a project automatically improves the institution's capacity for scientific collaboration. Moreover, required project output are scientific publications, which make sure that results are publicly available and at the same time are a reference for future grants.

(2) As a lot of money is involved financial channels get more complex. On the one hand this leads to a "leaking" of funds, on the other hand it may get ridiculously difficult to buy some inexpensive but badly needed consumables. In our small-scale project an account was installed in Ohrid, with authority for both partners, containing never more than € 1500.-. The account proved to be a great tool to allow HBI flexible repair works and purchase of spare chemicals and thus prevented unnecessary project delay.

(3) It goes without saying that the financing above is not sped up if the involved institutions do not get on well with each other. Ironically the financial channels also seem to be one reason for such animosities. Indeed large-scale financial means can create or escalate conflicts by one-sided distribution or high salaries of local project staff. In the case of Lake Ohrid, the World Bank decided to start a follow-up monitoring project, changing the lead from HBI to the Faculty of Science at Skopje University. Whereas this set-up may be reasonable from an organizational point of view, it led to a one-year quarrel, in which the faculty tried to take over the overall monitoring to secure as much money as possible for their institution. Because of the problems at the establishment of an acceptable proposal the World Bank finally decided that they will not invest in a follow-up monitoring under these conditions, leaving four scientific Macedonian institutions to their animosities. Although I was not there at the time it seems that a similar process accompanied the establishment of LOCP.

» *When dealing with collaboration issues a small-scale project approach guarantees a more efficient use of funding and has a smaller risk of disrupting the institutional network.*

Although most instruments can be handled well thanks to the student exchange, the experiences (chapter 5.3.2) indicate that the step to their regular application is an additional challenge. Particularly on the long run regular use is only possible with (international) funding. If repair works are necessary measurements can be delayed by several months, because instruments need to be shipped abroad, even for problem diagnosis.

In the case of HBI it took several years before instruments were taken into operation. One reason is certainly the lack in suitable training, as outlined above. A second problem was that several instruments were delivered without a clear scientific question linked to them.

My experience within this project partly questions whether transfer of high-tech technology to countries in political transition is a sensible option. I think that investments in HBI were well justified in order to install a monitoring station for probably the most special lake of Europe. Moreover the regular use of the boat and the scientific sampling equipment underlines the capacity of HBI. However new high-tech measurement devices should have been delivered in a step-by-step approach, where the donation of a new instrument is depending on (i) a clear scientific question linked to it and (ii) efficient and correct operation of the last donated instrument.

» *Donations of high-tech instruments must be well planned and performed in a step-by-step approach, depending on measurement performance and scientific output.*

Swiss (cantonal) monitoring programs are then successful if they are based on a clear question. The monitoring is then planned and adapted to answer this question. The mere collection of data without clear background can lead to measurement errors, which go often undetected for years or even for decades. Moreover there is no clear output to the political side. The same is suggested for future monitoring of Lake Ohrid. It is the task of the respective political unit to put up some clear questions in collaboration with scientists. It is further suggested that the scientific reporting is reviewed by international experts, similar to a peer-reviewed publication.

» *Any scientific monitoring project must be goal- and problem-oriented if it should provide information of scientific and political significance.*

5.4.2 Institutional Settings

International collaboration of HBI has boosted over the past year. The future looks also promising as HBI has developed – in the international context – from a monitoring station to a full partner of several projects. In terms of its institutional situation it is also important to establish good links to other scientific institutions and administrative units on a national level.

As has been indicated above the inter-Macedonian relations are certainly not ideal. As far as I know they have not improved through this project. It may even be that because of this project HBI was not forced to cooperate more on a national level, as some limited financing was available. However this is unlikely as the cutting of running costs was a national phenomenon and even good political relations would not have prevented temporal hardship.

A particular critique of local administration and the World Bank was the quality and regularity of reporting regarding the state of Lake Ohrid. Indeed reports were not issued every year and some of the contributions were rather descriptive with limited scientifically-based interpretations. However the lack in reporting must also be seen in the context of scientific culture in Macedonia. I myself have experienced the phenomenon of data-hiding from the side of the Hydrometeorological Institute – where it took me five months to get some averaged river runoff measurements and an intervention of the Swiss Cooperation Office was necessary to get some wind data – or from the World Bank Project Implementation Unit – which simply refused to give me statistical data they had assembled. In a similar way I can imagine that it is close to impossible for an external organization to receive any data from HBI.

An important question is now whether the international collaboration of HBI is sustainable on the long run if the institute is not well integrated nationally. I am convinced that national integration is important. However at the moment international projects can be vital to survive political transition and catch up with up-to-date science. Publication of results will definitely be improved through such projects. Moreover perception is changing as Macedonia has changed the accreditation system for scientists, which is now based on impact factor of their publications. In my view a more open publication policy will also lead to an improvement of national collaboration.

On the long run a vision needs to be defined and followed by a scientific institution as the HBI. In it the recognition is important that science is not valuable for its own sake but has the obligation to contribute to political and social discussion. A first step has been made at HBI by writing a vision paper, assisted by Jürg Blösch, Alfred Wüest, Mary Watzin, Marc Morell and myself after the seco Lake Ohrid conference. A first draft has been adapted in the following year and attached to the SCOPES proposal, giving reference of a clear planning at HBI. Ideally the vision will take increasingly clearer form and assist the HBI to find its future position within Macedonia and in an international context.

5.4.3 Combination of PhD Thesis with Cooperation Project – A Good Idea?

Looking back one central difficulty was working on two fields with very different indicators for success. In contrast to projects funded by the Swiss National Science Foundation, reports must have a totally different emphasis than the scientific tasks. To follow two different main strains of project aims was certainly a challenge. It was partly responsible for my late start of the publication process. Moreover, as shown in 5.3.1 it was not always possible to respond to all the project demands.

Still, both the seco-Eawag project, as well as the PhD thesis have turned out successful. The close collaboration with HBI has given me the chance to work on a project abroad and gain rich cultural experience. Moreover I could basically organize my own project, from the proposal via the financing to the actual management. The applied question, the collaboration in Macedonia and the regular exchange with seco representatives have always reminded me that there are different ways to see and emphasize things. The multitasking has also prevented me from the classical "mid-thesis-depression", as one project side has always moved forward. Finally I think that the additional experience will support me in my future work, in- or outside of the scientific world.

Summarizing I find that the combination of a PhD with the responsibility for the organization of a small project in an international setting does make a lot of sense. On the one hand the PhD setting allows a very close collaboration with the partner institution. On the other hand the PhD student gains additional experience and improves her/his job qualifications. Thus in projects, where complicated infrastructure and a scientifically interesting question are involved, similar projects are suggested to precede, accompany or follow actual investments.

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Од самиот почеток Гоце ми даваше подршка за работата и организацијата на моите, понекогаш чудни идеи.

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