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Primary Research Paper

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.

Introduction

Lake Ohrid and Lake Prespa form a very unusual lake system. Situated in south-eastern Europe between Albania, Macedonia and Greece (Fig. 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 m higher than Lake Ohrid (Table 1), separated by the Mali i Thate/Galicica mountain range, which consists of karst rock structures (Eftimi et al., 2001). Stable isotope 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 (Fig. 1) in 1935, dedicated to the study of Lake Ohrid and

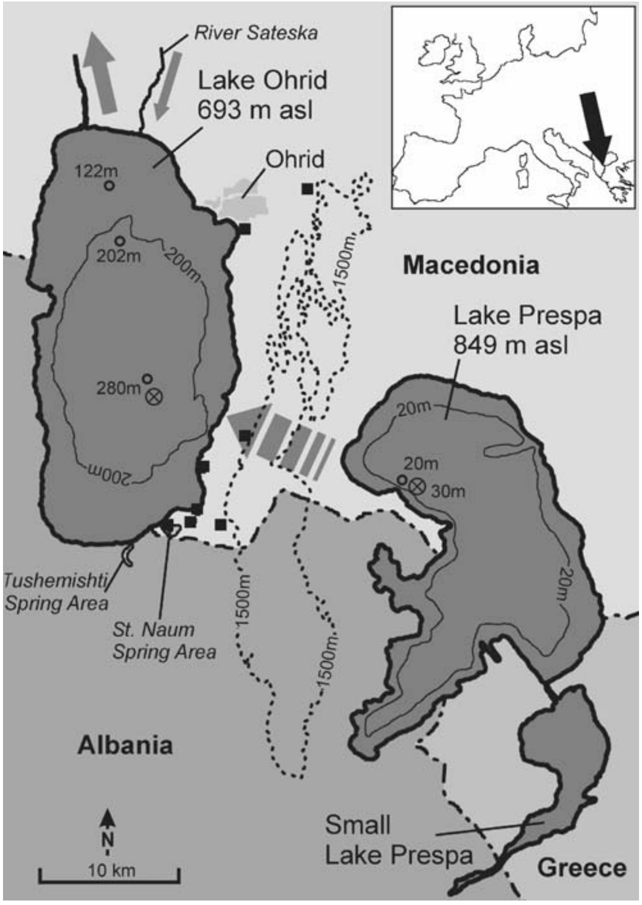

Figure 1. Overview of study area indicating the different sampling sites. The grey arrows indicate flow direction (the broken arrow signifies underground flow). Crossed circles indicate main lake sampling sites, empty circles are sediment core locations and filled squares are sampled karst springs. The inset map shows the location of the study area within Europe.

Table 1. Characteristics of Lake Prespa and Lake Ohrid

Property	Unit	Lake Prespa	Lake Ohrid
Altitude	m asl	849 (854)[a]	693
Catchment area	km^2	1300[b]	2610[c]
Lake surface area	km^2	254 (282)[a]	358
Maximal depth	m	~48 (54)[a]	288
Mean depth	m	14 (19)[a]	155
Volume	km^3	3.6 (4.8)[a]	55.4
Hydraulic residence time	yr	~11 (17)[a]	~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[c]
Number of tourists per year		<1000?	~50,000

[a]Value in parentheses: in the 1980s before recent water level decline of Lake Prespa.
[b]Including Small Lake Prespa and its catchment.
[c]Including Lake Prespa and its catchment.

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 & 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 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.

Although similar in surface area, Lake Prespa is much shallower than Lake Ohrid (Table 1). Archaeological findings indicate that its water level was subjected 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 $\tau \sim 70$ year (Table 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.

Study sites

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 (254–285 km^2) and its catchment (822–1775 km^2). In the following, we use the values from a recent compilation by Anovski (2001) given in Table 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 (Fig. 2). A topographic map from the 1940s (provided by the Hydrobiological Institute Ohrid (unpublished data, ca. 1940)) indicates that the present water level is ~5–6 m lower than in the 1940s. Significant lake level decreases have also been observed since the late 1980s (Fig. 3). To the south, Lake Prespa is connected to Small Lake Prespa (Fig. 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

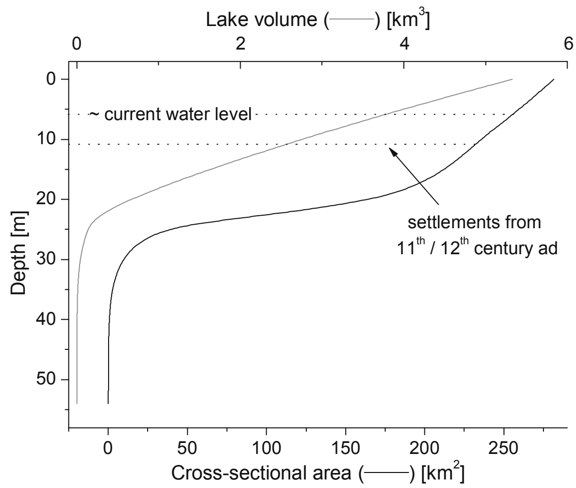

Figure 2. Hypsograph of Lake Prespa: cross sectional area and enclosed volume as a function of depth relative to the "historic" water level (~6 m above current level). Data are from a map from the 1940s (Hydrobiological Institute, unpublished data). The horizontal lines indicate known water levels of Lake Prespa.

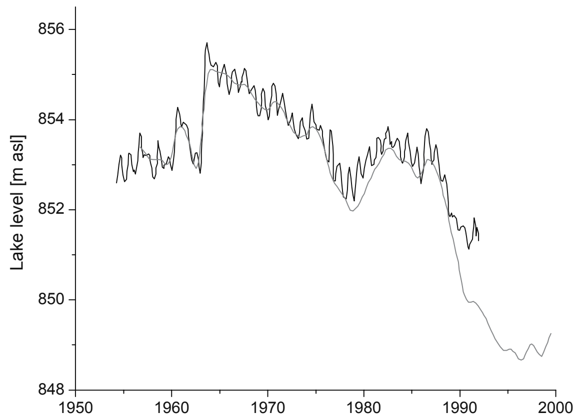

Figure 3. Development of Lake Prespa water level over the past five decades. The fine line shows seasonal fluctuations according to Greek measurements presented in Hollis and Stevenson (1997). The bold, grey line represents annual means from Albanian measurements, adapted from Anovski et al. (2001).

destratification of the entire water column from September to April/May and consequently all dissolved substances are homogenized annually.

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 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 (Fig. 1). In contrast to Lake Prespa, Lake Ohrid is an oligomictic lake with complete mixing occurring roughly once per decade (Hadzisce, 1966).

Karst springs

Along the western side of the Galicica/Mali i Thate Mountain Range (Fig. 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 (Fig. 1).

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 gives an overview of all samples and analytical parameters within our program.

Table 2. Overview of sampling and measurement program

Site	Sampling period	Number of samples	Parameters
Phosphorus load from Lake Prespa[a]			
Lake Prespa (0, 5, 10, 15, 20, 25, 30 m)	12/2002 9/2003	44	SRP, TP, DO
		36	NO_2^-, NH_4^+, TN
Lake Prespa sediment core	5/2002	43	TC, TIC, TN, Water content
		15	TP
		18	^{137}Cs, ^{210}Pb
Transfer to Lake Ohrid[a]			
Lake Prespa (0, 5, 15 m)	4/2001 9/2003 (3 different dates for isotopes)	3	$\delta^{18}O$, δD
		21	Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-}
		32	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	$\delta^{18}O$, δD
		11	Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-}
		10	TP
		12	SRP
Other springs	4/2001 9/2003	11	$\delta^{18}O$, δD
		10	Na^+, K^+, Ca^{2+}, Mg^{2+}, Cl^-, SO_4^{2-}
		10	TP
		21	SRP
Rain samples	5/2001 10/2002	8	$\delta^{18}O$, δD
Effect on Lake Ohrid[a]			
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 and 4/2003	76	TP, Water content
		13	^{137}Cs, ^{210}Pb

[a]Titles in italics correspond with the subtitles in ‘Materials and methods’ and ‘Results and discussion’ sections.

Lake Prespa underground outflow

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 (Fig. 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 $Q_{\mathrm{out,add}}$, 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

$$Q_{\mathrm{out,add}} = Q_{\mathrm{out,undgr}}(z=0) - Q_{\mathrm{out,undgr}}(z=h) + \underbrace{E_{\mathrm{net}} \cdot (A(z=0) - A(z=h))}_{i} \tag{1}$$

where $Q_{\mathrm{out,add}}$ ($\mathrm{m^3\ yr^{1}}$) is the additional water consumption, which leads to a change in the water balance, $Q_{\mathrm{out,undgr}}$ ($\mathrm{m^3\ yr^{1}}$) is the underground outflow for water level z, A ($\mathrm{m^2}$) is the lake surface area, and E_{net} ($\mathrm{m\ yr^{1}}$) is the net evaporation from the lake surface (= evaporation − precipitation = 0.37 $\mathrm{m\ yr^{1}}$). 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 (1) can be expressed for an arbitrary water level $z = h$ as

$$Q_{out,undgr}(z=h) = Q_{out,undgr}(z=0) \times \left(\underbrace{\frac{A(z=h)}{A(z=0)}}_{ii} \cdot \underbrace{\sqrt{\frac{V(z=h)\cdot A(z=0)}{V(z=0)\cdot A(z=h)}}}_{iii} \right) \quad (2)$$

where V (m^3) is the lake volume. If we combine Equations (1) and (2), three stabilizing processes are considered that are effective during the water level decline [letters correspond with the terms in Equations (1) and (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 (1) and (2) we find:

$$\begin{aligned} Q_{out,tot}(z=h) &= Q_{out,add} + Q_{out,undgr}(z=h) \\ &= Q_{out,undgr}(z=0) + E_{net} \\ &\quad \times (A(z=0) - A(z=h)) \quad (3) \end{aligned}$$

According to Equation (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

$$\tau(z=h) = \frac{V(z=h)}{Q_{out,tot}(z=h)} \quad (4)$$

will decrease with declining water level.

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; Fig. 1) water samples were collected ~bimonthly at 5 m depth intervals using a Niskin bottle (Table 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).

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

$$\langle X\rangle_h = \int_{z=z_{max}}^{h} X(z)A(z)dz \ \{\int_{z=z_{max}}^{h} A(z)dz\}^{\ 1} \quad (5)$$

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.

Sediment sampling

One sediment core was retrieved from a depth of 20 m in Lake Prespa, close to the water sampling site (Fig. 1), using a gravity corer (Kelts et al., 1986) and subsequently sectioned to 1–2 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 h 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 3 M 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 γ-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:

$$S_M = (1 - \mathrm{POR}) \cdot \mathrm{SR} \cdot \rho_{\mathrm{sed}} \tag{6}$$

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 $\rho_{\mathrm{sed}} = 2500$ 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:

$$V\frac{\partial\langle \mathrm{TP}\rangle}{\partial t} = \mathrm{P_{inp}} - \mathrm{P_{sed,net}} - \mathrm{P_{out}} \tag{7}$$

where $\langle \mathrm{TP}\rangle$ (mg m ^{3}) is the volume-averaged concentration of TP (Equation 5), $\partial\langle \mathrm{TP}\rangle/\partial t$ (mg m ^{3} yr ^{1}) is the rate of change of $\langle \mathrm{TP}\rangle$, P_{inp} (mg yr ^{1}) is the annual phosphorus input, $P_{\mathrm{sed,net}}$ (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 (7), all parameters represent annual or perennial averages. For our purposes, we used the linear model by Vollenweider (1969):

$$\frac{\partial\langle \mathrm{TP}\rangle}{\partial t} = \frac{1}{V}\mathrm{P_{inp}} - \sigma \cdot \langle TP\rangle - \frac{\beta}{\tau} \cdot \langle \mathrm{TP}\rangle \tag{8}$$

where $\mathrm{P_{sed,net}}$ from Equation (7) is assumed proportional to the total phosphorus content with the sedimentation rate constant σ (yr ^{1}) and the outflow $\mathrm{P_{out}}$ from Equation (7) is expressed by the water outflow (V/τ) times the average outflow concentration $\beta\ \langle \mathrm{TP}\rangle$ (where $\beta = \mathrm{TP_{surface}}/\langle \mathrm{TP}\rangle$). From Equation (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 $\tau^* \cdot \mathrm{P_{inp}}/V$ 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 and τ was replaced by Equation (4).

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 (Fig. 1), Lake Prespa and Lake Ohrid was collected from 2001 to 2003. In addition, rain water was collected at different altitudes (Table 2). The springs at St. Naum were sampled at a ~3 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 analyzed for TP and ions (Table 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 analyzed 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 h 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:

$$r_{PR} = \frac{C_{SN} - C_{PS}}{C_{PR} - C_{PS}} \cdot 100\% \tag{9}$$

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.

Effect on Lake Ohrid

In order to evaluate potential effects on Lake Ohrid, its P balance of the form of Equation (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). Water samples were treated and analyzed for TP using the same methods as for Lake Prespa. Three sediment cores were taken along the North-South axis of Lake Ohrid (Fig. 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 6) for all three cores. For Equation (8), the average, estimated between 1.5 and 3.5 cm sediment depth, was used as the current value of $P_{sed,net}$.

Results and discussion

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; Fig. 1).

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 3). For a consistency check, we compared this water balance with the seasonal level fluctuations of Lake Prespa measured by Hollis & Stevenson (1997) (Fig. 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 (Fig. 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 4 (third and fourth columns in Table 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 3. This

Table 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[a] [%]	Fraction during wet season[a] [%]
Runoff from subarea in E and SE (q 13 l km^{-2} s^{-1})	199		37	63
Runoff from karst subarea in W and SW (10% of total precipitation)	31		37	63
Runoff from subarea 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		

[a]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 4.

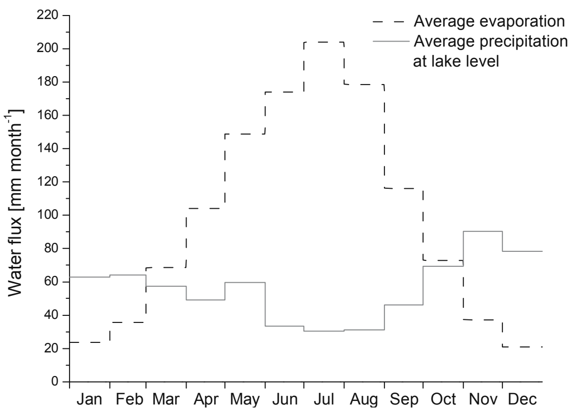

Figure 4. Seasonal evaporation (from Anovski et al., 2001) and precipitation (from Hollis & Stevenson, 1997; Ristevski et al., 1997) at the surface of Lake Prespa. 100 mm month^{-1} corresponds to 0.8 m^3 s^{-1}.

agreement indicates that the water balance by Anovski et al. (2001) is within the correct range.

However, this consistent water balance cannot explain the recent decrease of the Lake Prespa water level. This water loss, apparent in Figure 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 (Chavkalovski, pers. comm., 2002). According to Figure 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 ~5–6 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 (1) and (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 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.

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 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 (Fig. 1). Anoxic conditions prevail below 15 m from July to September 2003. This pattern has also been documented by others (Naumoski et al., 1997; Löffler 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^+ (Fig. 5b, c and d) indicates a significant input of organic material to the lake sediment. Figure 6 compares historic data from the work of Jakovljevic (1935) with recent measurements of DO from the same location. The most striking difference

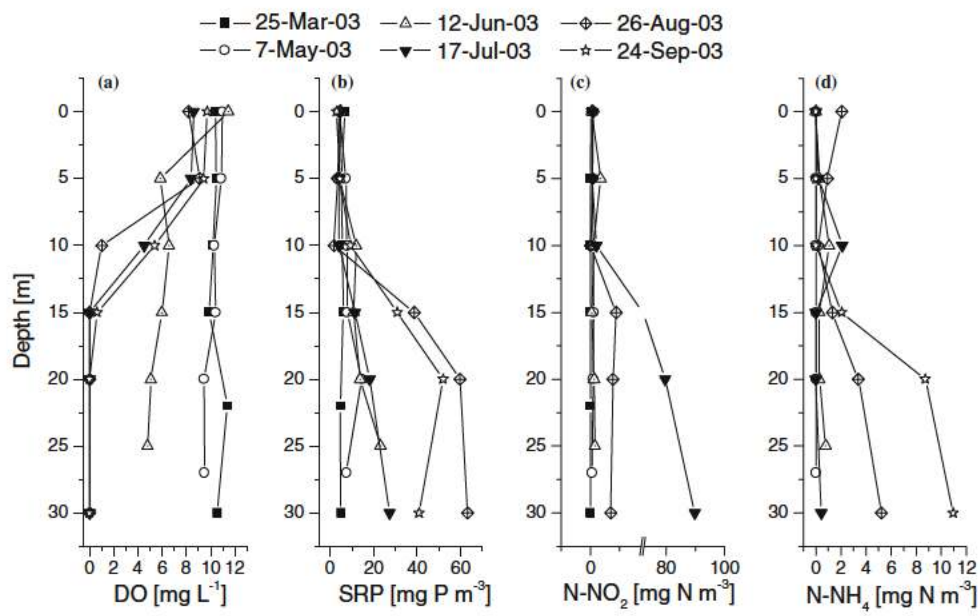

Figure 5. Development of (a) DO, (b) SRP, (c) NO_2^- and (d) NH_4^+ in the water column of Lake Prespa in the course of the productive season 2003.

appears in the summer profile (Fig. 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.

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 2000–2002 (Fig. 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 3.3 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 − 3.3 = 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.

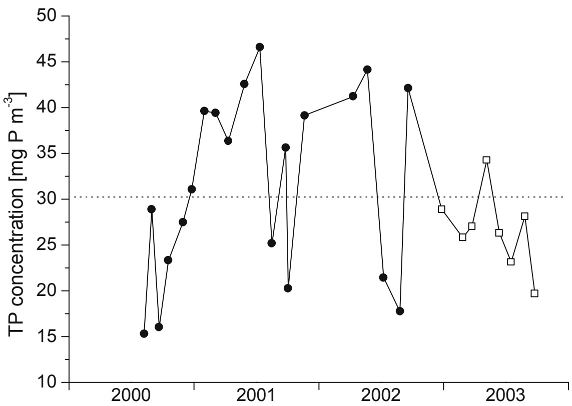

Figure 7. Seasonal development of average TP concentration in the top 15 m of Lake Prespa. The full circles are measurements by Jordanoski et al. (2002), the empty squares are our mea surements. The horizontal line is the average concentration over the whole period.

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; Fig. 8).

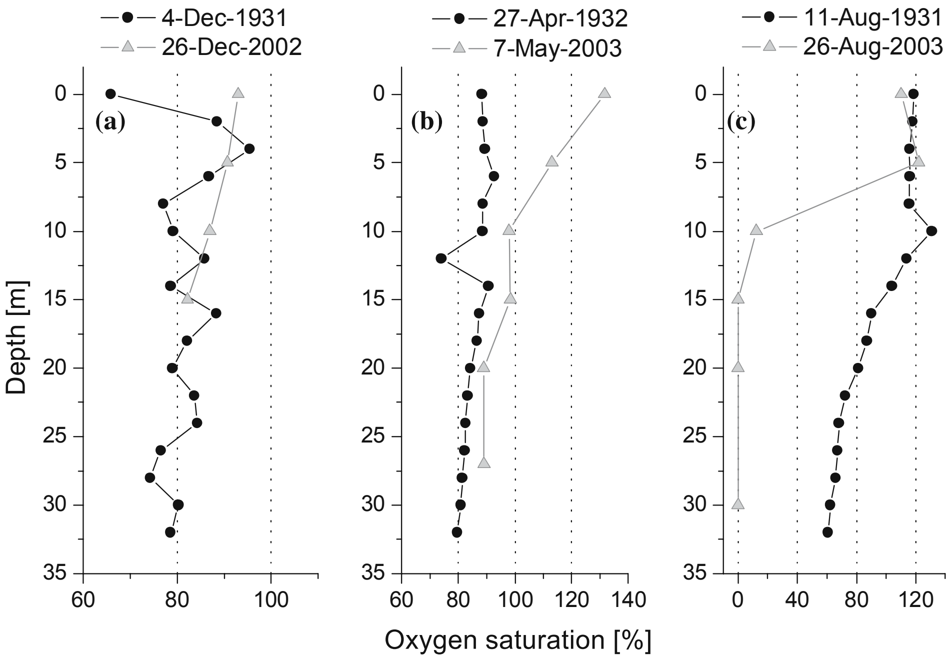

Figure 6. Comparison of oxygen saturation in profiles of Lake Prespa between historic results from Jakovljevic (1935) and recent measurements, for winter (a), spring (b) and summer (c). The surface point from December 1931 in Figure 6a is most probably a measurement error.

Such an increase in organic components towards the sediment surface is expected as a result of early diagenesis (e.g., Hupfer et al., 1995; Meyers & Ishiwatary, 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 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 (Fig. 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 (6), using the lake surface area as in Figure 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 (Fig. 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 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 (Fig. 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 (8)

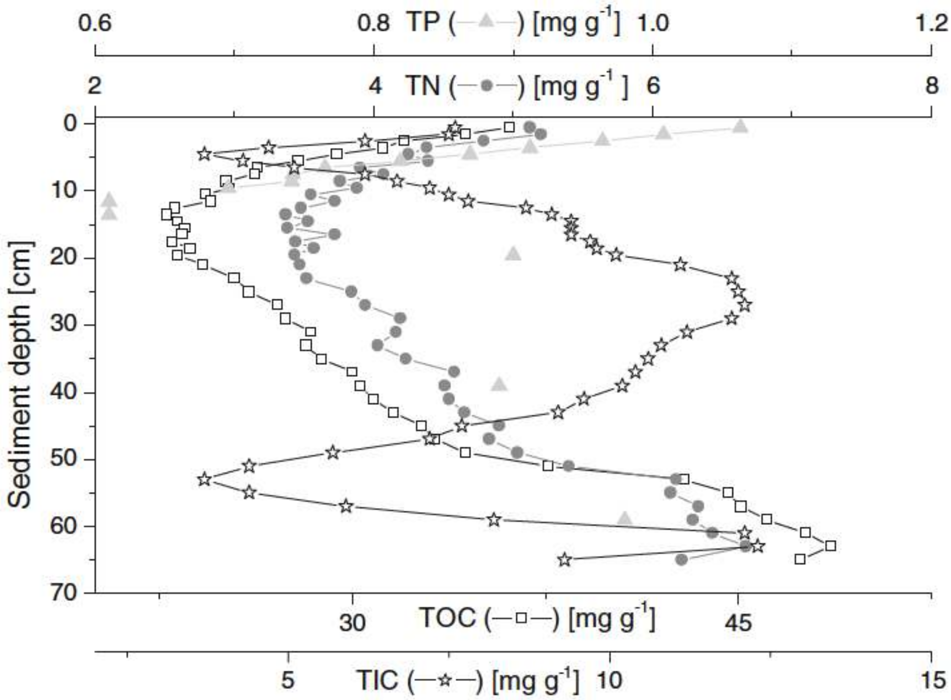

Figure 8. Content of TP, TN, TOC and TIC as measured in the sediment core from Lake Prespa, taken at 20 m depth in 2002 (Fig. 1).

Table 4. P balance of Lake Prespa and Lake Ohrid

P Balance[a]	Lake Prespa		Lake Ohrid
	Historic[b]	2003	2003
Equilibrium P concentration [mg P m^{-3}][c]	*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**

[a]P balance of Lake Prespa based on Equations (2, 4 and 8) and of Lake Ohrid based on Equation (8). Input data are in italics, results are in bold.
[b]For the historic situation of Lake Prespa the minimum P sedimentation rate and the 1992 TP concentration by Naumoski et al. (1997) were used.
[c]Equilibrium P concentration is the annual average of mean lake concentrations (Equation 5).

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 (4) and (8).

Results of these calculations are shown in Table 4. In this table the historic situation, before the recent increase of TP in the sediment core (Fig. 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 4 implies an increase in TP input from 52 to 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 t P 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 4).

Results from Equations (4 and 8) can be checked for plausibility with the observed internal balance. Figure 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 m, 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 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 (Fig. 5b). By depth-integrating the SRP profiles below 10 m (Equation 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., 2005). 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}.

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, pers. comm., 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.

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) color 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 $\sim$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 δ^{18}O from Anovski et al. (1980) and Eftimi & Zoto (1997) are plotted in Figure 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.

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, Fig. 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 (Fig. 1) and of rainwater percolating through the karst rocks, based on Equation (9). A similar balance can be done for all substances which are transported by underground water. In Table 5, the results for the stable isotopes δD and δ^{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). Measurements from the top 15 m 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 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 $\pm$ 5% of the spring water at St. Naum is fed from Lake Prespa and 57 $\pm$ 5% from local precipitation.

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 (Fig. 1). In Table 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 = 3.3 $m^3 s^{-1}$ (100 Mio $m^3 yr^{-1}$) unidentified (Tables 3 and 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

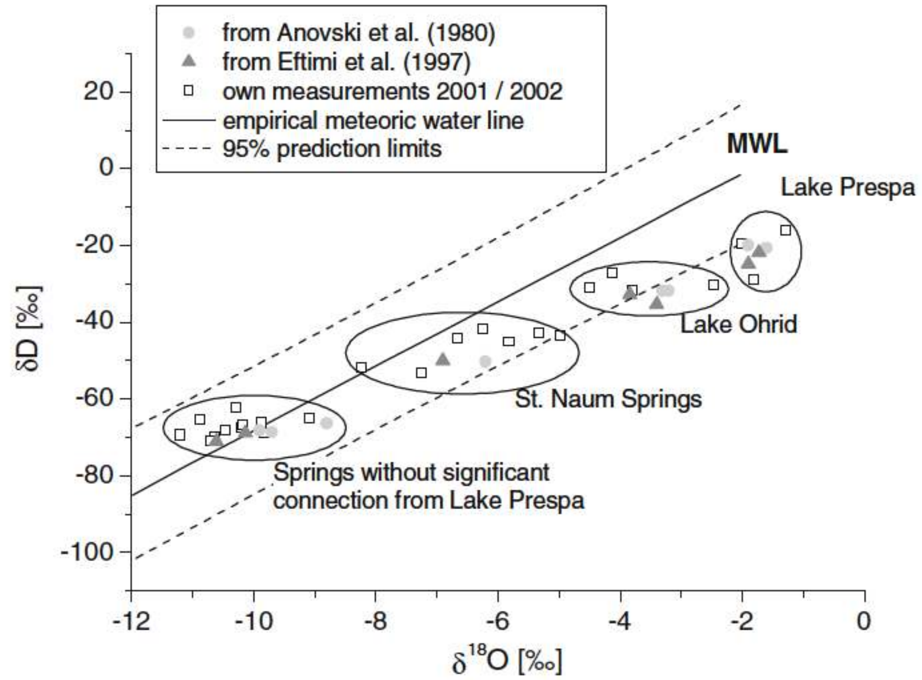

Figure 9. Ratios of stable isotopes δD and $\delta^{18}O$ in the two lakes, St. Naum Springs with a proven connection from Lake Prespa and various springs fed by precipitation only. The meteoric water line (MWL) is based on linear regression of local precipitation mea surements by us and Anovski et al. (1980).

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.

Table 5. Water composition at St. Naum springs

Substance	Share from Lake respa[a] [%]	Share from precipitation[a] [%]	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

[a]Ratios calculated, based on measured concentrations in Lake Prespa, in precipitation fed springs and in St. Naum Springs using Equation (9).

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

Table 6. Water flow from Lake Prespa in two spring areas

Spring area	Average discharge[a] [m^3 s^{-1}]	Share from Lake Prespa[b] [%]	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

[a]From Watzin et al. (2002).
[b]From own measurements for St. Naum Springs and from Eftimi & Zoto (1997) for the Tushemishti springs.

of 4 ± 1 mg P m ^{3} was found based on our sampling (Table 2, Fig. 10). According to this value and the water loads estimated from stable isotope and chloride measurements (Tables 5 and 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 (Fig. 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^{2+} 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_4^3 and Ca^{2+} species will be dominated by Ca^{2+} (Ca:P > 1000).

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

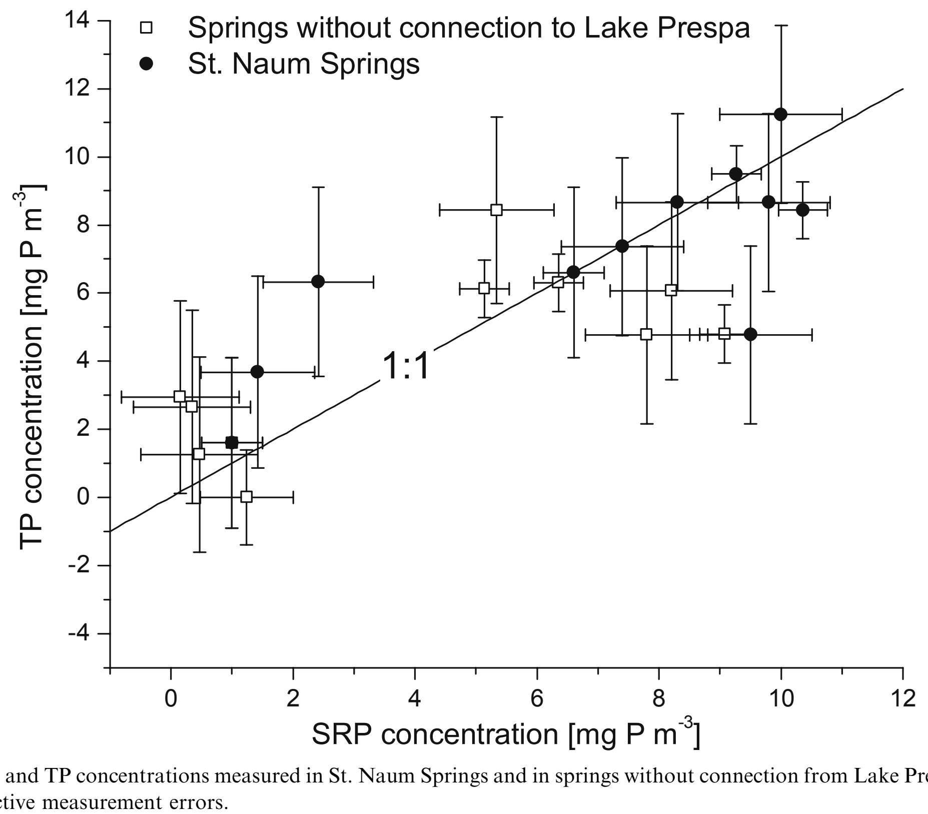

Figure 10. SRP and TP concentrations measured in St. Naum Springs and in springs without connection from Lake Prespa. Error bars show the respective measurement errors.

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), the underground flow can be calculated for different water levels (Fig. 11). The decline of the Lake Prespa water level since 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), we found the parameters σ = 0.15 yr^{-1} and β = 0.77 for Equation (8) (Table 4). Using the linear P model of Equation (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 8). (i) A smaller subterranean water flow $Q_{out,undgr}$ (Equation 2) consequently reduces P_{inp}, (ii) the decreasing Lake Prespa volume leads to an increase in P concentration 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 (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 11, the three effects (i) to (iii) on the phosphorus transport to Lake Ohrid are summarized using Equations (2, 4 and 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 pre-

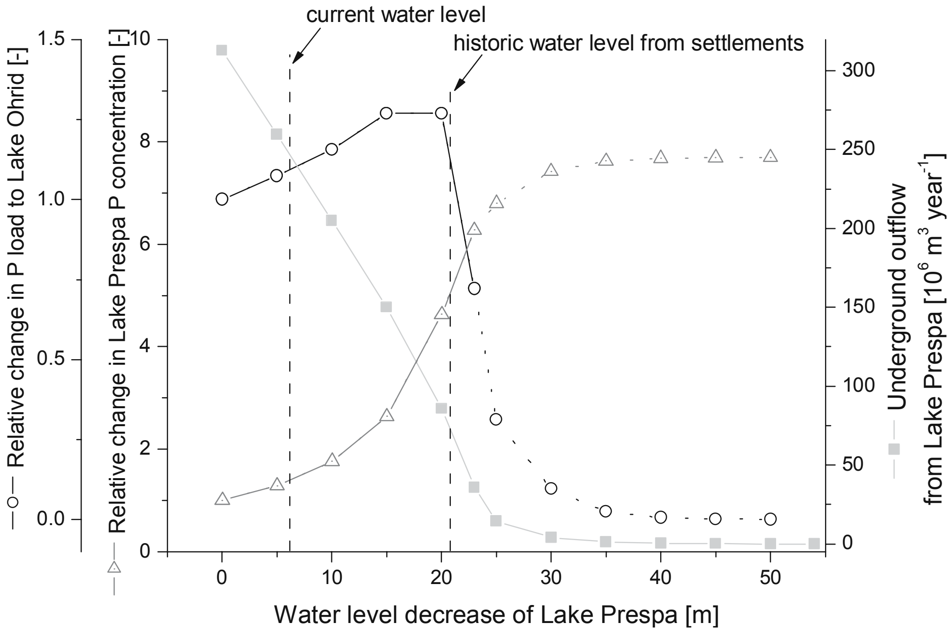

Figure 11. Effect of water level decline of Lake Prespa on its underground outflow, its P concentration and the P load to Lake Ohrid based on Equations (1), (2) and (8). The P load to Lake Ohrid is proportional to underground outflow times Lake Prespa TP concentration. Note that the figure is only valid 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) and (3) may no longer be appropriate.

dicted 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 (Fig. 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 (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.

Conclusions

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 four-fold 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 ultra-oligotrophic 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.

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.

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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