

General water chemistry and quality in a newly-created subtropical wetland lake

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Received 15 May 1997; accepted 23 June 1998

Key words: Lake Agmon, Lake Hula, Lake Kinneret, phosphorus, water quality

Abstract

Sixty km² of the southern Hula Valley (northern Israel) peat lands were flooded in 1994 as part of the Hula Valley Restoration Project. The small, shallow lake (110 ha, mean depth < 1 m) and network of ca. 90 km of canals created were designed to ameliorate problems (e.g., underground fires, soil subsidence, increased nutrient loading downstream to Lake Kinneret) resulting from drying the Lake Hula wetlands in the 1950s. This new wetland area now serves as the focus for developing eco-tourism in northern Israel. The initial development of this new ecosystem has been followed closely by a multi-disciplinary team of researchers, with an emphasis on water quality in the new lake and the potential impact of the project on Lake Kinneret. Here we report an overview of developments in general water chemistry of Lake Agmon during its first three years (1994–1996).

Water quality in Agmon was within general expectations for a shallow lake situated on peat. The first year of Agmon was characterized by the heavy influence of stream and drainage inflows with high pH, alkalinity, turbidity and electrical conductivity and high concentrations of sulfate and total dissolved solids. By the third year, however, many “in lake” processes (e.g., nutrient cycling and algal and macrophytic production) were well-developed and thus strongly affected lake water quality. Excessive phosphorus and nitrogen concentrations in the lake have led to hypertrophy, characterized by low dissolved oxygen concentrations and prolific blooms of nuisance algae. The management of this new ecosystem in the near future will require persistent, and innovative measures.

Introduction

A portion of the Hula Valley (in the northern watershed of Lake Kinneret, Israel) was flooded in the spring, 1994, in an attempt to ameliorate problems (e.g., underground fires, soil subsidence, increased nutrient loads to Lake Kinneret – Dimentman et al., 1992) that resulted from the drying of the Lake Hula wetlands in the 1950s. Known as the Hula Restoration Project (HRP), this endeavor created ca. 90 km of regulatory canals and small, shallow Lake Agmon (110 ha, mean depth < 1 m) which now serves as the focus for developing eco-tourism to be based on the assumed return of some of the many plant and animal species lost from the region. Moreover, the HRP is considered an important tool for management

of Israel's primary storage and supply reservoir ca. 20 km downstream, Lake Kinneret (Shaham 1996; Hambright and Zohary, 1998). The initial development of this new ecosystem has been followed closely by a multi-disciplinary research team emphasizing the water quality in Lake Agmon and the potential impact of the project downstream on Lake Kinneret. The primary purpose of the general water chemistry and quality program was to address the following key questions: 1) What will be the quality of water in Lake Agmon?; and 2) What will be the effect of the HRP on the quality of water entering Lake Kinneret?

Water quality is instrumental to the return and establishment of plant and animal communities to this restored portion of the extinct Hula wetland. Although there are general guidelines for the successful estab-

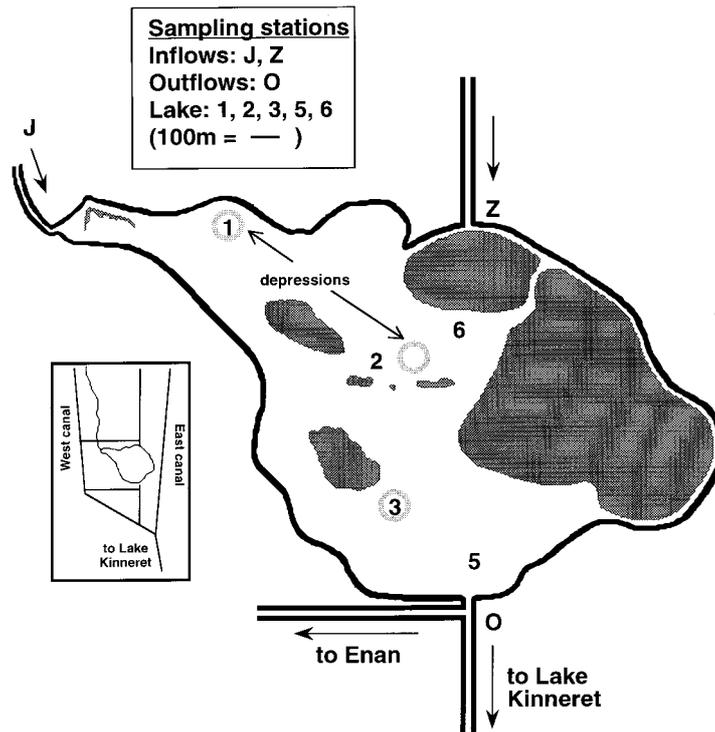


Figure 1. Detailed map of Lake Agmon showing sampling stations at the Jordan River inflow, J, peat drainage inflow, Z, outflow to Lake Kinneret, O, and in the lake, 1 (ca. 3 m deep), 2 (< 1 m), 3 (ca. 2.5 m), 5 (< 1 m), and 6 (< 1 m).

lishment and maintenance of wetlands (e.g., Mitsch and Gosselink 1993; Roggeri 1995), there is no formal, concrete definition of good water quality for wetland lakes. Rather, the functional characteristics of a particular wetland lake are generally used to ascribe qualitative definitions of water quality suitability for particular ecological or environmental components. For Lake Agmon, we adopted this approach as well, and defined good water quality as that which will be conducive to a rich assemblage of plant and animal life, but will remain within the constraints imparted by the regional geological and hydrological characteristics.

Here we report an overview of the general water chemistry of Lake Agmon during 1994–1996, and emphasize water quality. The question pertaining to the effects of the HRP on Lake Kinneret have been subjected to preliminary analysis elsewhere (Hambright and Zohary, 1996), though a more thorough and definite analysis will only be possible with more time. Certain biogeochemical- and algal-related parameters in Lake Agmon associated with water quality are given specific attention by Markel et al. (1998) and Zohary et al. (1998), respectively, and are not covered here.

This new ecosystem has already begun to foster abundant floral and faunal communities (e.g., Ashkenazi and Dimentman 1998; Kaplan et al., 1998) that will contribute, not only to successful restoration of a portion of a once lost ecosystem, but also to successful tourism development. However, an analysis of factors related to the re-flooding of previously dried peat soils, and some chemical and biological developments in the lake (e.g., low oxygen, algal blooms) suggest that the maintenance of good water quality in the future will require persistent and innovative management, with consideration of both Lake Agmon and Lake Kinneret.

Methods

Study site

Lake Agmon is a small, shallow lake (110 ha, mean depth < 1 m, maximum depth = 3.2 m) excavated in peat soils that were once the northern *Cyperus papyrus* swamp of the now extinct Lake Hula. The lake has two primary inlets, the Jordan River and the Z Canal, and one outlet. The Jordan River Inlet was constructed in

Table 1. Parameters included in the chemistry monitoring program and method of analysis.

Parameter	Method of analysis	Source
Total, total dissolved and nitrate nitrogen (NO_3^- -N)	Peroxodisulfate oxidation/széchrome NAS (diphenylamine sulfonic acid chromogene) method	Nydahl (1978), ARI (1977)
Nitrite (NO_2^- -N)	Sulfanilamide ethylenediamine method	APHA (1992)
Ammonia (NH_4^+ -N)	Phenolhypochloric method	Solórzano (1969)
Total and total dissolved phosphorus (PO_4^{3-} -P)	Persulfate oxidation/ascorbic acid method	APHA (1992)
Total dissolved solids	180 °C evaporation	APHA (1992)
Sulfate (SO_4^{2-})	Turbidimetric (BaCl_2) method	APHA (1992)
Alkalinity (CaCO_3)	Titration method	APHA (1992)
Turbidity (NTU)	Nephelometric method	APHA (1992)

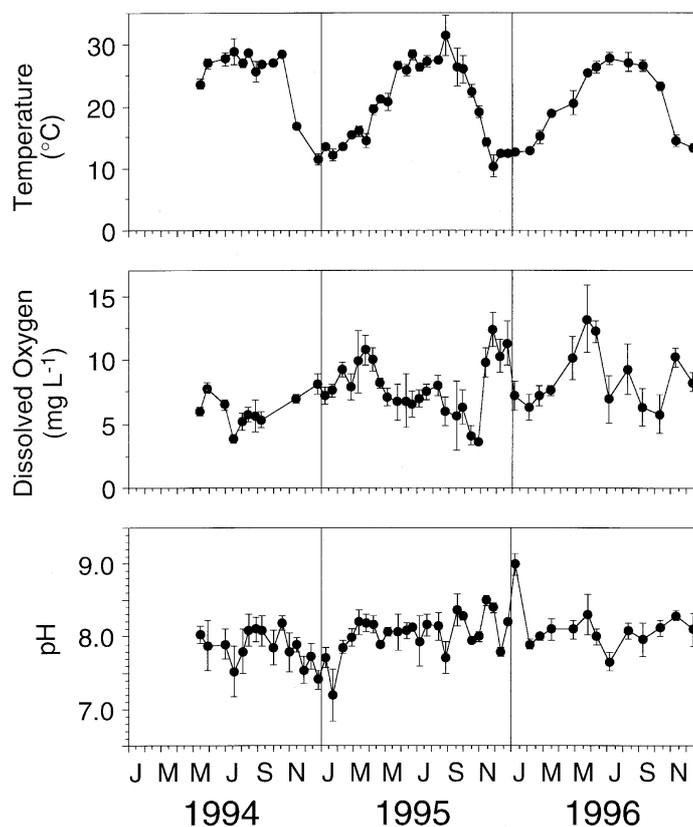


Figure 2. Biweekly (1994–1995) or monthly (1996) variations in temperature, dissolved oxygen and pH in Lake Agmon. Data are means \pm SD for all surface (0–60 cm) water samples from stations 1, 2, 3, 5, and 6.

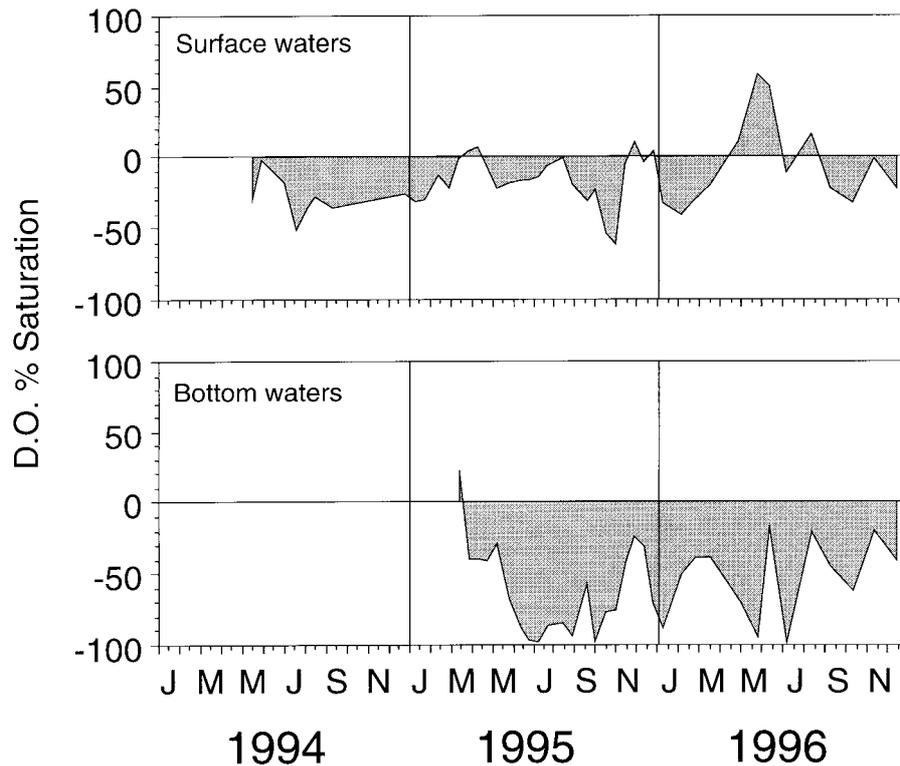


Figure 3. Relative dissolved oxygen concentrations (% of temperature-dependent saturation concentrations) in Lake Agmon surface waters (upper panel) and bottom waters (lower panel) at stations 1 and 3 during 1994–1996. Bottom waters were not sampled before March 1995.

the former bed of the Jordan River that served Lake Hula and now conveys ca. 2% of the annual Jordan River flow through the Agmon wetland (the remainder bypasses the wetland via the East and West Canals of the Jordan River – see Hambright and Zohary, 1998). Unlike the fresh water from the Jordan River, the Z Canal carries drainage waters from the agricultural areas north of the lake. Due to intense irrigation and maintenance of a high water table (Tsipris and Meron, 1998), the peat soils in this area impart to the water a characteristic brown-humic stain. The amount of water flowing into Lake Agmon through the two inlets varies with rainfall and water needs in the Hula Valley, though ca. 75% of the inflowing waters arrive via the Jordan Inlet. Outflows from the lake can be pumped into a large agricultural irrigation supply reservoir or continue downstream to Lake Kinneret.

Monitoring

Chemical monitoring in Lake Agmon consisted of biweekly (1994–1995) or monthly (1996) sampling from 8 stations in the lake and inflowing and outflowing canals (Figure 1). All sampling was con-

ducted between 07:00 and 10:00 hour local time. Samples for chemical analyses were collected using a depth-integrated (0–2 m) pipe sampler (May 1994–Feb 1995), or, either a 2- or 3.2-L van Dorn water sampler (Mar 1995–Dec 1996) at discrete depths (stations 1 and 3 – surface, mid-water, bottom; all other stations – mid-water only). The sampled waters were placed into 1-L plastic bottles, stored on ice and transported within 2 hours to the laboratory where they were processed according to standard methods specific for each parameter (Table 1). During 1995 and 1996, DO was determined periodically on discrete-depth samples by Winkler titration. Measurements of temperature and DO were also made at 20-cm intervals from the surface to the bottom in all lake stations and at mid-depth in Stations J, O, and Z using appropriate probes.

Results and discussion

Water temperature and DO in Lake Agmon ranged between 10 and 30 °C and 0 and 14 mg L⁻¹, respectively (Figure 2). Dissolved oxygen supersaturation

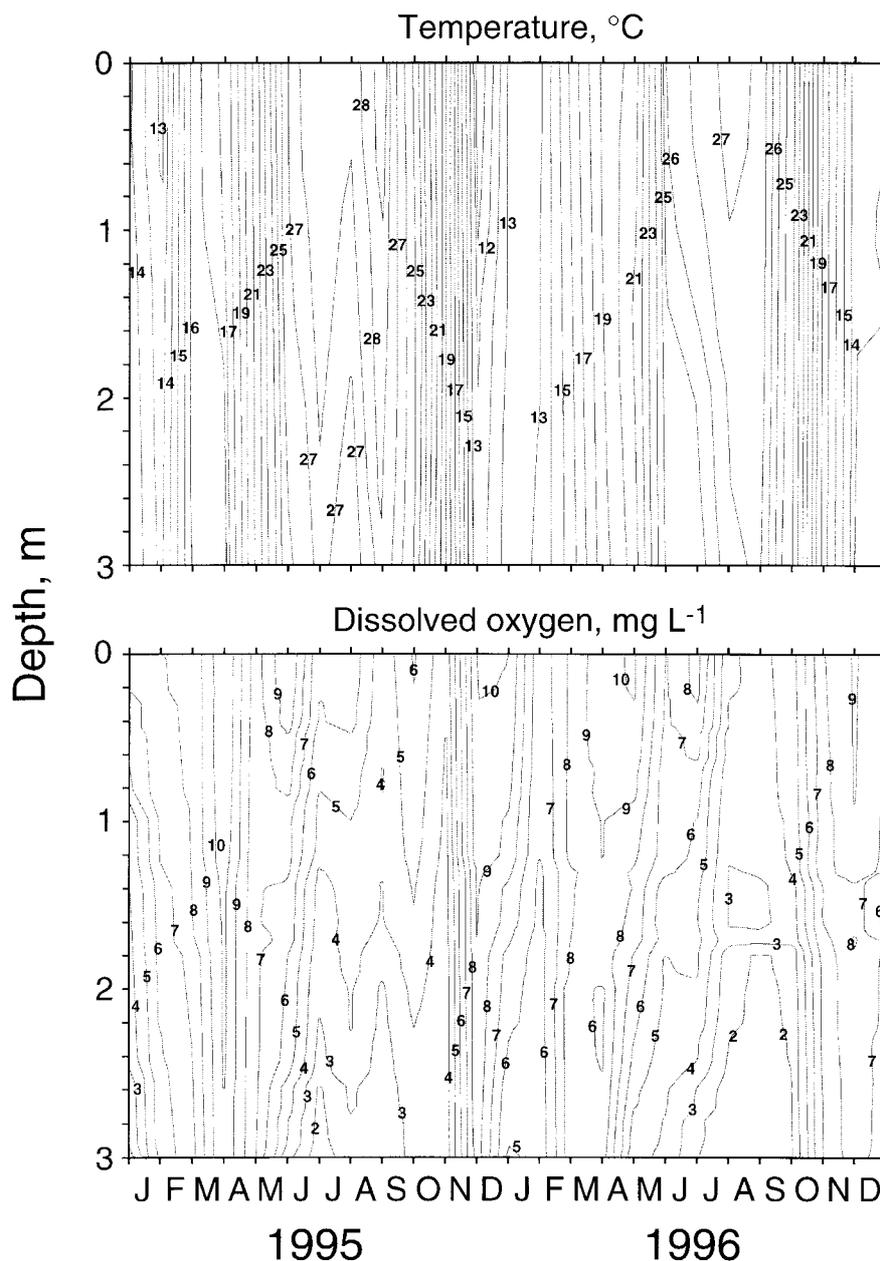


Figure 4. Isoleth diagram showing seasonal variation in temperature and dissolved oxygen as functions of depth in Stations 1 and 3 during 1995 and 1996.

and undersaturation at various times in both years demonstrates that other factors besides temperature were important in regulating DO (Figure 3). For example, during periods of high benthic algal production (e.g., spring 1995 and 1996) the DO concentrations in the upper waters exceeded temperature-dependent saturation concentrations, while extreme cases of un-

dersaturation (typically in summer, but also in other periods) occurred in conjunction with phytoplankton blooms (see Zohary et al., 1998 for further details). The DO concentrations in the bottom waters of stations 1 and 3 were typically below saturation, with mean (\pm SD) summer concentrations of 1.1 ± 1.5 and 4.5 ± 3.0 mg L⁻¹ during 1995 and 1996, respectively.

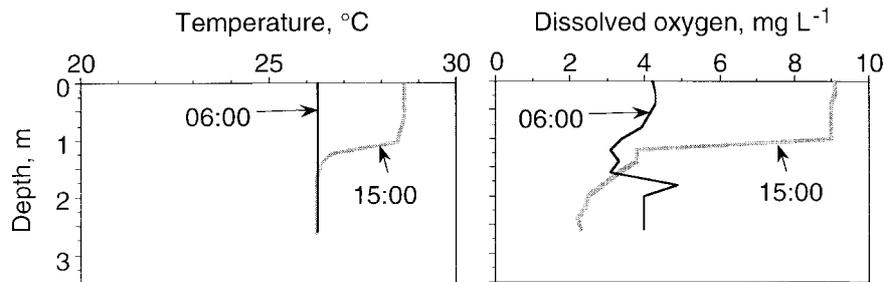


Figure 5. Depth profiles of mean temperature and dissolved oxygen concentration at Stations 1 and 3 in Lake Agmon at 06:00 and 15:00 hours on 22 September 1994.

Table 2. Comparison of Jordan River water quality entering Lake Agmon as affected by peat soils. Data are means (\pm SE) for all sampling dates in 1995 and 1996 for Station J ("pre-peat") and Station Z ("post-peat"). An asterisk (*) indicates significant differences ($P \leq 0.05$) detected between stations using an unpaired t-test of the null hypothesis $H_0: \mu_J = \mu_Z$.

	"Pre-peat" – J		"Post peat" – Z	
	mean	(SE)	mean	(SE)
Temperature, °C	18.4	(1.0)	19.2	(1.3)
Dissolved oxygen, mg L ⁻¹	6.7	(0.5)	5.7	(0.5)
Alkalinity, mg CaCO ₃ L ⁻¹	153	(6)	168	(7.7)
Total phosphorus, μ g L ⁻¹	59	(10)	77	(13)
Total dissolved P, μ g L ⁻¹	29	(3)	24	(3)
pH	8.04	(0.05)	*	7.58 (0.12)
SO ₄ ⁻² , mg L ⁻¹	44	(13)	*	618 (111)
Total dissolved solids, mg L ⁻¹	277	(18)	*	1368 (190)
Turbidity, NTU	2.3	(0.5)	*	24.5 (5.8)
Conductivity, mS	0.40	(0.03)	*	1.6 (0.2)
N-NH ₄ ⁺ , mg L ⁻¹	0.11	(0.04)	*	5.87 (2.09)
N-NO ₃ ⁻ , mg L ⁻¹	1.20	(0.21)	*	10.59 (3.89)
Total nitrogen, mg L ⁻¹	3.05	(0.85)	*	10.07 (2.90)

Vertical profiles of temperature at Stations 1 and 3 revealed thermal stratification during the morning throughout most of the summer (Figure 4). On many days, strong afternoon winds destroyed this stratification, making the lake polymictic. The morning temperature gradient was generally only a few degrees. Nevertheless, vertical density differences between adjacent water layers at the high summer temperatures typical for Lake Agmon (27–29 °C) can be high relative to the thermal differences, making stratification extremely stable (Hutchinson, 1957; Hambright et al., 1994). Consequently, DO profiles revealed substantial vertical stratification in the water column during summers, with gradients of up to 5 mg O₂ L⁻¹. DO gradients, even in the upper meter of the lake, also revealed that at times, oxygen consumption (via organic

matter decomposition and respiration) in the lower waters often exceeded the input of oxygen into the system via algal production and re-aeration from the atmosphere (for example, note DO profiles in May–June, Figure 4 and the mid-day decline of DO in bottom waters, Figure 5), attesting to the importance of respiration in shallow, highly productive ecosystems (Mitsch and Gosselink, 1993; V.-Balogh and Vörös, 1997).

It is important to consider that the measurements of temperature and DO were made in the morning hours. These data tend to reflect conditions that were prevalent during the night time – cooler surface temperatures and no photosynthetic production of oxygen. Temperature and DO profiles taken at 06:00 and 15:00 hours on 22 Sept 1994 (Figure 5) revealed the dramatic

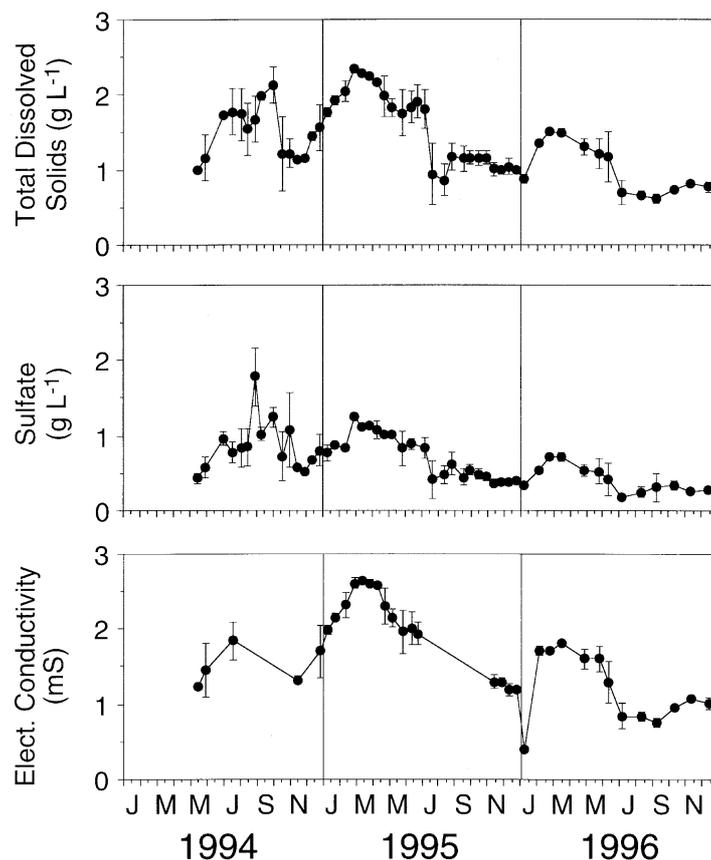


Figure 6. Biweekly (1994–1995) or monthly (1996) variations in total dissolved solids, and sulfate concentrations and in electrical conductivity in Lake Agmon. Data are means \pm SD for all surface (0–60 cm) water samples from stations 1, 2, 3, 5, and 6.

change in vertical structure of the water column created by day-time warming of the surface waters and algal photosynthesis.

The waters of Lake Agmon, in general, reflected a strong influence of inflowing waters from both the Jordan River and the Z Canal. The lake exhibited high pH (fluctuating mostly around 8; Figure 2) and alkalinity (ca. 3 meq L^{-1}) throughout the 3-yr period, reflecting in general the strong effect of Jordan River inflows (Table 2). Short-term fluctuations in pH and alkalinity also reflect the effects of organic matter decomposition in the bottom sediments and deep depressions (e.g., station 3; see also Markel et al., 1998) and the relatively high algal productivity in spring (benthic mats) and summer (phytoplankton) (Zohary et al., 1998).

The concentrations of total dissolved solids and sulfate, as well as electrical conductivity, were always very high in the lake (Figure 6). Comparison of these parameters in the two inflow sources (Jordan and Z Canal) with in-lake concentrations (cf. Table 2, Figure

6) demonstrates the impact of the peat soils on lake water chemistry. Although the Jordan River was the primary source of water for the lake (ca. 75% – Markel et al., 1998), the overall salinity in the lake was usually similar to that of water draining the peat soils from the north (i.e., the lake contained high concentrations of peat drainage constituents). The primary constituents of TDS were sulfate and calcium derived from dissolution of secondary gypsum that formed when the Hula peat soils were dried in the late 1950's (Markel et al., 1998). The decline in TDS, sulfates, and conductivity over the three-year period suggests that the “peat soil effects” on Lake Agmon diminished through time. Part of this decline could be due to a gradual depletion of some ions from the upper peat soils that had previously accumulated over the four decades of dehydration.

In contrast, trends evident in DO (see above), turbidity, nitrogen, and phosphorus (Figure 7) indicate that “internal lake effects” strengthened over the

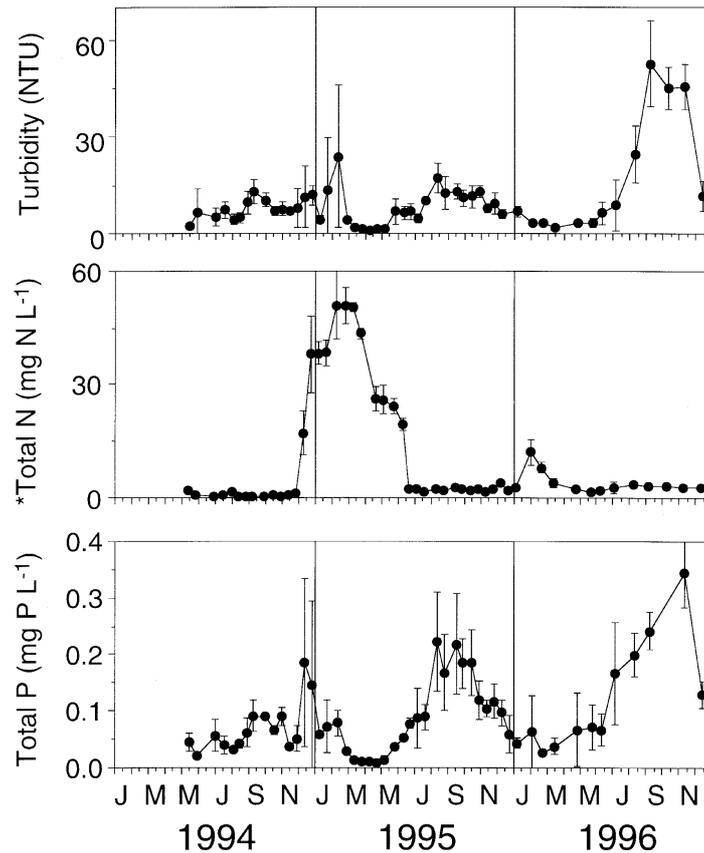


Figure 7. Biweekly (1994–1995) or monthly (1996) variations in turbidity, total nitrogen, and total phosphorus in Lake Agmon. Data are means \pm SD for all surface (0–60 cm) water samples from stations 1, 2, 3, 5, and 6. * Total nitrogen values shown prior to March 1995 are total dissolved inorganic nitrogen and do not include particulate or dissolved organic forms.

course of the first three years of this new lake. Turbidity values were always high in the lake as in the inflowing Jordan River waters. However, sharp increases in turbidity in the summers of 1995 and 1996 reflect the intensity of the summer algal blooms (Zohary et al., 1998). The main source of nitrogen for Lake Agmon was nitrate and like sulfate, and was derived from the peat soils mainly via nitrification of ammonia produced from organic matter decomposition (Avnimelech et al., 1978). The strong dependency of in-lake concentrations on inflows can be seen especially in winter-spring 1994/95, in which flooding in the Hula peat soils washed huge amounts of nitrates and ammonia into the lake (Figure 8). For example, during Dec–Jan of this winter, nitrate concentrations in the inflow waters ranged between 70 to 133 mg N L⁻¹. In contrast, during the drier seasons, nitrate concentrations in the inflows were much lower (typically ca. 1 mg L⁻¹) and there was a shift in the predom-

inant forms of nitrogen to particulate and dissolved organic forms as in-lake biological processes became increasingly prevalent.

The most prominent indication of the shift from “peat soil effects” to “internal lake effects” is the three-year trend of increasing summer phosphorus concentrations in the lake (Figures 7 and 8). Interestingly, there seems to be little effect of Jordan River inflows on phosphorus concentrations in the lake. With the exception of a high input of phosphorus to the lake during the wet 1994/95 winter, phosphorus concentrations in the Jordan River rarely surpassed 0.1 mg L⁻¹ and were more typically below 0.06 mg L⁻¹. Nevertheless, phosphorus concentrations in Lake Agmon during summer generally exceeded 0.1 mg P L⁻¹, with peak concentrations in excess of 0.3 and 0.4 mg L⁻¹ in 1995 and 1996, respectively.

Details of the exact mechanisms involved in the high summer phosphorus concentrations are as yet un-

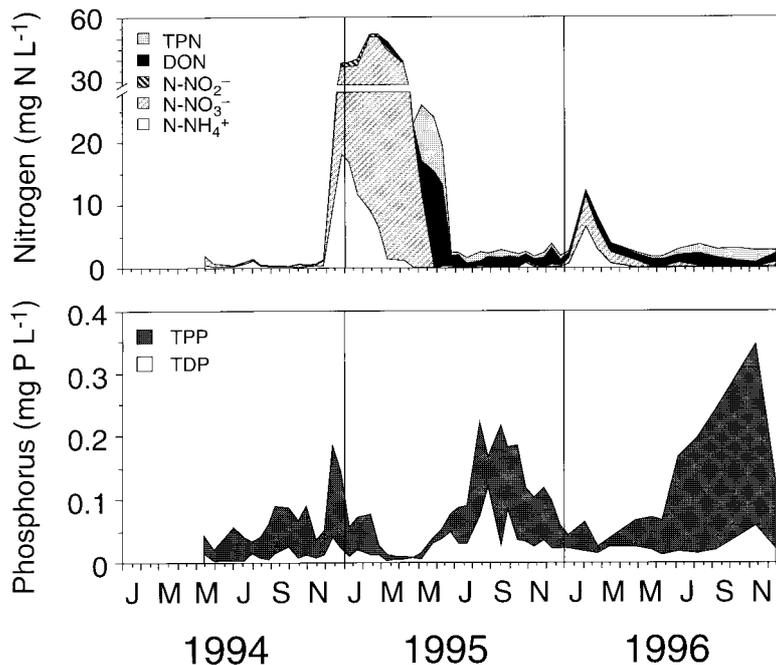


Figure 8. Biweekly (1994–1995) or monthly (1996) variations in nitrogen (TPN, total particulate nitrogen; DON, dissolved organic nitrogen; N-NO₂⁻, nitrite nitrogen; N-NO₃⁻, nitrate nitrogen; NH₄⁺, ammonia nitrogen) and phosphorus (TPP, total particulate phosphorus; TDP, total dissolved phosphorus) fractions in Lake Agmon. Data are means for all surface (0–60 cm) water samples from stations 1, 2, 3, 5, and 6. Data for TPN and DON prior to March 1995 are not available.

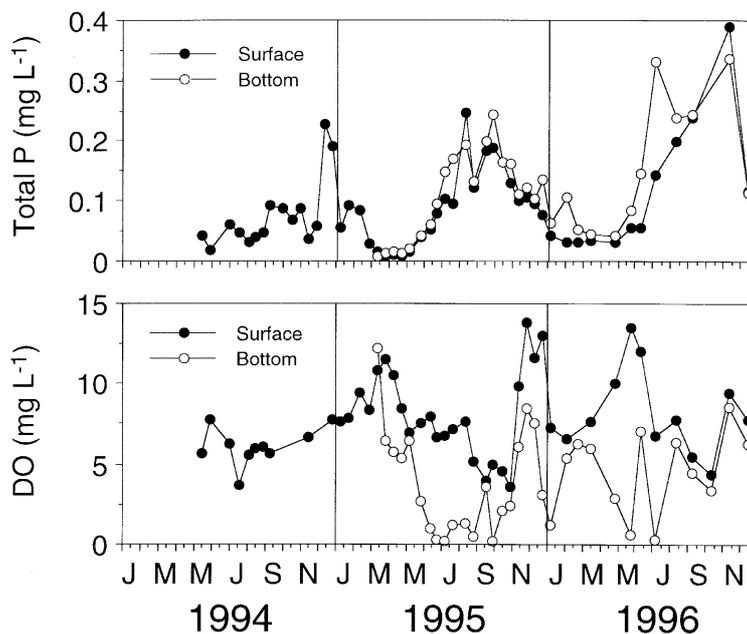


Figure 9. Biweekly (1994–1995) or monthly (1996) variations in total phosphorus and dissolved oxygen concentrations in surface and bottom waters of stations 1 and 3 in Lake Agmon. Data are means for all surface (0–60 cm) water samples from stations 1 and 3. Data for bottom waters prior to March 1995 are not available.

clear. However, we speculate that they are in part due to redox- and pH-dependent phosphorus release from the sediment-water interface (*sensu* Frevert, 1979, 1980; Eckert et al., 1997) of phosphorus either in the peat soils or of phosphorus deposited on the lake bottom during the winter high loading period.

The primary pools of phosphorus in Lake Agmon sediments are the particulate fractions; calcium-, metallo- and organic matter-bound phosphorus (Hambright and Eckert, unpubl.). A fourth pool of phosphorus is the dissolved organic and inorganic forms in the interstitial (or pore) waters within the sediments. The majority of calcite-bound phosphorus originates from the Jordan River headwaters and is not readily soluble at high pH. Therefore much of this P source is probably unavailable to the biota of Lake Agmon. The second major pool of phosphorus in the lake sediments is that bound within organic material in the peat soils which can be released upon organic matter decomposition. As submerged peat soils rapidly become anoxic, decomposition occurs via the anaerobic microbially-mediated processes such as fermentation, nitrate and sulfate reduction, and methanogenesis (Ponnamperuma, 1972; Mitsch and Gosselink, 1993; Reddy and D'Angelo, 1994; Markel et al., 1998). Although the phosphorus is probably re-mineralized, its ultimate fate is unknown. However, given the prevailing pH and redox conditions in the sediments and those of the overlying water (see Markel et al., 1998), this phosphorus probably becomes adsorbed by iron oxides at the oxic-anoxic interface. This metallo-bound phosphorus would be released into the overlying water if, and when, the overlying water became anoxic. Although we do not have detailed data of redox conditions in the water column, especially near the sediments, comparisons of surface and deep water (> 2.5 m) concentrations of DO and phosphorus support the role of metallo-bound phosphorus as a potential source for the summer increases (Figure 9). DO concentrations were consistently lower at 3 m compared with surface concentrations while TP concentrations were consistently higher at 3 m. Although these data are not necessarily indicative of the dynamics at the sediment-water interface in the mostly < 1 m deep lake, they do provide insight into the possibility of redox-dependent phosphorus release from the sediments. Moreover, the accumulation rates of total phosphorus during the summers of 1995 and 1996 (calculated as the slope of the increasing portion of the summer peaks in Figure 7; 1.26 and 0.86 $\mu\text{g P L}^{-1} \text{d}^{-1}$, respectively) are well below the potential

redox-dependent release rates expected for sediments containing ca. 1 mg P g_{dw}^{-1} of sediments and undergoing daily fluctuations between oxic and anoxic conditions (Eckert and Trüper, 1993).

Comparison of inflow (Stations J and Z) and outflow (Station O) phosphorus concentrations in 1994 and 1995 (unpubl.) suggested that Lake Agmon served as a sink for inflowing phosphorus, as concentrations in the outflow were always lower than in the inflow waters. It is possible to estimate a general phosphorus mass balance based on concentrations in the lake and inflowing waters (Table 3) by using an approach detailed in Markel et al. (1998) in which it is assumed that the ratio of Jordan River to Z Canal inflows was 75:25 and that evaporation averaged ca. 25% of inflows annually. This rough mass balance also suggests that in 1994, Lake Agmon served as a phosphorus sink. However, in 1995 and 1996, the lake was probably a source for phosphorus downstream, with 8 and 47%, respectively, more phosphorus in the lake than can be explained by inflow sources.

Interestingly, if the lake was indeed a source for phosphorus downstream, the "One Gram Rule" developed by Richardson et al. (1997) does not seem to accurately predict phosphorus dynamics for Lake Agmon, because loading rates were likely < 1 g P $\text{m}^{-2} \text{yr}^{-1}$. According to the "One Gram Rule", phosphorus will be efficiently retained by wetlands via peat/soil accretion as long as loading rates are at or below 1 g P $\text{m}^{-2} \text{yr}^{-1}$. Based on mean inflow phosphorus concentrations given in Table 2, an average daily inflow of 25,000 $\text{m}^3 \text{d}^{-1}$ (Markel et al., 1998) and a lake surface area of 100 ha, phosphorus loading in Lake Agmon was only ca. 0.6 g P $\text{m}^{-2} \text{yr}^{-1}$. Although this value is below 1, the actual rate for permanent P storage in Lake Agmon sediments is unknown. Moreover, phosphorus dynamics in Lake Agmon may be more heavily influenced by high calcium levels or other factors besides soil accretion (Richardson et al., 1997).

It should be noted that the process of phosphorus release from the sediments is not necessarily restricted to the lake. There are nearly 60 km^2 of peat soils in the Hula Valley that have been flooded (via elevation of the ground water table). These soils are anoxic below the water's surface and reducing conditions are therefore prevalent (unpublished data). As such, an understanding of the dynamics of phosphorus is essential for proper management of this new ecosystem.

Table 3. Annual mass balance of phosphorus for Lake Agmon during 1994–1996 based on annual mean concentrations of P (mg L^{-1}) in the lake ($P_{\text{(lake-actual)}}$) and its inflows (P_J – Jordan Inlet and P_Z – Z Canal) and outflow (P_O – O Canal); number of samples indicated in parentheses. Internal loading was calculated as $P_{\text{(Internal load)}} = P_{\text{(lake-actual)}} - P_{\text{(lake-theoretical)}}$, where $P_{\text{(lake-theoretical)}}$ was calculated according to Markel et al. (1998) as $P_{\text{(lake-theoretical)}} = (0.75P_J + 0.25P_Z)(\text{Evaporation degree})$, where evaporation degree was estimated as 1.25 (i.e., 25% of annual inflows).

	1994	1995	1996
$P_{\text{(lake-actual)}}$	0.071 (45)	0.099 (89)	0.117 (65)
P_J	0.108 (8)	0.071 (19)	0.041 (13)
P_Z	0.093 (2)	0.077 (14)	0.077 (13)
P_O	0.071	0.099	0.117
$P_{\text{(lake-theoretical)}}$	0.130	0.091	0.063
$P_{\text{(Internal load)}}$	-0.059 (84%)	0.008 (8%)	0.054 (47%)

Conclusions

The primary goals of this monitoring program were in reference to water quality, both within the newly created lake and flowing downstream to Lake Kinneret. Although this term is ubiquitous throughout the aquatic sciences literature, its definition is often ambiguous (Parparov and Hambright, 1996; Boon and Howell, 1997). Clearly the definition of water quality will be dependent on the viewpoint of those providing the definition as well as the system for which the definition is intended. Therefore, any meaningful discussion of water quality pertaining to this newly created system must be preceded by formal definition.

Using the system of water quality developed by Smith (1990) for New Zealand streams and lakes, and recently suggested for use in Israel for Lake Kinneret (Hambright et al., 1998), “good water quality” for water resources should be defined as that which is “suitable for all intended uses”. Hence, for Lake Agmon and the surrounding canals, we should define water quality based on the Hula Project goal of establishment of suitable wetland habitats for the return of much of the previously absent Lake Hula flora and fauna that flourished prior the 1958 drainage (i.e., eco-tourism). For waters flowing downstream to Lake Kinneret, the definition should again be based on the project goal of reducing nitrate loading to Lake Kinneret, but until the final stages of the project are completed this aspect can not be appropriately assessed. To date, most researchers in the Hula Project would consider the project successful with respect to re-establishment of much of the “natural” flora and

fauna to the region (see Ashkenazi and Dimentman, 1998; Kaplan et al., 1998; Shy et al., 1998; Zohary et al., 1998). As such, the variability in the various chemical (and biological) parameters thus far observed in Lake Agmon provide a reasonable basis for defining the limits of “good water quality” for this new ecosystem.

Early speculation that water quality in Lake Agmon would be determined primarily by peat soils (with high concentrations of nitrates, sulfates, etc.) has given way to the reality that high in-lake phosphorus concentrations and algal blooms will also be principal determinants of water quality in the lake in the future. The level of eutrophy is high and cyanobacterial blooms are common and persistent (see Zohary et al., 1998). There are indications that the level of eutrophy increased gradually over the first three years of this new system (e.g., summer DO and phosphorus concentrations, also see Zohary et al., 1998). Whether this trend will continue further or whether it reflects the development from initial creation to present steady state conditions can only be addressed with time. Nevertheless, we suggest that continued success of this project, especially regarding the future of eco-tourism, will be dependent on management’s efforts to prevent stagnation (i.e., large-scale anoxia and sulfide- and metals-based toxicity, fish kills, habitat loss, etc.) of this newly created wetland.

Acknowledgments

We thank J. Easton, B. Kaplan, and E. Yas’ur for field work and V. Gandel-Flerovsky, R. Shraks, D. Eliyahu,

S. Shmerkin, C. Michaeli, E. Hadar, N. Pinhas, D. Sinanski, and S. Yehskal for chemical analyses. Discussions with D. Markel were helpful in establishing the potential role of metallo-bound sediment phosphorus in in-lake phosphorus dynamics. We also thank T. Zohary for providing invaluable discussion and comments on early drafts of this manuscript, and A. Harpaz and G. Shaham for adamantly pursuing the realization of the restoration project and for inviting our participation. Funding was provided jointly by Keren Kayemet L'Israel (JNF), Israel Lands Authority and the Israel Ministry of Agriculture.

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