Hydrobiologia (2005) 548:85–99
DOI 10.1007/s10750-005-1149-6

Primary Research Paper

Water-level fluctuations in Mediterranean reservoirs: setting a dewatering threshold as a management tool to improve water quality

Luigi Naselli-Flores* & Rossella Barone
Dipartimento di Scienze Botaniche, Via Archirafi, 38-90123, Palermo, Italy
*(*Author for correspondence: E-mail: luigi.naselli@unipa.it)*

Received 4 January 2005; in revised form 16 March 2005; accepted 27 April 2005

Key words: phytoplankton dynamics, cyanoprokaryotes, Mediterranean climate, eutrophication

Abstract

Water-level fluctuations, often linked to seasonal climatic trends, are a natural phenomenon which occur in almost all aquatic ecosystems. In some climatic regions, as the Mediterranean one, they are particularly wide due to the occurrence of two well separated periods: the rainy winter and the almost completely dry summer. Precipitation is concentrated in the first period, whereas in the second strong evaporation losses take place. According to these climatic features, and to ensure a continuous supply of water throughout the year, man-made lakes store water during winter and are subjected to dewatering during summer to compensate the lack of precipitation. These ecosystems are thus characterised by rather wide water level fluctuations which were observed to transform them from potentially warm monomictic lakes into polymictic or atelomictic ones. These changes deeply affect the biological structure and the functions of the water bodies impairing the response of some ecosystem properties, as resilience and resistance, since the impacts are immense enough to move the systems out of their homeostatic plateau of, respectively, deep or shallow lakes. In order to understand to what extent a reservoir can be ''emptied'' without changing its ecosystemic identity (deep or shallow lake *sensu* Padisák & Reynolds, 2003) and to set a ''dewatering threshold'', the results from two different hydrological years, one with a dewatering so intense as to disrupt thermal stratification in midsummer, and the other one with water enough to allow the maintenance of the reservoir's thermal structure throughout the summer, are compared. Former investigations have shown that the persistence of thermal stratification has a positive value in Sicilian reservoirs: a notable decrease in total phytoplankton biomass and in the relative occurrence of cyanoprokaryotes was observed in the high-level year with a stable thermal stratification. Although the solving of the external load problems causing eutrophication phenomena remain the main task to improve the water quality of this Mediterranean island, a management procedure, based on the maintaining of the ecosystem within its homeostatic plateau through the setting of a dewatering threshold, is suggested.

Introduction

Aquatic ecosystems are naturally subject to water-level fluctuations (WLF). These are mainly due to seasonal variation in the abundance of precipitation and/or in the increase of evaporative losses.

In Mediterranean regions, due to the segregation of two well-separated seasons, the wet winter, where most of precipitation is concentrated, and the arid summer with no precipitation, water level fluctuations generally show a wide range. Actually, one of the most common typology of natural aquatic ecosystem in this area is represented by astatic waters: water bodies characterised by very wide vertical movements of water surface that can lead to a complete desiccation (temporary waters). In this last case, WLF have a selecting effect on the biotic components of these ecosystems, allowing the success of organisms producing resting stages (Williams, 1987).

In Sicily, the largest and more populated island of Mediterranean Sea, a strong increase in the number of artificial lakes occurred in the second half of the last century to meet the always increasing demand of a constant availability of water for both drinking and irrigation purposes throughout the year, as well as the necessity to store water to produce energy. According to Mediterranean climate, the most characteristic feature of Sicilian man-made lakes is represented by marked water-level fluctuations (Naselli-Flores, 2003).

In spite of the vital importance that freshwater resources represent in this Mediterranean island, the last decades were signed by the lack of any management procedure of both water bodies and their catchments areas, which have often enhanced eutrophication processes and have favoured the appearance of toxic algae. All this resulted in an endangering of water quality making it not suitable for the projected purposes (Naselli-Flores, 1999). Such situation is particularly evident in the years characterised by low precipitation and a consequent diminished water income (Barone & Naselli-Flores, 1994; Naselli Flores & Barone, 1998) or in those ecosystems where exploitation of water resources (and thus WLF) is greater (Naselli-Flores, 2000). Actually, the exploitation or over-exploitation of water resources causes very wide water-level fluctuations which were observed to influence the ecosystem functioning both from the bottom (Naselli Flores & Barone, 2003) and from the top (Naselli Flores & Barone, 1997) of the food webs.

A general decrease in water quality has been observed in aquatic ecosystems, independently from their trophic state, during low-water periods (e.g. Arfi, 2003; Geraldes & Boavida, 2003; Kangur et al., 2003; Nõges et al., 2003). In addition, depending on the specific morphology of a water body, human-induced WLF may negatively interfere with the entire trophic structure of the ecosystem, from phytoplankton to fish, also including benthic communities (Gusakov, 2001) and macrophyte development (Chow-Fraser et al., 1998) endangering not only water quality but also fish resource exploitation. Nevertheless, as pointed out by Coops & Hosper (2002) and Roelke et al. (2003), WLF may be used as a management tool for freshwater ecosystems and represent a way to improve water quality.

In Sicily, a tendency towards drier periods from the seventies onward was underlined by Bonaccorso et al. (2003). This may further enhance the amplitude of WLF and thus it seems urgent, as pointed out by Zalewski (2002), to deeply understand the evolutionarily established resistance and resilience of the lake-catchment ecosystem to stressing factors, in order to use ecosystem properties as a management tool.

In order to investigate the influences that water-level fluctuations have on ecosystem functioning, the limnological features of a Sicilian reservoir were studied in 2001 and 2003. In this paper, the results of a comparison between the two periods, one characterised by low water and the second one by higher levels, is shown. This was possible thanks to a different management procedure applied in 2003, which was agreed with the management board of the reservoir, to compare the behaviour of this ecosystem under different hydraulic regimes, and to evaluate the viability of an improvement of the water quality through a more careful management of water-level fluctuations.

Materials and methods

Description of the studied system

The limnological investigation was carried out in Lake Arancio, a hypertrophic reservoir located 85 km south of Palermo and 5 km from Sicily's southern coast (Fig. 1). In order to accumulate as much water as possible, the reservoir exhibits a rather complicate feeding system: it receives water both from a "direct" catchment area of about 140 km^2, and from three different "linked" catchments (about 65 km^2), connected through pipelines, which irregularly bring water to the lake. A fourth income of water derives from the nearby River Belice which occasionally discharges unknown amounts of water into the lake, and a

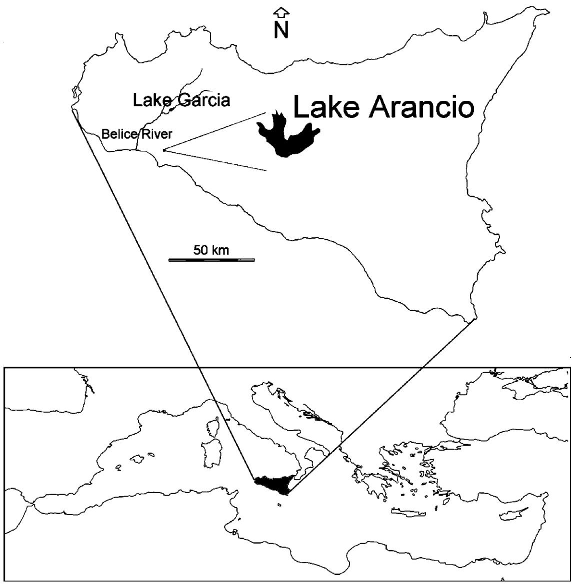

Figure 1. Location of the studied reservoir.

fifth is due to an "emergency" line which connects another reservoir, Lake Garcia, to Lake Arancio. Lake Garcia is a mesotrophic (phytoplankton biomass ranging between 0.4 and 2.7 mg l^{-1}), rather deep (43 m) reservoir, located about 30 km North Lake Arancio (Calvo et al., 1993). Its filling was completed in 1992 but the water stored has never been used because the distribution network has not been completed yet.

The characteristics of Lake Arancio, very rich as it is in nutrients, with hard, carbonate rich waters and relatively high sulphate concentrations, are typical of Sicilian reservoirs. The region is characterised by a semi-arid Mediterranean climate and water inflow to the reservoir is confined to the winter months. During summer there is no inflow to compensate the strong drawdown caused by evaporation and water usage to suffice requirements of agriculture. As a consequence, the water body experiences strong water-level fluctuations on an annual basis upon which a longer (11–12 years) mesoclimatic periodicity is superimposed (see Mosetti 1996; Padisák, 1998). This, due to typical climatic oscillations and variation in the amount of precipitation, causes the occurrence of high and low water level years. Table 1 lists some of the main morphological features of the water body whereas a detailed description of the reservoir's morphology is given in Barone & Naselli-Flores (1994). Data on phytoplankton and food web structure are reported in Naselli-Flores & Barone (1997, 1998, 2003).

In order to better understand the effects caused by WLF, the management board of the reservoir agreed to apply a different strategy in refilling the water body in 2003. Actually, instead of adding water from Lake Garcia in one step when the Lake Arancio's volume had reached the minimum holding they accepted to gradually re-fill the reservoir as the water was used. This different

Table 1. Main morphological features of the studied reservoir in the years of sampling compared to project ones

Parameter	Unit	Project value	May 2001	September 2001	May 2003	September 2003
Water level	m a.s.l.	180	175.3	164.1	176.6	170.5
Volume	10^6 m^3	32.0	21.5	2.8	25.1	11.3
Surface	km^2	3.2	2.5	0.9	2.8	1.8
Average depth	m	10.0	8.6	3.1	9.0	6.3

hydraulic input was decided and monitored to minimize thermocline disturbance in summer period, in the attempt to maintain the thermal stability of the reservoir.

Sampling and measurements

Samples were taken and measurements were done at a site located 250 m from the dam, in the middle of the reservoir. In this paper water samples collected weekly between April and November 2001 and 2003, were taken into consideration. In particular, samples for phytoplankton composition and biomass were collected from depths corresponding to 100%, 50% and 1% of the surface irradiance, as measured by a Li-Cor quantum sensor. These samples were mixed and a 100-ml subsample was immediately fixed with Lugol's solution for counting. Phytoplankton species were identified using the most updated phycological literature. Cell counting was performed using a Zeiss-Axiovert 100 inverted microscope, following the Utermöhl method. In several cases, it was necessary to dilute the sample up to 10 times to perform counting.

Microcystis density evaluations were simplified by the fact that colonies tended to disintegrate rather easily in Lugol's solution. Before placing the samples in the sedimentation tubes, they were squashed to increase colony disintegration. Cell numbers in the few colonies intact in spite of squashing were evaluated by counting their number in a reference volume and then extrapolating to the entire colony.

Biomass was estimated from biovolume, assuming unit specific mass density, by geometrical approximations according to Hillebrand et al. (1999). At least 200 specimens or colonies of the most abundant phytoplankton taxa were measured to calculate biovolume. This was calculated for each organism or colony and then the average biovolume was used as the final value.

Temperature profiles were measured with a multiparameter probe (Idromar IM5052) every 0.50 m, between 09:30 and 10:30 h, at the same time intervals. Mixing depth was estimated from temperature profiles ($\Delta T > 1$ °C m^{-1}) and considered equal to the epilimnetic zone when the reservoir showed stratification. In case of absence of stratification, mixing depth was taken equal to the depth of the sampling station.

Euphotic depth was estimated from irradiance profiles as 1% of subsurface value.

Water samples for main nutrients (Reactive phosphorus = RP, NO_3–N, NH_4–N) and total phosphorus (TP) were collected immediately below surface, at mid-depth and near the bottom. Water chemical analyses were performed according to Tartari & Mosello (1997).

An attempt to evaluate the amount of reactive phosphorus flushed out the system was performed according to the formula:

$$(V_t - V_{t-1}) * [RP_{H'}] \qquad [10^3\text{g}] \tag{1}$$

where $(V_t - V_{t-1})$ represents the amount of water spilled out weekly and $[RP_{H'}]$ is the averaged concentration of hypolimnetic reactive phosphorus in two consecutive weeks.

Reactive phosphorus concentration in the hypolimnion was evaluated according to the formula:

$$V_H * [RP_H] - (1) \qquad [10^3\text{g}] \tag{2}$$

where V_H is the estimated volume of the hypolimnion at a given date and $[RP_H]$ is the corresponding hypolimnetic concentration of reactive phosphorus.

Reactive phosphorus concentration in the epilimnion was evaluated according to the formula:

$$V_E * [RP_E] \qquad [10^3\text{g}] \tag{3}$$

where V_E is the estimated volume of the epilimnion at a given date and $[RP_E]$ is the corresponding epilimnetic concentration of reactive phosphorus.

The volumes of epilimnetic and hypolimnetic part of the water body were calculated according to the surface-volume diagram and the corresponding centimetric tables updated to 1996, which were supplied by the management board of the reservoir as well as the hydrological data set.

Relative water column stability (RWCS) was calculated, according to Padisák et al. (2003a), by comparing the density difference between bottom (D_b) and surface (D_s) water to the density difference between 4 °C (D_4) and 5 °C (D_5) of pure water, using the formula:

$$\mathrm{RWCS} = \frac{D_b - D_s}{D_4 - D_5}$$

Water density was calculated from temperature values using a Water Density Calculator, available on the Internet, which calculates water density at a given temperature between −8 and 108 °C using 5-point Lagrange interpolation (Senese, 2003).

Results

The physical scenario

The reservoir, according to Mediterranean climate features, is characterised by wide water-level fluctuations. The amplitude of these fluctuations varies from year to year depending on the amount of water stored in winter, when precipitation occurs (Fig. 2). The amount of precipitation in the area does not directly influence the annual water-storing due to the different sources which feed the lake and the unpredictable management procedures applied. In the studied years the water level of Lake Arancio decreased by 11 m in 2001 and by 6 m in 2003 (Fig. 3a, b). In 2001, the depth at the sampling station varied between a maximum of 15 m at the beginning of July, to a minimum of 4 m at the beginning of October. The year 2003 was marked both by a slightly larger water income and by the different management of the income itself. The depth varied less, between 16 m in mid-June and 10 m at the beginning of September.

The stored volumes fluctuated between 21.5 and 2.8×10^6 m^3 in 2001 (Fig. 3a). The maximum holding was recorded at the end of April 2001 and until the end of June the level decreased slowly mainly because of evaporation losses and due to the checking procedures to control the regular opening of the outlets. Since July, starting the irrigation period, the water losses increased constantly and an average value of 1.25×10^6 m^3 per week was utilised until the end of September. The average weekly reduction of the water column depth amounted to 0.86 m. On the 6th of October, to meet agriculture demands in the district, the management board started re-filling the reservoir with water from Lake Garcia. In total, 13×10^6 m^3 of hypolimnetic water from Lake Garcia were added to Lake Arancio hypolimnetic waters in 6 weeks, from October 6 to November 10. Of these, about 7×10^6 m^3 were used to continue irrigating and 6×10^6 m^3 increased the reservoir volume to 8.9×10^6 m^3 and the depth at the sampling station passed from 4 to 8 m. This value remained more or less stable throughout all winter since no precipitation occurred during this period of prolonged drought, striking the island, as well as other southern Mediterranean zones, in the years 2000–2001. The more rainy 2002 and 2003 allowed to store more water and in 2003 (Fig. 3b) the volume ranged between 25.1×10^6 m^3 (at beginning of May) and 11.3×10^6 m^3 (at the beginning of September). As in 2001, about 12×10^6 m^3 were added from lake Garcia in 2003 but, conversely to 2001, in this year they were added continuously, and with increasing amounts from the end of August, since the beginning of the irrigating season. Moreover, this last spanned over

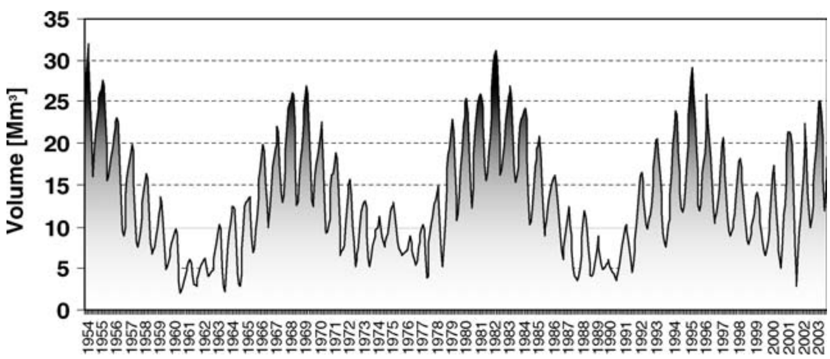

Figure 2. Storage volume changes in Lake Arancio since its building.

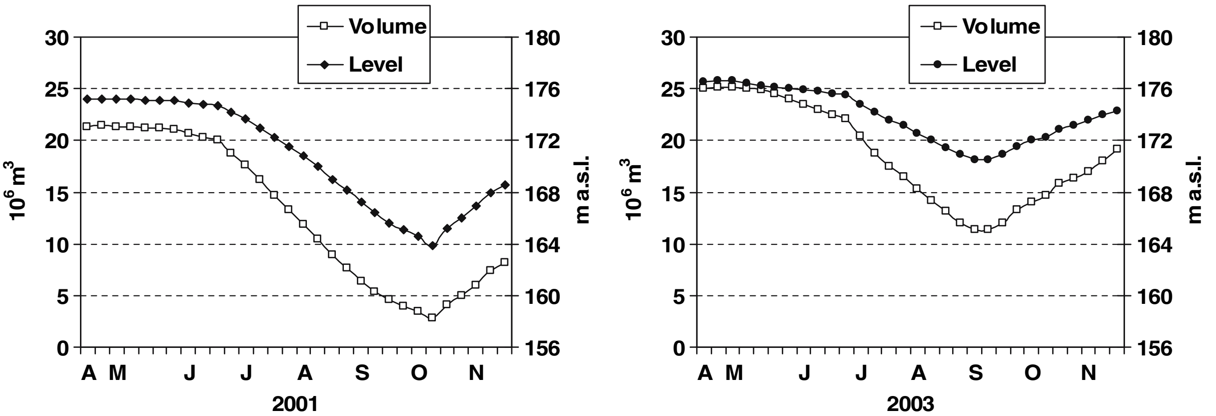

Figure 3. Water-level and volume fluctuations in the two studied years.

a shorter period due the higher precipitation occurred in spring and fall. In addition, the different management procedure ensured the same amount of water for irrigation to the district in both years (about 25×10^6 m^3) but contributed to maintain the levels of Lake Arancio in 2003 higher than those recorded in the former years.

The different exploitation of water resources in the two studied years is reflected in the thermal profiles (Figs. 4 and 5). The 2001 water level drawdown disrupted the thermal stability of the water body causing the elimination of the thermocline in mid-summer 2001, whereas in 2003 the reservoir exhibited a stable stratification till the end of October. From a thermal point of view, Lake Arancio is a potentially monomictic water body. The onset of stratification generally occurs at the end of March and the pycnocline in the reservoir is found between 4.5 and 6 m of depth. In 2001, at the beginning of July an increasing tendency to mixing could be observed in the water column due to the decrease of depth. Accordingly, stability of the water column stratification reached its maximum of 220 at the end of June 2001 (Fig. 6). With the onset of irrigation, stability rapidly decreased and the water column of the reservoir experienced a two-week period of isothermy in mid-September. A second peak in stability was observed in mid-October. This occurred just after the starting of the water income from Lake Garcia, when Lake Arancio doubled its volume and its water level increased by more than 2 m in a few days. Winter circulation started in mid-November. In 2003, the higher levels guaranteed relative stability of the water column to reach higher values, with a maximum above 370 in mid-August. Moreover, it never went below 70 in the studied period and the lake experienced a regular development and disruption of its thermocline.

With regard to underwater light climate, in 2001, the ratio between mixing depth and euphotic depth was above 4 for most of the summer period, reaching a peak above 12 at the beginning of July (Fig. 7). In 2003, sensibly lower values were recorded, thus stressing a much higher availability of light in the epilimnetic layers.

The chemical scenario

Dissolved oxygen and pH values generally follow the temperature trends in the water column. Higher pH values, ranging between 8.1 and 8.7 in 2001 and between 8.0 and 8.4 in 2003, and percentage oxygen saturation above 100% were commonly recorded in the epilimnetic layers. Conversely, below the pycnocline, a rapid decrease of these values was recorded.

The absence of oxygen in the hypolimnion, accompanied by pH values below 7.6, lasted as far as the RWCS was higher than 30 in 2001; these values rose up during September 2001 circulation, getting equal to epilimnetic ones, and dropped again to a low in mid-October when stratification was re-established (Naselli-Flores, 2003). Conversely, 2003 was characterised by a permanent hypolimnetic anoxia and by hypolimnetic pH values below 7.3.

In accordance with its high trophic state, the reservoir exhibits high nutrient and total phos-

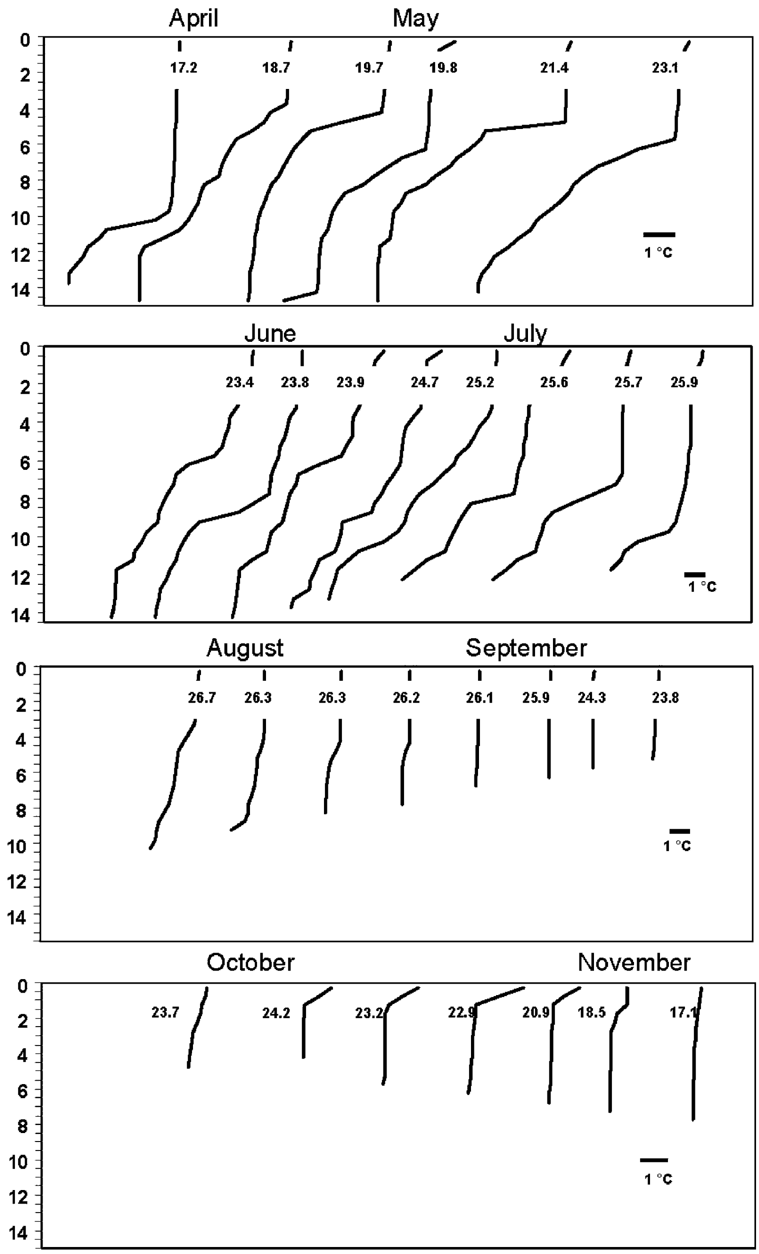

Figure 4. Temperature profiles recorded in lake Arancio in 2001. Inserted numbers denote water temperatures at 10 cm.

phorus concentrations (Fig. 8). In particular, TP superficial values were always above 200 $\mu g\ l^{-1}$ in 2001. In 2003, lower values, ranging between 20 and 150 $\mu g\ l^{-1}$ were recorded in the epilimnion. Hypolimnetic layers were in both periods characterised by quite high TP values (420–1600 $\mu g\ l^{-1}$), 3–4 times (2001) to 40 times (2003) higher than those recorded in the surface layer.

Reactive phosphorus and ammonium concentrations followed an analogous trend with bottom values generally higher than superficial ones. Nevertheless, 2003 epilimnetic values ($4 < RP < 9\ \mu g\ l^{-1}$; $0 < N\text{–}NH_4 < 42\ \mu g\ l^{-1}$) were sensibly lower than those recorded in 2001 ($12 < RP < 45\ \mu g\ l^{-1}$; $24 < N\text{–}NH_4 < 145\ \mu g\ l^{-1}$).

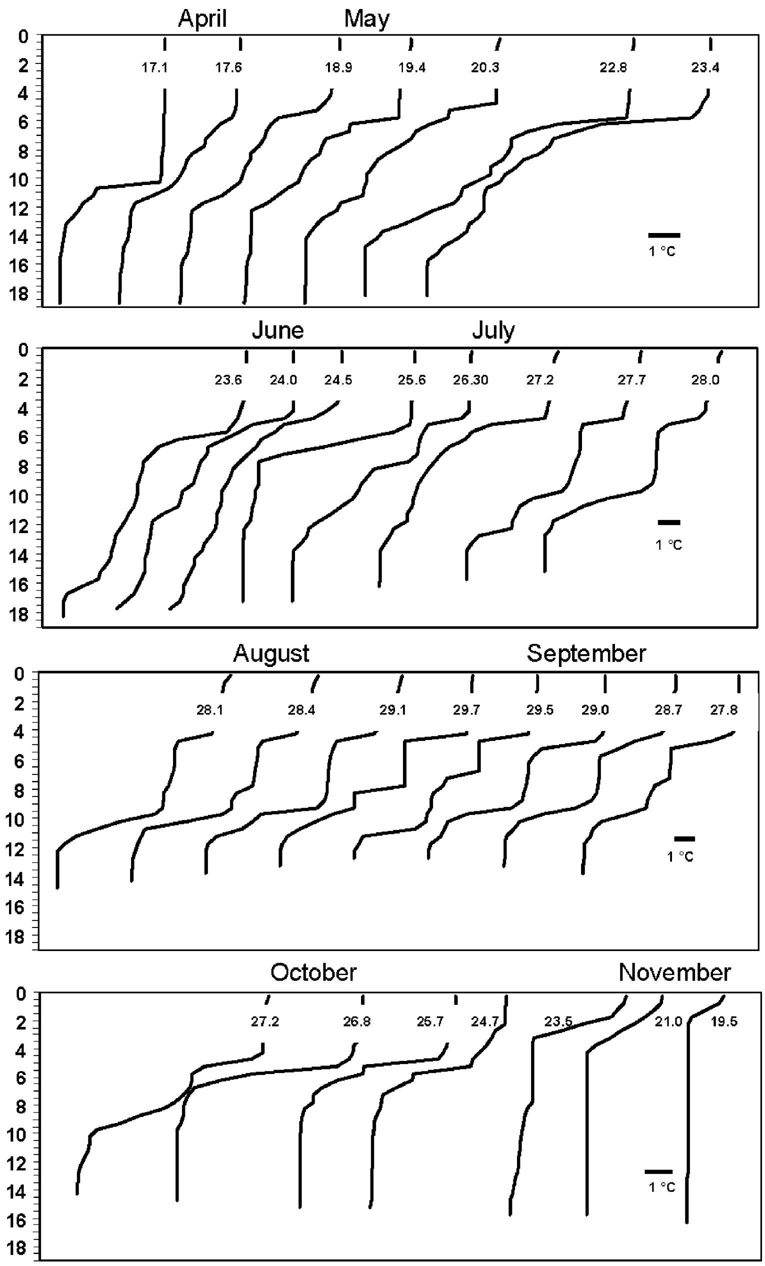

Figure 5. Temperature profiles recorded in lake Arancio in 2003. Inserted numbers denote water temperatures at 10 cm.

Nitrate showed its highest values in April of both years. Afterwards, the values constantly decreased. Even in this case, 2003 was marked by lower concentrations compared to 2001.

Figure 9 shows that in 2001 almost the totality of hypolimnetic reactive phosphorus was flushed out the system or, since the lack of water income from the catchment in this season, transferred to

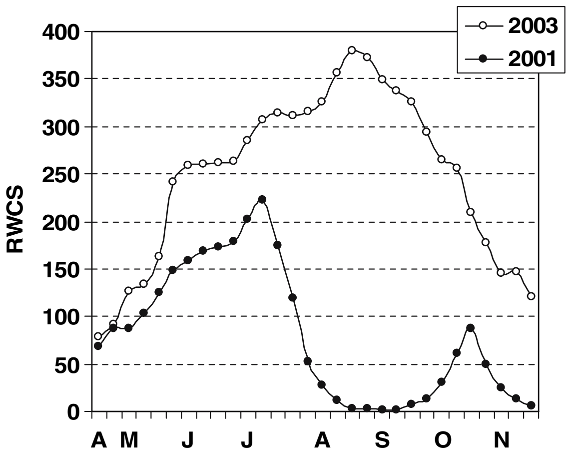

Figure 6. Relative water column stability (RWCS) in Lake Arancio in the two study years.

epilimnetic layers. Conversely, in 2003, the hypolimnetic reserve of reactive phosphorus always show values higher than the epilimnetic one suggesting a stronger segregation of these two compartments in summer.

Phytoplankton dynamics

The lack of nutrient limitation is well depicted by the phytoplankton biomass values (Fig. 10), generally above 25 mg l^{-1}, which reached values above 10^3 mg l^{-1} in late summer and autumn 2001. In 2003, a comparable trend occurred but

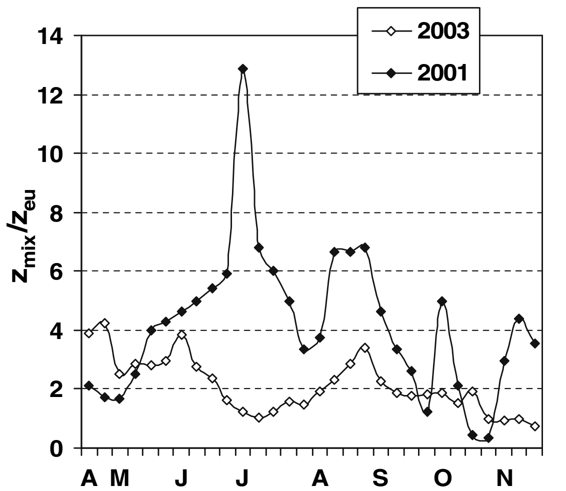

Figure 7. Changes in the mixing depth/euphotic depth ratio in Lake Arancio in the two study years.

lower biomass values were attained and the lack of October peak was observed. In addition, the increasing trend of biomass, which in 2001 started at the beginning of June and lasted in a peak in mid-July, in 2003 was a bit delayed and less intense.

Phytoplankton dynamics in 2001, as shown on the left side of Figure 10, was discussed in detail in Naselli-Flores & Barone (2003). It can be important to stress that in 2001, very large amounts of *Microcystis* spp. dominated phytoplankton assemblage since the beginning of May to mid-October with percentage values ranging between 75 and 100%.

Phytoplankton composition in 2003 shows a more diversified structure (Fig. 10, right side) with *Microcystis aeruginosa* reached, at maximum, only a 37% share of total biomass in the second half of August and absolute values did not exceed 40 mg l^{-1}. *Ceratium furcoides*, *C. hirundinella*, and several species of *Pediastrum* (*P. simplex*, *P. duplex*, *P. boryanum*, *P. biwae* and many of their forms) dominated the assemblage in spring and early summer, accompanied by the colony forming *Chlamydomonas passiva*. Mid-summer was characterised by the co-dominance of *Microcystis* spp. and *Pediastrum* spp., the latter being the most represented in September samples, accompanied by *Coelastrum microporum* and *C. pseudomicroporum*. Since the beginning of October, *Closterium aciculare* became more and more important, forming more than 75% of phytoplankton biomass at the end of the investigated period. According to the functional classification of phytoplankton proposed by Reynolds et al. (2002), a sequence $\mathbf{L_M} \rightarrow \mathbf{J}$ (with **M** representatives) $\rightarrow$ **P** is recognizable in 2003 whereas in 2001 a rather permanent presence of representatives of the **M** association was recorded in the studied period. This is in accordance with the greater light availability and the more stable stratification which occurred in the reservoir in 2003. Actually, $\mathbf{L_M}$ is a codon whose main representatives are *Ceratium* spp. and *Microcystis* spp., generally typical of summer epilimnia in eutrophic lakes. **J** and **M** coda, the first one represented by low-light sensitive species as *Pediastrum* spp. and *Coelastrum* spp., and the second one by *Microcystis* spp., are typical in shallow, nutrient rich environments. **P** is a codon collecting diatoms as *Fragilaria crotonensis*

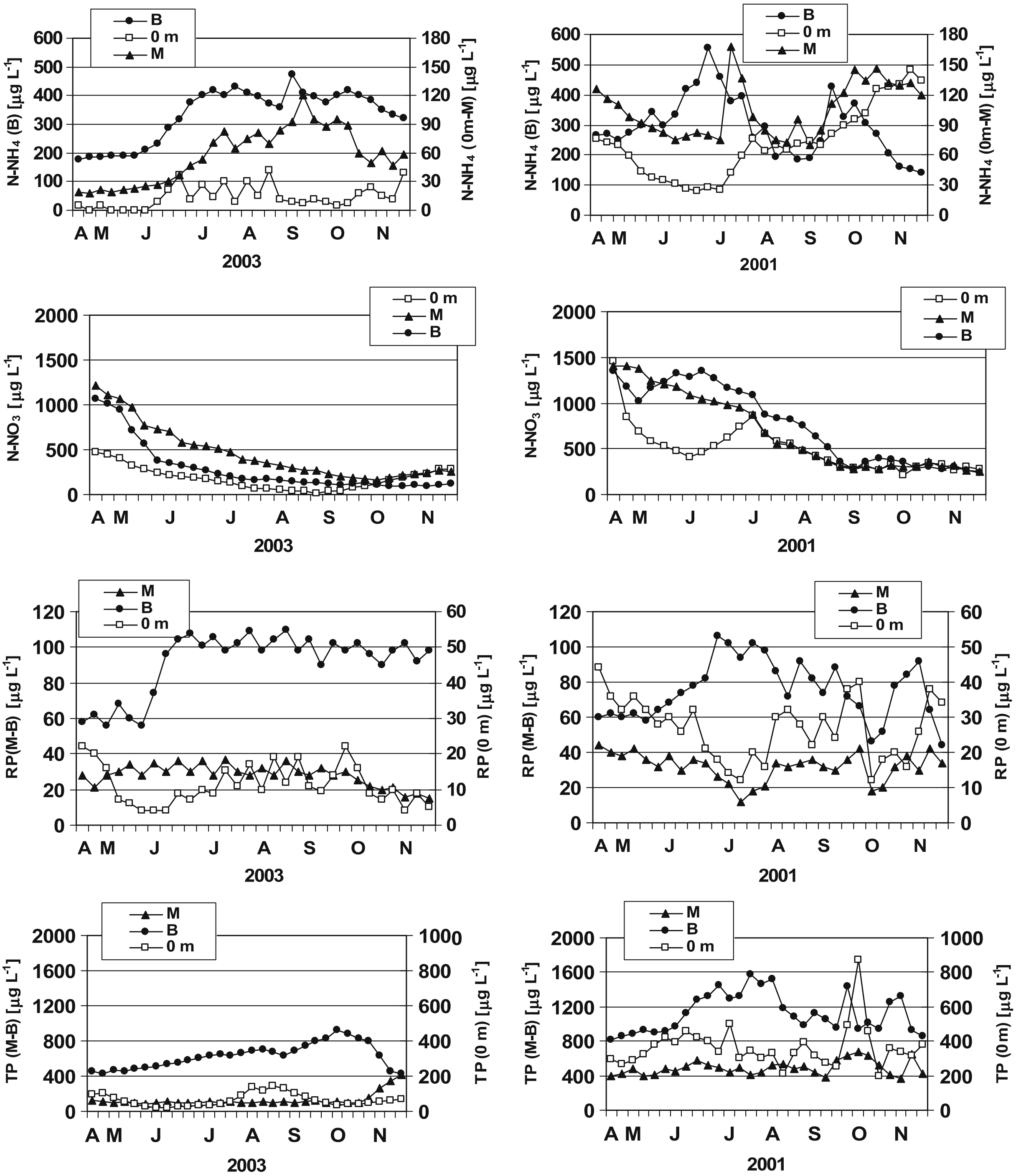

Figure 8. Inorganic nitrogen, reactive phosphorus (RP) and total phosphorus (TP) dynamics in Lake Arancio in the two study years. 0 m = values immediately below surface, B = near the bottom values, M = mid-depth values.

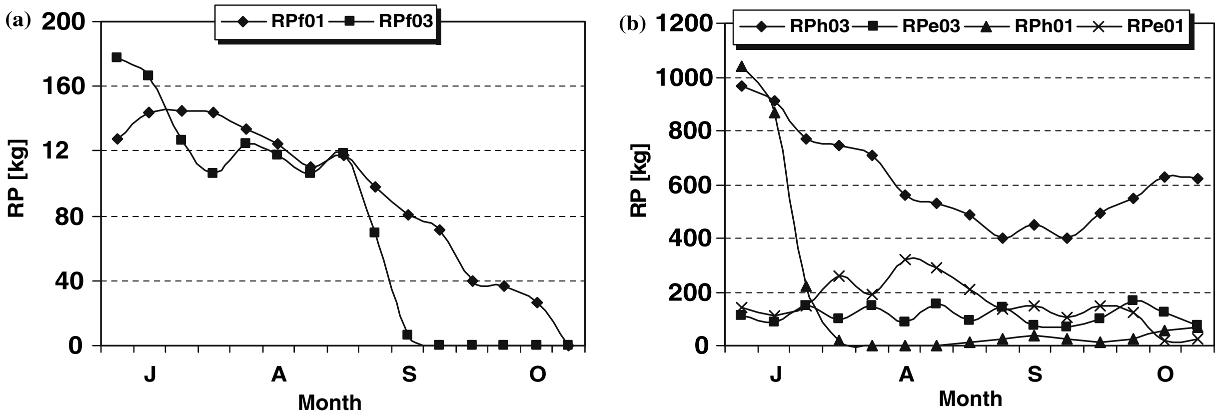

Figure 9. Estimated amount of reactive phosphorus flushed out the system (a) and estimated hypolimnetic and epilimnetic reactive phosphorus amounts (b) in the studied periods. RPf01 = reactive phosphorus flushed in 2001; RPf03 = reactive phosphorus flushed in 2003; RPe01 = epilimnetic reactive phosphorus in 2001; RPe03 = epilimnetic reactive phosphorus in 2003; RPh01 = hypolimnetic reactive phosphorus in 2001; RPh03 = hypolimnetic reactive phosphorus in 2003.

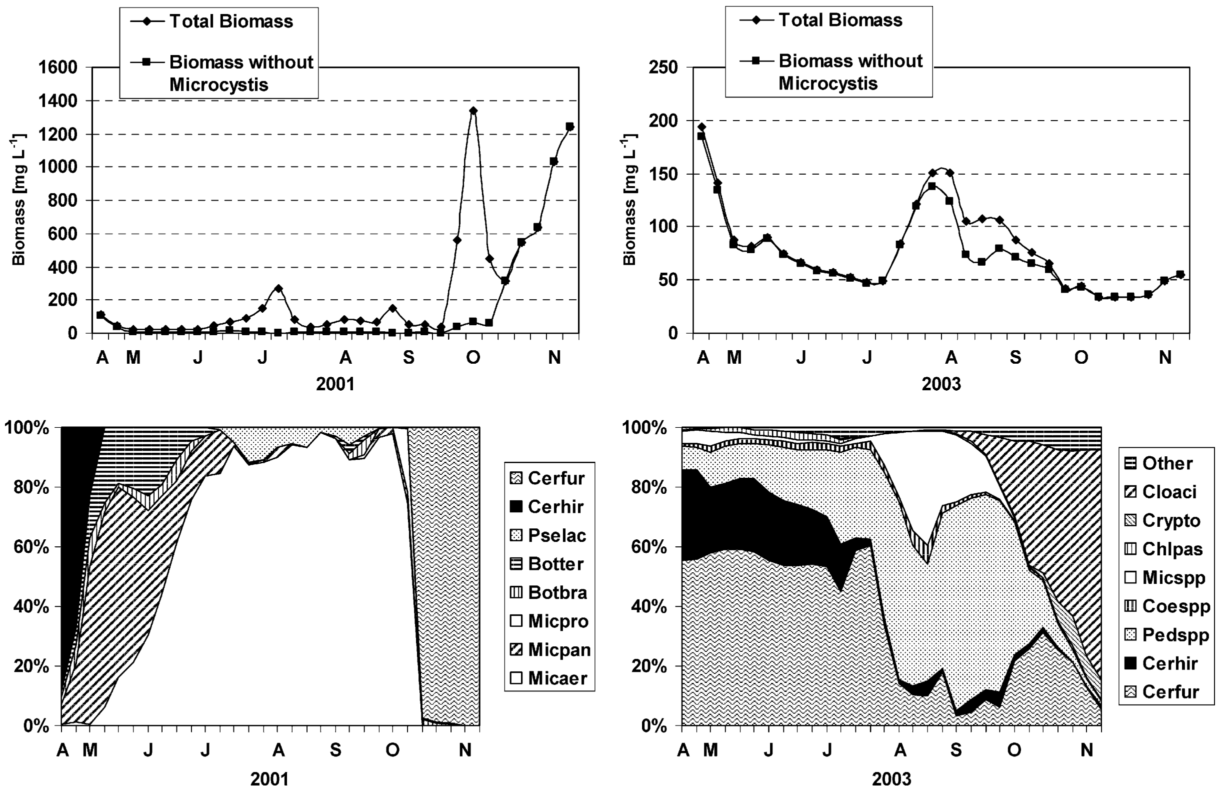

Figure 10. Above: Seasonal development of phytoplankton biomass in the two study years. Below: Relative composition of phytoplankton biomass in the same periods. Cerfur: *Ceratium furcoides*; Cerhir: *Ceratium hirundinella*; Pselac: *Pseudosphaerocystis lacustris*; Botter: *Botryococcus terribilis*; Botbra: *B. brauni*; Micpro: *Microcystis protocystis*; Micpan: *Microcystis panniformis*; Micaer: *Microcystis aeruginosa*; Micspp: *Microcystis* spp.; Cloaci: *Closterium aciculare*; Crypto: Cryptomonad flagellates; Chlpas: *Chlamydomonas passiva*; Coespp: *Coelastrum* spp0. Pedspp: *Pediastrum* spp.

and *Aulacoseira granulata* in temperate lakes but mainly represented by *Closterium aciculare* or by other low form resistance desmids in warmer climates (Padisák et al., 2003b).

Discussion

In Sicilian reservoirs, the dewatering, which corresponds with the warm period, often lead to an anticipate breaking of the thermocline. In normal conditions, a sufficiently deep Mediterranean lake would start developing its thermocline at the beginning of March, being stratified till the end of November. Nevertheless, depending on the strength of the wind blowing on their surface and on their specific morphometry, even shallow, usually not-stratifying, water bodies may show an atelomictic (daily stratification) behaviour during summer period, when wind is not strong enough to make them circulating (Naselli-Flores, 2003).

Moreover, in Mediterranean aquatic ecosystems, nutrient income from the catchment mainly occurs in winter and spring, whereas in summer, when precipitation are scarce or absent, it mainly occurs from internal loading, when re-circulation events occur. Since the capacity to retain phosphorus is correlated to residence time, these environments, which in winter receive the nutrients not having any water outflow (residence time tends to infinity), act as a phosphorus sink (Straškraba et al., 1995). Their internal loading may constantly increase if a careless management of the catchment does not provide an adequate treatment of the inflowing waters. Sicilian reservoirs, for instance, lacking any treatment of inflowing waters, likely experienced a constant increase in their internal loading since their building.

The analysis of the thermal profiles indicates that summer dewatering, although causing a depth decrease, leaves unchanged the thickness of the epilimnion by dragging down the thermocline. This promotes a "migration" of anoxic zones of reservoir bottom into the circulating part of the water body. In 2001, 14 hectares per week of reservoir bottom were re-exposed to circulation and released their nutrient content to the upper layers (Naselli-Flores, 2003). The same event took place in 2003 as underlined by the trend of phytoplankton biomass, which increases, both in summer 2001 and 2003, likely thank to the hypolimnetic and sedimentary nutrient reserve which, week by week, support phytoplankton growth. In 2003 dewatering was partly compensated by a regular water income from Lake Garcia. Thus, a minor area of reservoir bottom was influenced by circulation, stratification persisted all along the summer, and hypolimnetic nutrient reserve was more confined and less available to permit the phytoplankton biomass increase, as conversely occurred in 2001.

As a consequence of the operational procedures to which reservoirs undergo, which allows the breaking of the thermal stratification and the release of nutrient-rich hypolimnetic waters into the upper layers, phytoplankton biomass is sustained throughout the summer. In 2003, Lake Arancio, due to the different re-filling strategy adopted, maintained its stratification throughout all summer. Thus, a feedback regulation of phytoplankton biomass, mediated by epilimnetic nutrient depletion, was observed. In this case, nutrient income from the hypolimnion likely occurred in autumn when lower temperature values did not suffice growth requirements of cyanoprokaryotes. An analogous pattern can be observed by comparing phytoplankton results in the low water period 1990/1991 (Barone & Naselli-Flores, 1994) with those achieved in the high water level 1993 (Naselli-Flores & Barone, 1998).

In addition, a further consequence of the anticipate breaking of the thermocline is represented by the increase of the mixing depth/euphotic depth ratio which contribute to make optically deep those environments lying in the upper part of the trophic range (Naselli-Flores, 1998). This event may cause the success of those phytoplanktonic organisms adapted to low light contents or able to regulate their buoyancy as cyanoprokaryotes are. These organisms, in 2001, constituted almost the totality of phytoplankton biomass in summer months (Naselli-Flores, 2000). In 2003, the mixing depth/euphotic depth ratio showed notably lower values, making the environment more suitable to the development of members of the association **J** which are more sensitive to settling into low light (Reynolds et al., 2002).

In ecological sciences, resilience is the ability of ecosystems to regain their structure and function after an external physical force has impaired them (Reynolds, 2002). Mediterranean reservoirs, when subject to extreme water-level fluctuations, experience a dramatic alteration of their thermal stratification patterns which may transform themselves from deep monomictic water bodies to amictic, polimictic or atelomictic environments (Naselli-Flores, 2003). Such dualistic changes in ecosystemic identity bring them outside their homeostatic plateau, thus preventing resilience to act, simply because the nature of the ecosystem itself has been transformed from a well established lake typology to another (see Padisák & Reynolds, 2003). The "new" environmental conditions select a biota among the available pool of organisms which are present at the moment of transformation and the ecosystem sets new thresholds of its properties (e.g. a new homeostatic plateau and new resilience and resistance thresholds). Thus, the organisms which, among the others, best fit these altered conditions become dominant.

Moreover, phytoplankton biomass can grow in a closed system to a point which is established by the availability of resources, mainly light and nutrients. Epilimnia of Sicilian reservoirs, which in summer do not receive any water income could be regarded as closed systems. Without new energy inputs a feedback regulation of phytoplankton biomass should take place and a considerable reduction in their productivity should be recorded. Conversely, WLF allow a regular energy input which, according to traditional ecological theories, make these environments tending towards a steady state of maximum attainable biomass. The deepening of the mixing zone at the breaking of the thermocline contributes to a decreased availability of light and favours those organisms which are less entrained in water circulation, as buoyant cyanoprokaryotes.

Cyanoprokaryotes, and in particular *Microcystis* spp., demonstrated to be very well adapted to the above described conditions. These organisms, forming in Sicilian reservoirs very dense bloom, were also found to be toxic (Via-Ordorika et al., 2004). Thus, a recovery plan of water quality is particularly urgent.

From the analysis of the results achieved in the present study the negative effects produced by very wide WLF could be counteracted by the resilience of the ecosystem itself. Thus, it seems very important, in order to guarantee water quality and to limit the appearance of blue-green algae, to prevent such transformation from a monomictic ecosystem to a polymictic one.

Concluding remarks

Former investigations carried out in Lake Arancio have shown very high diversity values likely sustained by WLF which acted as an intermediate disturbance for phytoplankton assemblages (Naselli-Flores & Barone, 1998). Actually, neither all eutrophic ecosystems are characterised by low diversity values nor show the permanent dominance of toxin-producing cyanoprokaryotes.

Most of the Sicilian reservoirs belong to the upper part of the trophic range: the first management step should be addressed towards supplying urban centres with adequate waste water treatment plants and a rational use of fertilizers in agriculture. In the meantime, a water quality increase could be obtained through a more careful management of water distribution.

To achieve this task, two ways are possible: to stop water drawdown above a threshold which avoids the anticipate disruption of the thermocline or, according to ecohydrological principles (Zalewski, 2002), to manage the hydrology of reservoirs in order to increase the capacity of these ecosystem to absorb human-induced impacts. The first choice is rather drastic and can be in conflict with the social and economic needs of those regions which, for climatic reasons, strongly depend on the necessity to exploit the water stored in the reservoirs. The latter is more difficult and needs a careful and more global territorial planning. In particular, eutrophication problems due to the inadequate control of agricultural run-off and to the lacking of urban waste water treatment plants can and must be solved. In the meantime, and even after this reduction will be achieved, the threshold can be set for every reservoir according to its specific morphology, and should be carefully considered in the management plans of these peculiar ecosystems as well as in the building plans of a new dam. In particular, in Lake Arancio by keeping the RWCS above a value of 50, an improvement of water quality was observed. By the simple measurements of temperature profiles and the calculation of a synthetic index, management boards can have a tool to understand the limit of dewatering to which an ecosystem may be exposed still avoiding its exit from the homeostatic plateau.

References

Arfi, R., 2003. The effects of climate and hydrology on the trophic status of Sélingué Reservoir, Mali, West Africa. Lakes & Reservoirs: Research and Management 8: 247 257.

Barone, R. & L. Naselli-Flores, 1994. Phytoplankton dynamics in a shallow, hypertrophic reservoir (Lake Arancio, Sicily). Hydrobiologia 289: 199–214.

Bonaccorso, B., I. Bordi, A. Cancelliere, G. Rossi & A. Sutera, 2003. Spatial variability of drought: an analysis of the SPI in Sicily. Water Resources Management 17: 273–296.

Calvo, S., R. Barone, L. Naselli-Flores, C. Fradà Orestano, G. Dongarrà, A. Lugaro & G. Genchi, 1993. Limnological studies on lakes and reservoirs of Sicily. Il Naturalista Siciliano 1–292.

Chow-Fraser, P., V. Lougheed, V. Le, B. Thiec Crosbie, L. Simser & J. Lord, 1998. Long-term response of the biotic community to fluctuating water-level and changes in water quality in Cootes Paradise Marsh, a degraded costal wetland of Lake Ontario. Wetlands Ecology and Management 6: 19–42.

Coops, H. & S. H. Hosper, 2002. Water-level management as a tool for the restoration of shallow lakes in the Netherlands. Lake and Reservoir Management 18: 292–297.

Geraldes, A. M. & M. J. Boavida, 2003. Distinct age and landscape influence on two reservoirs under the same climate. Hydrobiologia 504: 277–288.

Gusakov, V. A., 2001. The effect of the hydrological regime in the Rybinsk Reservoir on the distribution and dynamics of benthic *Cyclops*. Water Resources 28: 94–104.

Hillebrand, H., C.-D. Dûrselen, D. Kirschtel, U. Pollingher & T. Zohary, 1999. Biovolume calculation for pelagic and benthic microalgae. Journal of Phycology 35: 403–424.

Kangur, K., T. Möls, A. Milius & R. Laugaste, 2003. Phytoplankton response to changed nutrient level in Lake Peipsi (Estonia) in 1992–2001. Hydrobiologia 506/509: 265–272.

Mosetti, F., 1996. Sulle regolarità di alcune oscillazioni climatiche (On the regularity of some climatic fluctuations, in Italian). Atti 5° Workshop "Clima Ambiente e Territorio nel Mezzogiorno", C.N.R., Roma: 259–271.

Naselli-Flores, L., 1998. Phytoplankton assemblages in reservoirs: is it chemical or physical constraints which regulate their structure?. International Revue of Hydrobiology 83: 351–360.

Naselli-Flores, L., 1999. Limnological aspects of Sicilian reservoirs: a comparative ecosystemic approach. In Tundisi J. G. & M. Straškraba (eds.), Theoretical Reservoir Ecology and its Applications. Backhuys Publishers, Leiden: 283–311.

Naselli-Flores, L., 2000. Phytoplankton assemblages in twenty-one Sicilian reservoirs: relationships between species composition and environmental factors. Hydrobiologia 424: 1–11.

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

Naselli-Flores, L. & R. Barone, 1997. Importance of water-level fluctuation on population dynamics of cladocera in a hypertrophic reservoir (Lake Arancio, south-west Sicily, Italy). Hydrobiologia 360: 223–232.

Naselli-Flores, L. & R. Barone, 1998. Phytoplankton dynamics in two reservoirs with different trophic state (Lake Rosamarina and Lake Arancio, Sicily, Italy). Hydrobiologia 369/370: 163–178.

Naselli-Flores, L. & R. Barone, 2003. Steady-state assemblages in a Mediterranean hypertrophic reservoir. The role of Microcystis ecomorphological variability in maintaining an apparent equilibrium. Hydrobiologia 502: 133–143.

Nõges, T., P. Nõges & R. Laugaste, 2003. Water level as the mediator between climate change and phytoplankton composition in a large shallow temperate lake. Hydrobiologia 506/509: 257–263.

Padisák, J., 1998. Sudden and gradual responses of phytoplankton to global climate change: case studies from two large, shallow lakes (Balaton, Hungary and the Neusiedlersee Austria/Hungary). In George D. G., J. G. Jones P. Puncochar C. S. Reynolds & D. W. Sutcliffe (eds.), Management of Lakes and Reservoirs during Global Change. Kluwer Academic Publishers, Dordrecht: 111–125.

Padisák, J. & C. S. Reynolds, 2003. Shallow lakes: the absolute, the relative, the functional and the pragmatic. Hydrobiologia 506/509: 1–12.

Padisák, J., F. A. R. Barbosa, R. Koschel & L. Krienitz, 2003a. Deep layer cyanoprokaryota maxima are constitutional features of lakes: examples from temperate and tropical regions. Archiv für Hydrobiologie/Advances in Limnology 58: 175–199.

Padisák, J., É. Soróczki-Pintér & Z. Rezner, 2003b. Sinking properties of some phytoplankton shapes and the relation of form resistance to morphological diversity of plankton – an experimental study. Hydrobiologia 500: 243–257.

Reynolds, C. S., 2002. Resilience in aquatic ecosystems – hysteresis, homeostasis, and health. Aquatic Ecosystem Health Management 5: 3–17.

Reynolds, C. S., V. L. Huszar, C. Kruk, L. Naselli-Flores & S. Melo, 2002. Towards a functional classification of the freshwater phytoplankton. Journal of Plankton Research 24: 417–428.

Roelke, D., S. Augustine & Y. Buyukates, 2003. Directing the fall of Darwin's "grain in the balance": manipulation of hydraulic flushing as a potential control of phytoplankton dynamics. Texas Water Resources Institute, Technical Report 245: 14 pp.

Senese, F. A., 2003. http://antoine.frostburg.edu/chem/senese/javascript/water-density.html.

Straškraba, M., I. Dostálková, J. Hejzlar & V. Vyhnálek, 1995. The effect of reservoirs on phosphorus concentration. Internat. Rev. Ges. Hydrobiol. 80: 403–413.

Tartari, G. A. & R. Mosello, 1997. Metodologie analitiche e controlli di qualità nel laboratorio chimico dell'Istituto Italiano di Idrobiologia. Documenta dell'Istituto Italiano di Idrobiologia 60: 160 pp.

Via-Ordorika, L., J. Fastner, R. Kurmayer, M. Hisbergues, E. Dittmann, J. Komarek, M. Erhard & I. Chorus, 2004. Distribution of microcystin-producing and non-microcystin-producing *Microcystis* sp. in European freshwater bodies: Detection of microcystins and microcystin genes in individual colonies. Systematic and Applied Microbiology 27: 592–602.

Williams, D. D., 1987. The ecology of temporary waters. Timber Press, Portland, 205 pp.

Zalewski, M., 2002. Guidelines for the integrated management of the watershed. Phytotechnology and Ecohydrology. UNEP, DTIE, IETC, Freshwater Management Series 5, 188 pp.