

# The effects of an artificial wetland dominated by free-floating plants on the restoration of a subtropical, hypertrophic lake

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## Abstract

Lake Rodó (Montevideo, Uruguay) is a small, urban, hypertrophic lake undergoing restoration. In this study, we evaluated the nutrient removal efficiency and water quality improvement attributable to a water recirculation system, consisting of the lake and three connected pools converted to artificial wetlands dominated by free-floating hydrophytes. *Eichhornia crassipes* and *Spirodela intermedia* dominated the hydrophyte community during summer and winter, respectively, with the biomass production being maintained throughout the year. The maximum production values of *E. crassipes* were 11.3 and 5.6 g DW m<sup>-2</sup> d<sup>-1</sup> in the summers of 1998 and 2000, respectively, while those of *S. intermedia* were 2.7 and 0.8 g DW m<sup>-2</sup> d<sup>-1</sup> in the summers of 1999 and 2000, respectively. The aquatic plant community reduced the concentration of nutrients in the water column but did not significantly affect the sediment concentrations. Harvesting the hydrophytes removed the equivalent of 58–88% and 39–78% of the nitrogen (N) and phosphorus (P) load associated with the water column, respectively. In contrast, the harvests accounted for only 1–2% of the N and P load associated with the sediments. In the pools, the combination of water recirculation and hydrophytes generally diminished the algal biomass and the associated N and P, compared to that observed for the lake. The combined use of adequate aquatic plant harvests and hydraulic management increased the efficiency of the system and, therefore, seems to be a useful tool for restoring small, shallow lakes in tropical and subtropical regions.

## Key words

algal removal, aquatic plants, eutrophication, nutrient reduction, restoration, shallow lake.

## INTRODUCTION

The hypothesis of alternative stable states (Scheffer 1990; Scheffer *et al.* 1993) suggests that shallow lakes can have two states within a wide range of nutrient concentrations: a clear-water state dominated by aquatic plants, with low algal biomass and high transparency, and a turbid state dominated by phytoplankton. The reduction of the internal nutrient load might promote the shift from the turbid state to the clear-water state (Moss 1990; Moss *et al.* 1996; Jeppesen 1998; Scheffer 1998). However, the restoration of eutrophic, shallow lakes only seems feasible if biomanipulation and nutrient reduction are implemented together (Kasprzak 1995).

In temperate zones, the establishment and growth of submerged aquatic plants play a key role in maintaining a low algal biomass as a result of nutrient competition, light reduction and other complex physical, chemical and biological mechanisms (Scheffer 1998; Søndergaard & Moss 1998). However, harvesting macrophytes might constitute a significant in-lake measure for reducing the internal nutrient load (Carpenter & Adams 1977). Tropical and subtropical shallow lakes also can support a significant biomass of free-floating plants (e.g. *Pistia stratiotes* L., *Eichhornia crassipes* (Mart.) Solms). Their biomass and water surface cover increase with increasing eutrophication, with their harvesting generally being required to prevent interference with other uses of the ecosystems. In addition to these negative impacts, the free-floating plants with copious root systems also act as a filter, retaining the particulate matter suspended in the water column (Poi de

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Neiff *et al.* 1994; Maine *et al.* 1999; Kim & Kim 2000), thereby also limiting the availability of light and nutrients for phytoplankton. These plants have been widely used in artificial sewage treatments for phosphorus (P) and nitrogen (N) removal (Záková *et al.* 1994) and, more recently, to treat cellulose effluents (Dellarossa *et al.* 2001).

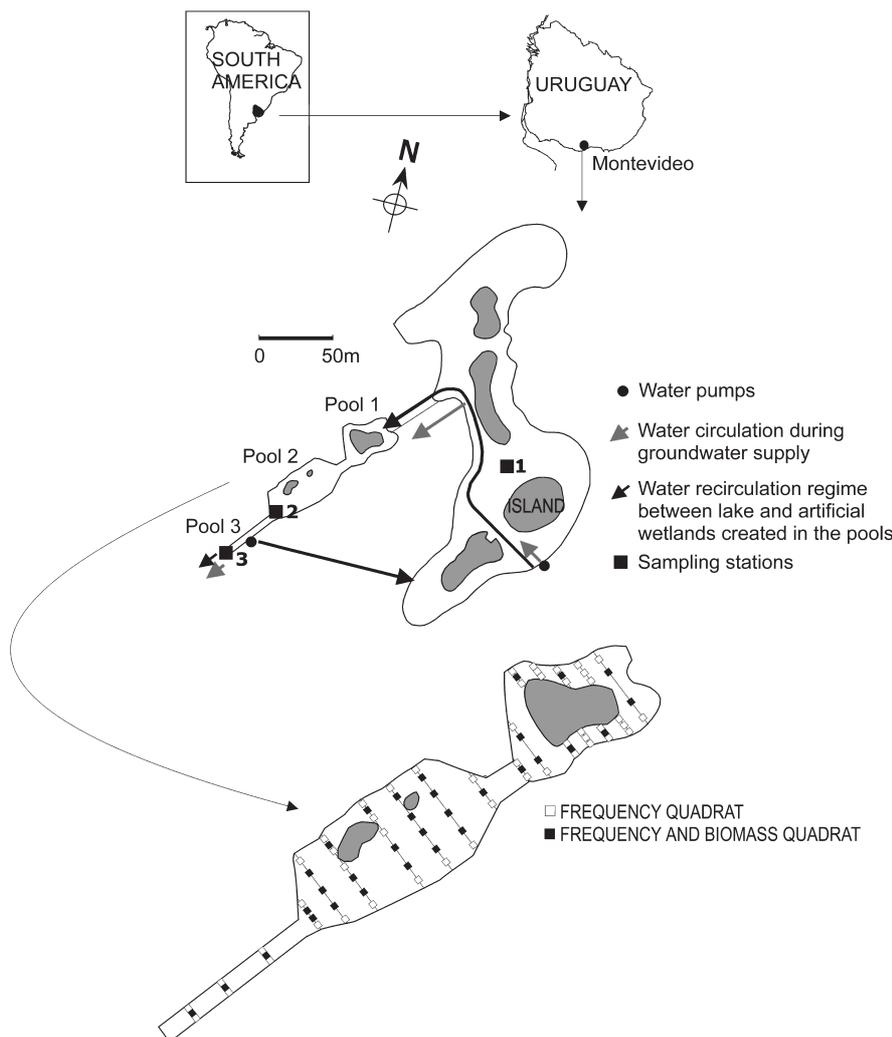
An alternative strategy for restoring hypertrophic, shallow lakes is the creation of artificial wetlands, whereby a water recirculation system is established to reduce the internal nutrient load (Lowe *et al.* 1992). Several mechanisms are implied in the nutrient removal, including plant assimilation, sedimentation, adsorption and precipitation of P, and denitrification and volatilization of ammonia ( $\text{NH}_4^+$ ) (DeBusk & Reddy 1987; Weisner *et al.* 1994). As wetlands can act as nutrient sinks, sources or transformers (Mitsch & Gosselink 1986), the functioning of the artificial wetland design needs to be evaluated, as does the macrophyte-associated nutrient budget in relation to other compartments of the lake.

The objective of this work was to evaluate the effects of a constructed wetland (free-floating macrophyte-based system) on a shallow, hypertrophic lake undergoing restoration. We also analysed the improved water quality, and discuss the most adequate and efficient management, based on the uses of the system.

## MATERIALS AND METHODS

### Study area

Lake Rodó is an artificial lake, located in a park of Montevideo city, Uruguay ( $35^{\circ}55'S$ ,  $56^{\circ}10'W$ ). It is a small (1.3 ha), shallow system (maximum depth,  $Z_{\text{max}} = 2.5$  m), without emergent or floating plants, but with an incipient development of submerged macrophytes (*Potamogeton pectinatus* L.). The lake discharges its waters to Río de la Plata, after it passes through three pools covered by introduced free-floating plants (Fig. 1, Table 1). Eutrophication symptoms resulting from nutrients and organic matter (OM) received from the drainage basin are manifested as



**Fig. 1.** The geographic location of Lake Rodó, identifying the physicochemical (1, 2 and 3) and sediment (1) sampling stations. The arrows indicate the previous (grey) and current (black) water circulation patterns. At the bottom of the figure, the transects followed for sampling hydrophyte biomass and frequency in the pools are indicated with squares.

nuisance phytoplankton blooms in the lake (Scasso *et al.* 2001). The production and accumulation of excessive OM also has caused oxygen (O<sub>2</sub>) depletion at the bottom of the lake and pools, as well as the production of methane and sulphide gases. For these reasons, a restoration programme was developed in 1996. The measures applied to the lake are as follows.

### Nutrient control

1. Spring 1996. The lake was completely emptied and nearly 1 m of sediment was removed. The pluvial water inlet was eliminated.

2. Summer 1997. Ground water was used to refill the lake, and as a permanent water supply (360 m<sup>3</sup> d<sup>-1</sup>).

3. Winter 1998. The supply of ground water was eliminated due to its high nutrient concentration. Instead, a water recirculation system was implemented between the lake and the pools (170 m<sup>3</sup> d<sup>-1</sup>) (Fig. 1, Table 1). *Eichhornia crassipes* and *P. stratiotes* were introduced into the pools, creating an artificial wetland. Other species (e.g. *Spirodela intermedia* W. Koch.) occurred naturally.

4. Spring and summer 1998–1999, 1999–2000. A hydrophyte-harvest programme was implemented to increase the nutrient removal efficiency of the recirculation system. Beginning in October 1998, *E. crassipes* was harvested monthly and completely removed in autumn, leaving inoculums for the next growing season. In winter and spring of 1999, *S. intermedia* was harvested biweekly.

### Biomanipulation

1. Winter and spring 1997, 1998 and 1999. Small omnivorous–planktivorous fish were removed to diminish the predation pressure on zooplankton, thereby increasing the consumption of phytoplankton.

2. Spring 1997. Native piscivorous fish were introduced to decrease planktivorous fish by means of predation.

### Methodology

The hydrophyte community of the pools was studied over two growing seasons: October 1998–February 1999 and August 1999–March 2000. A systematic, stratified sampling was conducted monthly to determine simultaneously the hydrophyte species frequency and biomass, using a total of 82 quadrants of 1 m<sup>2</sup> and 32 quadrants of 0.22 m<sup>2</sup>, respectively (Fig. 1). Samples were separated into species and weighed. Dry weight (DW) was estimated by drying 10 random entire plants of each species at 80°C for 48 h. The hydrophyte data from the three pools were considered together in order to simplify the analyses. The harvested biomass was separated into species and weighted *in situ* (DW estimation followed the same procedure explained above).

Each species cover area (CA, m<sup>2</sup>) was determined as the product of the pool areas (m<sup>2</sup>) and their frequency (%). The species-specific biomass registered in all pools was calculated on the basis of the following procedure: (i) CA was multiplied by the average biomass (g DW m<sup>-2</sup>) registered in each sampling; and (ii) the previous result was divided by the total area of the pools and expressed in g DW m<sup>-2</sup>. Production was calculated as the temporal biomass increment (Roberts *et al.* 1985) and, therefore, the harvested biomass (DW) was added to the estimated one for each sampling date.

After drying, whole plants were macerated and three random subsamples were analysed for nutrient concentration. Plant N concentration was determined by the Kjeldahl method (American Public Health Association 1985), using 0.1 g of plant powder. To analyse P concentration (0.02 g of plant powder), the plant samples were digested with concentrated perchloric-nitric-sulphuric (1:5:0.5) acids, and the oxidized reactive form was determined by the Murphy and Riley (1962) technique. The N and P concentrations in the standing crop (nutrient standing stock) of each species (expressed in gm<sup>-2</sup>) were

**Table 1.** Morphometrical data for Lake Rodó and the associated pools, based on criteria listed by Håkanson (1981)

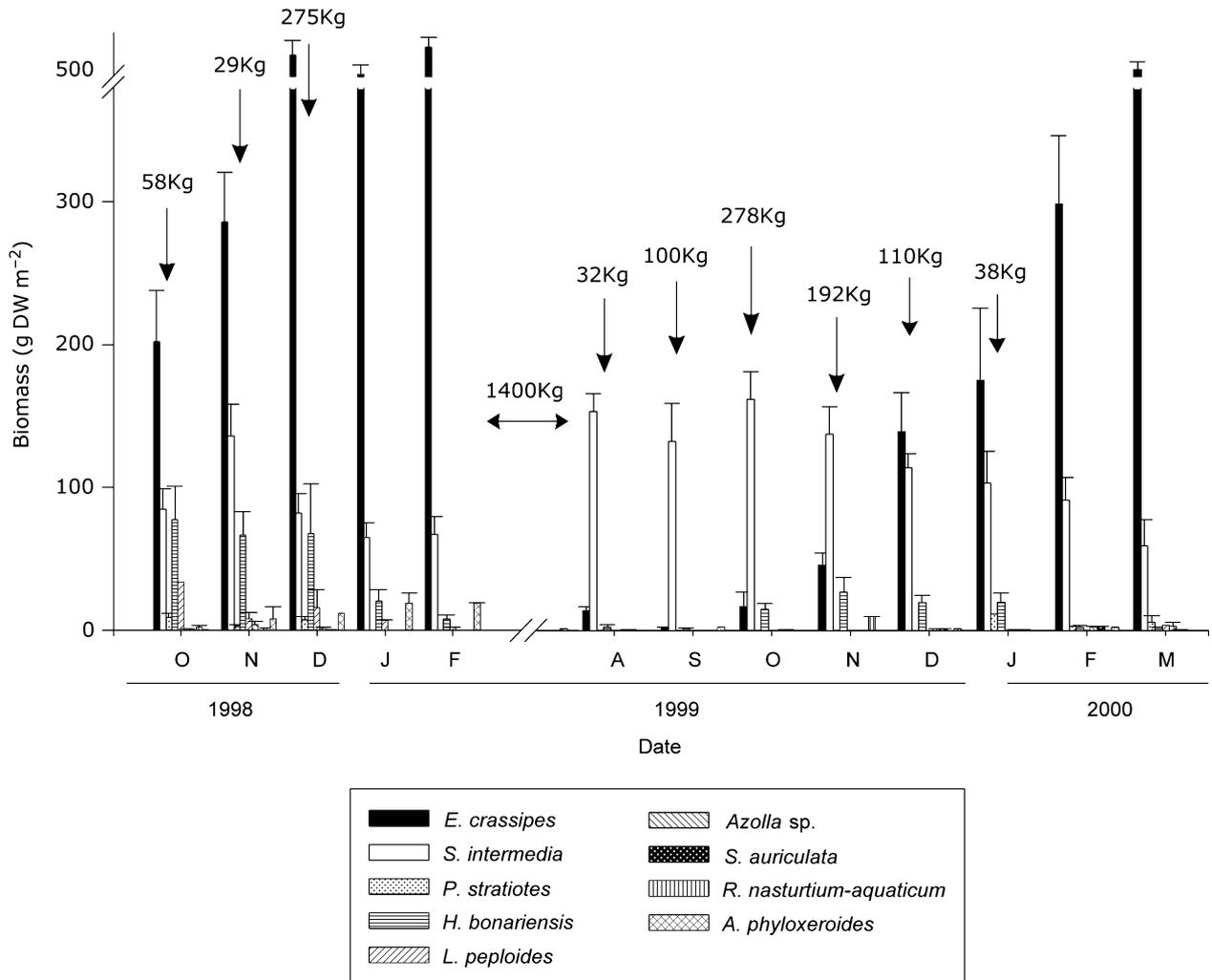
Characteristics	Lake	Pool 1	Pool 2	Pool 3
Total lake area (m <sup>2</sup> )	15 000	630	980	170
Lake area (m <sup>2</sup> )	12 000	450	920	170
Mean depth (m)	1.7	0.4	1.1	1.0
Maximum depth (m)	2.5	1.0	1.1	1.0
Volume (m <sup>3</sup> )	20 794	181	1008	169
Maximum length (m)	257	36	50	37.5
Maximum width (m)	75	33	24	4.5
Shoreline length (m)	807	103	125	84
Shoreline development	2.1	1.3	1.2	1.8

calculated by multiplying the species-specific biomass by the percentage of N and P. Nutrient uptake or release were estimated as the temporal variation of N and P standing stock of each species (Chapin & Van Cleve 1989).

The water physicochemical parameters were measured in the deepest part of the lake (station 1) and in the pools 2 and 3 (stations 2 and 3) at midday (Fig. 1). The Secchi disk depth (SD), temperature, dissolved oxygen (DO), electrical conductivity and pH were measured biweekly *in situ* with Horiba equipment (Horiba, Kyoto, Japan). Lake surface and bottom samples were collected for nutrient analyses with a Ruttner bottle (Hydro-Bios, Kiel, Germany), with an integrated surface–bottom sample taken for chlorophyll *a* (Chl *a*). Suspended solids (SS) (American Public Health Association 1985), soluble reactive phosphorus (SRP)

(Murphy & Riley 1962), total phosphorus (TP) (Valderrama 1981), nitrate as nitrogen ( $\text{NO}_3^-$ ) (Müller & Widemann 1955), ammonium as nitrogen ( $\text{NH}_4^+$ ) (Koroleff 1970), total nitrogen (TN) (Valderrama 1981), silicate (SR) (Mullin & Riley 1955) and Chl *a* (hot ethanol, Nusch 1980) were determined biweekly, from January 1997–January 1999, and monthly until March 2000. Ten sediment samples (10-cm depth) were collected bimonthly from station 1 with a Kajak-corer (6-cm diameter) (Uwitec, Mondsee, Austria), with TN and TP analysed for every 2 cm of the sample, according to Valderrama (1981). Mean daily air temperatures were taken from the National Meteorology Service.

To evaluate the nutrient removal capacity of the water recirculation system, the sediment and water column nutrient load were estimated. The sediment nutrient load was determined for the uppermost 2 cm (containing the



**Fig. 2.** The monthly variation of the main hydrophyte species biomass. The arrows indicate the biomass of the community harvested in each month (dry weight). The break over the x-axis represents a period of *Eichhornia crassipes* biomass extractions, when no sampling was conducted.

**Table 2.** Temporal variation of hydrophyte nitrogen (N) and phosphorus (P) content (%) and total nutrient content in the community ( $\text{gm}^{-2}$ ). The mean and standard error are indicated ( $n = 3$ ), and no-value months correspond to an absence of those species

Species	Element	1998					1999					2000		
		Oct	Nov	Dec	Jan	Feb	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar
<i>E. crassipes</i>	N	3.21 ± 0.11	3.25 ± 0.03	2.18 ± 0.02	1.10 ± 0.02	1.27 ± 0.02	2.45 ± 0.01	3.22 ± 0.00	3.56 ± 0.06	3.72 ± 0.06	3.06 ± 0.00	2.36 ± 0.02	2.54 ± 0.05	2.97 ± 0.05
	P	0.33 ± 0.00	0.38 ± 0.03	0.27 ± 0.01	0.19 ± 0.00	0.23 ± 0.02	0.28 ± 0.02	0.32 ± 0.00	0.39 ± 0.02	0.60 ± 0.01	0.29 ± 0.01	0.22 ± 0.01	0.62 ± 0.07	0.55 ± 0.16
<i>S. intermedia</i>	N	2.99 ± 0.05	2.69 ± 0.12	2.91 ± 0.08	2.37 ± 0.04	2.33 ± 0.02	3.32 ± 0.01	3.51 ± 0.02	3.28 ± 0.04	3.46 ± 0.04	3.05 ± 0.06	2.19 ± 0.03	3.53 ± 0.12	2.97 ± 0.12
	P	0.32 ± 0.01	0.37 ± 0.03	0.40 ± 0.02	0.28 ± 0.01	0.21 ± 0.02	0.30 ± 0.01	0.37 ± 0.01	0.35 ± 0.01	0.36 ± 0.02	0.33 ± 0.00	0.20 ± 0.01	0.52 ± 0.02	0.50 ± 0.02
<i>P. stratiotes</i>	N	3.52 ± 0.05	3.30 ± 0.02	2.74 ± 0.04	3.97 ± 0.05	-	-	-	-	-	-	3.20 ± 0.12	3.33 ± 0.09	2.78 ± 0.08
	P	0.87 ± 0.01	0.48 ± 0.02	0.39 ± 0.00	0.33 ± 0.10	-	-	-	-	-	-	0.26 ± 0.01	0.31 ± 0.05	0.30 ± 0.01
<i>H. bonariensis</i>	N	2.56 ± 0.06	3.58 ± 0.15	2.96 ± 0.21	3.85 ± 0.08	2.61 ± 0.11	3.73 ± 0.14	4.16 ± 0.11	4.12 ± 0.13	2.60 ± 0.04	3.22 ± 0.06	4.30 ± 0.15	3.99 ± 0.16	4.60 ± 0.17
	P	0.37 ± 0.03	0.48 ± 0.01	0.45 ± 0.01	0.60 ± 0.01	0.67 ± 0.02	0.67 ± 0.02	0.50 ± 0.02	0.52 ± 0.02	0.23 ± 0.06	0.40 ± 0.00	0.51 ± 0.03	0.53 ± 0.04	0.46 ± 0.05
<i>L. peploides</i>	N	3.46 ± 0.02	2.15 ± 0.07	1.61 ± 0.17	1.43 ± 0.04	1.83 ± 0.15	-	-	-	-	-	-	-	3.65 ± 0.13
	P	0.82 ± 0.01	0.65 ± 0.01	0.65 ± 0.05	0.45 ± 0.03	0.71 ± 0.04	-	-	-	-	-	-	-	0.47 ± 0.03
<i>Azolla</i> sp.	N	3.46 ± 0.06	2.83 ± 0.05	3.57 ± 0.00	-	-	-	-	-	-	3.24 ± 0.05	3.03 ± 0.13	3.68 ± 0.04	3.57 ± 0.07
	P	0.29 ± 0.02	0.28 ± 0.00	0.41 ± 0.01	-	-	-	-	-	-	0.25 ± 0.03	0.25 ± 0.01	0.25 ± 0.01	0.21 ± 0.01
<i>S. auriculata</i>	N	4.13 ± 0.00	2.76 ± 0.00	3.57 ± 0.00	-	-	3.76 ± 0.12	-	3.62 ± 0.09	3.67 ± 0.04	3.56 ± 0.02	3.38 ± 0.18	3.12 ± 0.05	3.65 ± 0.07
	P	0.46 ± 0.07	0.29 ± 0.02	0.48 ± 0.02	-	-	0.32 ± 0.05	-	0.27 ± 0.01	0.40 ± 0.04	0.30 ± 0.01	0.29 ± 0.01	0.26 ± 0.01	0.36 ± 0.01
<i>R. nasturtium-aquaticum</i>	N	6.02 ± 0.07	4.22 ± 0.17	-	-	-	3.97 ± 0.16	5.09 ± 0.25	3.89 ± 0.13	-	-	-	-	-
	P	0.64 ± 0.02	0.42 ± 0.01	-	-	-	0.40 ± 0.01	0.54 ± 0.02	0.41 ± 0.02	-	-	-	-	-
A. phylloxoeroides	N	3.57 ± 0.07	3.53 ± 0.17	1.99 ± 0.02	1.60 ± 0.09	2.55 ± 0.08	2.70 ± 0.05	2.72 ± 0.04	3.78 ± 0.04	-	3.48 ± 0.06	-	1.26 ± 0.02	2.80 ± 0.16
	P	0.36 ± 0.04	0.48 ± 0.03	0.18 ± 0.01	0.22 ± 0.01	0.25 ± 0.01	0.26 ± 0.02	0.21 ± 0.04	0.35 ± 0.01	-	0.27 ± 0.02	-	-	-
Community ( $\text{g m}^{-2}$ )	N	12.6	16.0	18.7	7.8	11.0	5.8	5.4	6.6	7.1	13.9	7.6	11.2	17.1
	P	1.6	2.0	2.5	1.3	1.8	0.5	0.6	0.7	0.8	1.5	0.7	2.4	3.1

nutrient fraction most rapidly interchangeable with the water column) and extrapolated to the whole lake area. The nutrient load in the water column was estimated with the average concentration of surface and bottom samples and extrapolated to the lake volume.

The study period was divided into three hydrological stages; denominated groundwater supply, water recirculation and hydrological isolation. The last stage occurred as the water recirculation system malfunctioned and was used as a comparison with the other stages. One-way ANOVA analyses were performed to analyse the differences of: (i) total hydrophyte biomass (sum of species-specific biomasses) between the two growing seasons (two levels); and (ii) lake physicochemical variables (station 1) among the above-noted hydrological stages (three levels). The differences in physicochemical variables among the lake and the hydrophyte-covered pools were analysed by two-way repeated measures ANOVAs, for all the periods, except for the groundwater supply. The differences in OM and nutrient content in the sediment among the three hydrological stages and sediment depth were analysed by two-way ANOVAs (factors: stages and depth). Normality was assessed by the Kolmogorov-Smirnov's tests. ANOVA assumptions were tested by Cochran's tests and by visual inspection of fitted values against residual terms. In the case of heteroscedasticity, the variables were log or square-root transformed. Post-hoc comparisons were performed by the Tukey's HSD test. Correlation analyses were done with Spearman's tests.

## RESULTS

### Hydrophyte community

The hydrophyte community was composed of nine species (Fig. 2): five free-floating (*E. crassipes*, *P. stratiotes*, *S. intermedia*, *Salvinia auriculata* AUBL and *Azolla* sp. Lam.) and four small, emergent (*Hydrocotyle bonariensis* Lam., *Ludwigia peploides* (HBK) Raven, *Rorippa nasturtium-aquaticum* (L.) Hayek and *Alternanthera phyloxeroides* (Mart.) Gris). All of the emergent species, except *R. nasturtium-aquaticum*, were epiphytes on floating mats of *E. crassipes* and *S. intermedia*.

The community biomass showed significant differences ( $F = 11.41$ ,  $P < 0.001$ ) between the two growing periods (Fig. 2). The maximum values were achieved in summer 1998 (721 g DW m<sup>-2</sup>, mean of December, January and February values) and March 2000 (573 g DW m<sup>-2</sup>). *Eichhornia crassipes* and *S. intermedia* reached the highest biomass values during the warmer and colder months, respectively. *Eichhornia crassipes* represented 87% of the community biomass in summer and *S. intermedia* represented 90% at the beginning of the spring. *Hydrocotyle bonariensis* was the third most important species, in terms of its total biomass (Fig. 2). The maximum production values of *E. crassipes* were 11.3 and 5.6 g DW m<sup>-2</sup> d<sup>-1</sup> in December 1998 and March 2000, respectively. The maximum production values of *S. intermedia* were 2.7 and 0.8 g DW m<sup>-2</sup> d<sup>-1</sup> in December 1999 and February 2000, respectively.

**Table 3.** The mean and range of physicochemical characteristics of Lake Rodó (station 1) during groundwater supply, water recirculation and water isolation stages

Stage	Oxygen saturation (%)		SD (m)	T (°C)	SS (mg L <sup>-1</sup> )	NO <sub>3</sub> (µg L <sup>-1</sup> )	NH <sub>4</sub> (µg L <sup>-1</sup> )	TN (µg L <sup>-1</sup> )	SRP (µg L <sup>-1</sup> )	TP (µg L <sup>-1</sup> )	SR (µg L <sup>-1</sup> )	Chl-a (µg L <sup>-1</sup> )
	Surface	Bottom										
Groundwater supply												
Mean	113.0	99.9	0.6	18.3	32.9	6429.6	15.9	13 753.9	15.4	149.3	7979.5	123.2
Minimum	60.4	55.3	0.3	10.7	19.6	2648.0	0.0	6772.8	nd	100.9	2557.5	34.2
Maximum	173.9	176.3	1.4	25.8	58.5	9746.2	95.0	24 710.6	42.3	195.4	18 284.0	232.5
Recirculation												
Mean	122.6	51.7	0.5	19.1	23.0	392.4	31.2	2315.4	21.7	190.5	7201.7	76.8
Minimum	78.2	0.3	0.3	11.0	12.9	58.2	0.0	988.8	nd	106.4	1970.2	13.6
Maximum	192.1	106.1	1.1	28.8	31.6	1752.2	224.8	4155.2	83.4	431.0	20 489.3	138.1
Water isolation												
Mean	129.9	38.1	0.4	19.3	30.6	297.5	57.6	2421.3	50.1	269.7	6801.1	133.9
Minimum	99.8	0.0	0.3	10.0	23.3	62.2	4.3	1073.9	nd	133.7	285.2	62.3
Maximum	167.6	98.3	0.5	29.2	39.0	1029.6	164.4	3447.2	112.7	394.4	15 060.0	225.9

nd, undetectable by the method used; Chl-a, chlorophyll-a; NH<sub>4</sub>, ammonia; NO<sub>3</sub>, nitrate; SD, Secchi disk depth; SR, silicate; SRP, soluble reactive phosphorus; SS, suspended solids; T, temperature; TN, total nitrogen; TP, total phosphorus.

Meteorological conditions differed between the growing periods. Winter 1999 was characterized by lower mean daily water and air temperatures (9.8°C and 0.4°C, respectively) than winter 1998 (12.2°C and 1.4°C). Three freezing events were registered only during the second winter (1999).

The percentage of N and P in the biomass varied among species and growing periods (Table 2). *Eichhornia crassipes* and *S. intermedia* showed the same temporal pattern, achieving the maximum N and P percentages in spring (October, November, December) and the minimum in January and February. The maximum nutrient standing stock of the community was observed in late spring (1998 and 1999) and late summer 2000 (Table 2). The nutrient load associated with the macrophyte community was correlated with total biomass ( $r_s = 0.81, P < 0.001$  and  $r_s = 0.79, P < 0.001$  for N and P, respectively), but not with the nutrient concentration of the species. Temporal variations of N and P concentrations were coupled in

*E. crassipes* ( $r_s = 0.74, P < 0.01$ ), *S. intermedia* ( $r_s = 0.57, P < 0.05$ ), *S. auriculata* ( $r_s = 0.64, P < 0.05$ ), *R. nasturtium-aquaticum* ( $r_s = 0.90, P < 0.05$ ) and *A. phylloxeroides* ( $r_s = 0.74, P < 0.05$ ).

*Eichhornia crassipes* presented the highest nutrient assimilation values, especially during the second growing period. The maximum values of N assimilation by *E. crassipes* were 148.9 (spring 1998) and 202.6 mg N m<sup>-2</sup>d<sup>-1</sup> (autumn 2000). *Spirodela intermedia* assimilated a maximum of 41.0 and 78.0 mg N m<sup>-2</sup>d<sup>-1</sup> in the spring of 1998 and 1999, respectively. The maximum P assimilation by *E. crassipes* was 21.1 and 52.2 mg P m<sup>-2</sup>d<sup>-1</sup> in spring of 1998 and in summer of 2000, respectively. For *S. intermedia*, these values were 8.6 and 7.5 mg P m<sup>-2</sup>d<sup>-1</sup> in the spring of 1998 and 1999, respectively. Negative assimilation values were measured during some periods, especially for *E. crassipes*, after the maximum of biomass had been achieved in December 1998. The same occurred for *S. intermedia* when *E. crassipes* started its growing cycle.

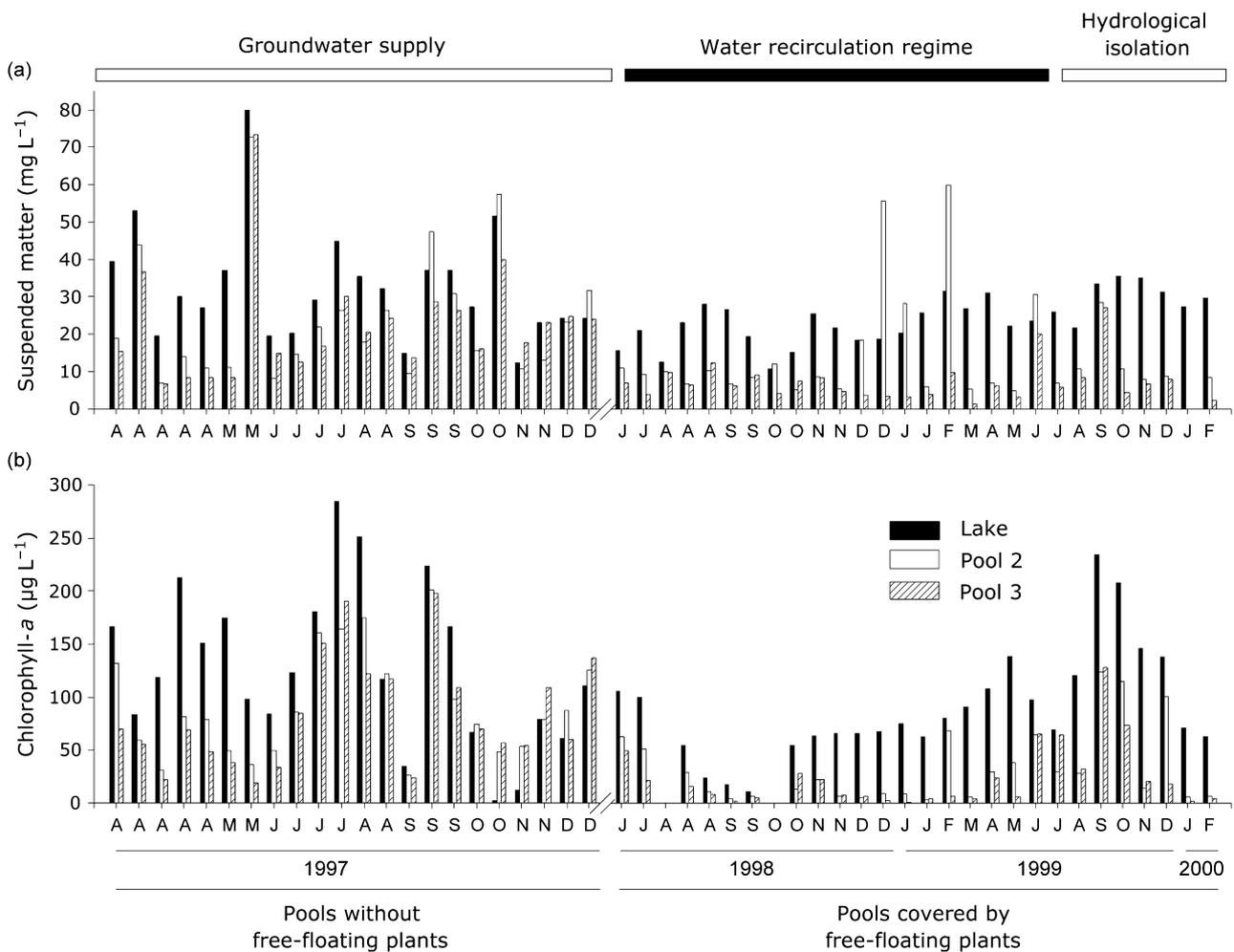


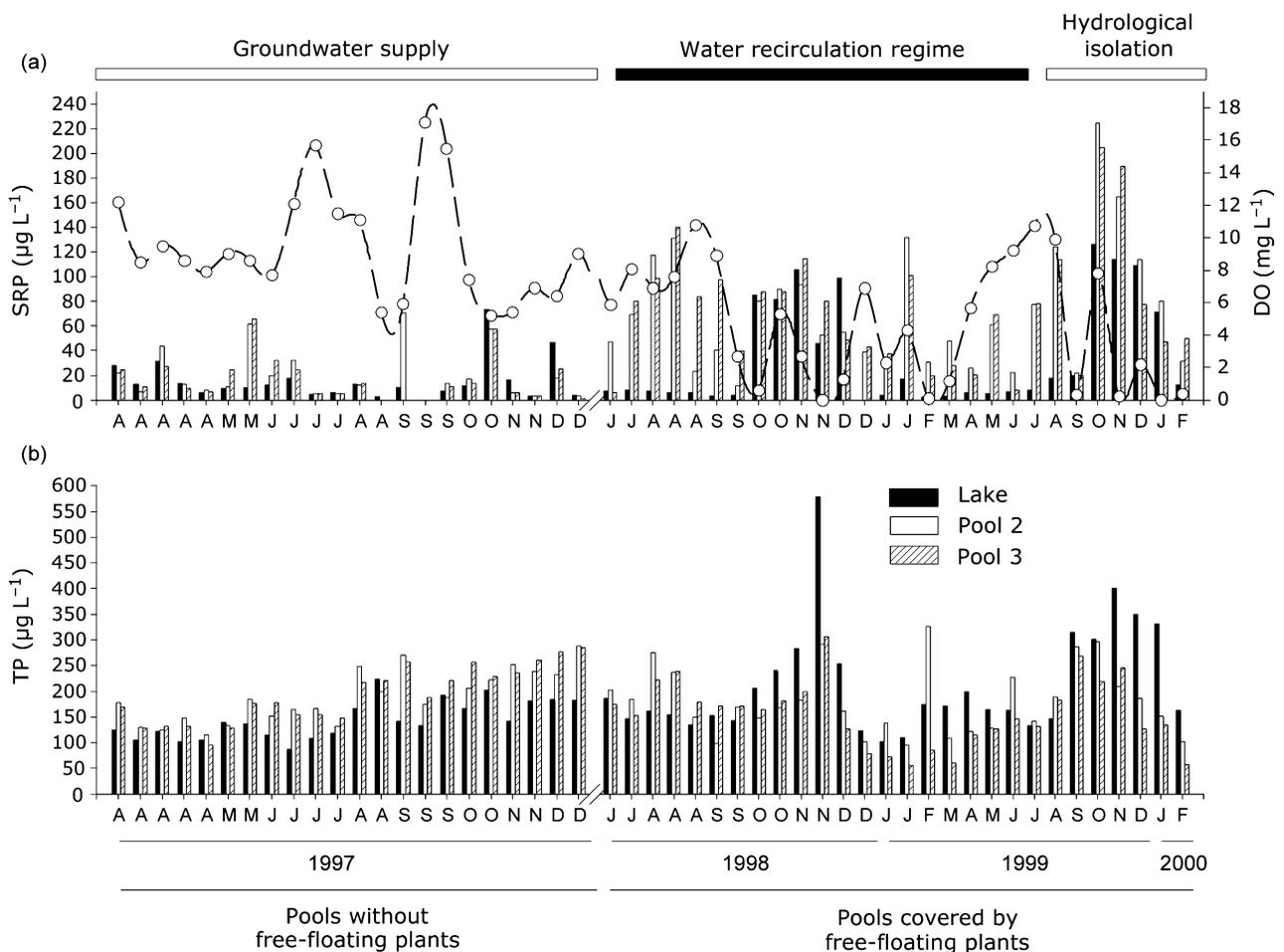
Fig. 3. Temporal variation of: (a) suspended matter and (b) chlorophyll-a in the lake (surface and bottom average) and pools.

### Water and sediment physicochemical analysis

Low water transparency, elevated algal biomass and high nutrient concentrations indicated a hypertrophic condition for Lake Rodó (Table 3). The lowest algal biomass was observed during the water recirculation period ( $F = 3.66$ ,  $P < 0.05$ ), although the Secchi disk depth showed no significant change during this period. In contrast, an increase in the algal biomass and a reduction in water transparency were registered during the hydrological isolation stage (Fig. 3, Table 3). Except for the hydrological isolation stage, spring clear-water phases ( $SD > 1.0$  m) were registered for two or three weeks. The euphotic zone depth included the entire water column most of the time during the first two hydrological stages, but not during the hydrological isolation period. The  $O_2$  concentration at the bottom of the lake decreased during the recirculation and hydrological isolation periods ( $F = 5.98$ ,  $P < 0.01$ ), especially during the warmest months (Fig. 4, Table 3). After the groundwater supply was eliminated, the most

significant change was a reduction in the TN concentration ( $F = 47.1$ ,  $P < 0.001$ ), related mainly to a decreased  $NO_3^-$  level ( $F = 85.6$ ,  $P < 0.001$ ) associated with groundwater contamination (Fig. 5, Table 3). In contrast, the TP and SRP concentrations increased ( $F = 6.2$ ,  $P < 0.01$ ;  $F = 3.6$ ,  $P < 0.05$ , respectively), especially the SRP concentration when the lowest  $O_2$  values were registered at the bottom (Fig. 4). The SR remained constant.

The lake and pools showed differences in the water physicochemical parameters during the study period (Figs 3, 4 and 5). Lower  $O_2$  concentrations ( $F = 6455.5$ ,  $P < 0.001$ ), SS ( $F = 3334.9$ ,  $P < 0.001$ ), Chl *a* ( $F = 2758.7$ ,  $P < 0.001$ ), TP ( $F = 217.2$ ,  $P < 0.001$ ), TN ( $F = 14.1$ ,  $P < 0.01$ ) and SR ( $F = 311.7$ ,  $P < 0.001$ ) were recorded in the pools. In contrast, the inorganic, dissolved forms of N and P were higher in the pools compared to the lake ( $NH_4^+$ :  $F = 41.8$ ,  $P < 0.001$ ;  $NO_3^-$ :  $F = 9.6$ ,  $P < 0.01$  and SRP:  $F = 3076.6$ ,  $P < 0.001$ ). The constructed wetland had the capacity to remove 52% of the SS (average), with a maximum capacity in summer (95%). The same estimations of retention of algal



**Fig. 4.** Temporal variation of: (a) soluble reactive phosphorus (SRP) (bars) and dissolved oxygen (DO) (at the bottom of the lake,  $\circ$ ) and (b) total phosphorus (TP). Soluble reactive phosphorus and TP represent the concentrations in the lake (surface and bottom average) and pools.

biomass (Chl *a*) were 52% and 99% (summer), and 36% and 75% for TN (summer and winter). Total phosphorus was removed only during spring and summer, with the maximum capacity measured during summer (65%). All these estimations corresponded to the water recirculation period, with a hydraulic residence time of  $\approx 3.5$  days.

During the period in which ground water was the major water source, the lowest OM percentage ( $F = 12.4$ ,  $P < 0.001$ ) and nutrient concentrations were registered in the sediment, the first decreasing with depth (Fig. 6). A temporal tendency to increased values, however, was observed at the end of this period (from 3.8 to 5.6% OM, from 287.4 to 941.4  $\mu\text{g Pg DW}^{-1}$  and from 1980.0 to 9148.0  $\mu\text{g Ng DW}^{-1}$ , top 2-cm average). This OM and P enrichment continued to increase during the water recirculation period, reaching values at the sediment surface (2 cm depth) of  $\approx 20.4\%$  OM and 1390.0  $\mu\text{g Pg DW}^{-1}$ . The same pattern was not observed for N (Fig. 6). During the hydrological isolation, the concentrations of both nutrients increased in the sediment (Fig. 6) and were significantly

higher than those in the previous stages ( $F = 9.1$ ,  $P < 0.001$  and  $F = 15.7$ ,  $P < 0.001$  for N and P, respectively).

The minimum and maximum N and P loads associated with the water column were  $\approx 13.6\text{--}76.4$  kg and 2.2–11.6 kg, respectively, whereas  $\approx 1000\text{--}2700$  kg of N and 170–360 kg of P were associated with the sediments. We estimated that 39.6 kg of N and 5.4 kg of P were removed by biomass harvest in the first growing period and 25.9 kg of N and 2.7 kg of P in the second one. This is equivalent to 88% and 78% of the N and P loads associated with the water column, respectively, in the first period, and 58% and 39% during the second. In contrast, the hydrophyte harvests accounted for 2% of the N and P loads associated with the sediments in the first growing period, and only 1% in the second.

## DISCUSSION

### Plant community

*Eichhornia crassipes* was the dominant and most productive species during the warmer months. The measured biomass and production were similar or lower than the values

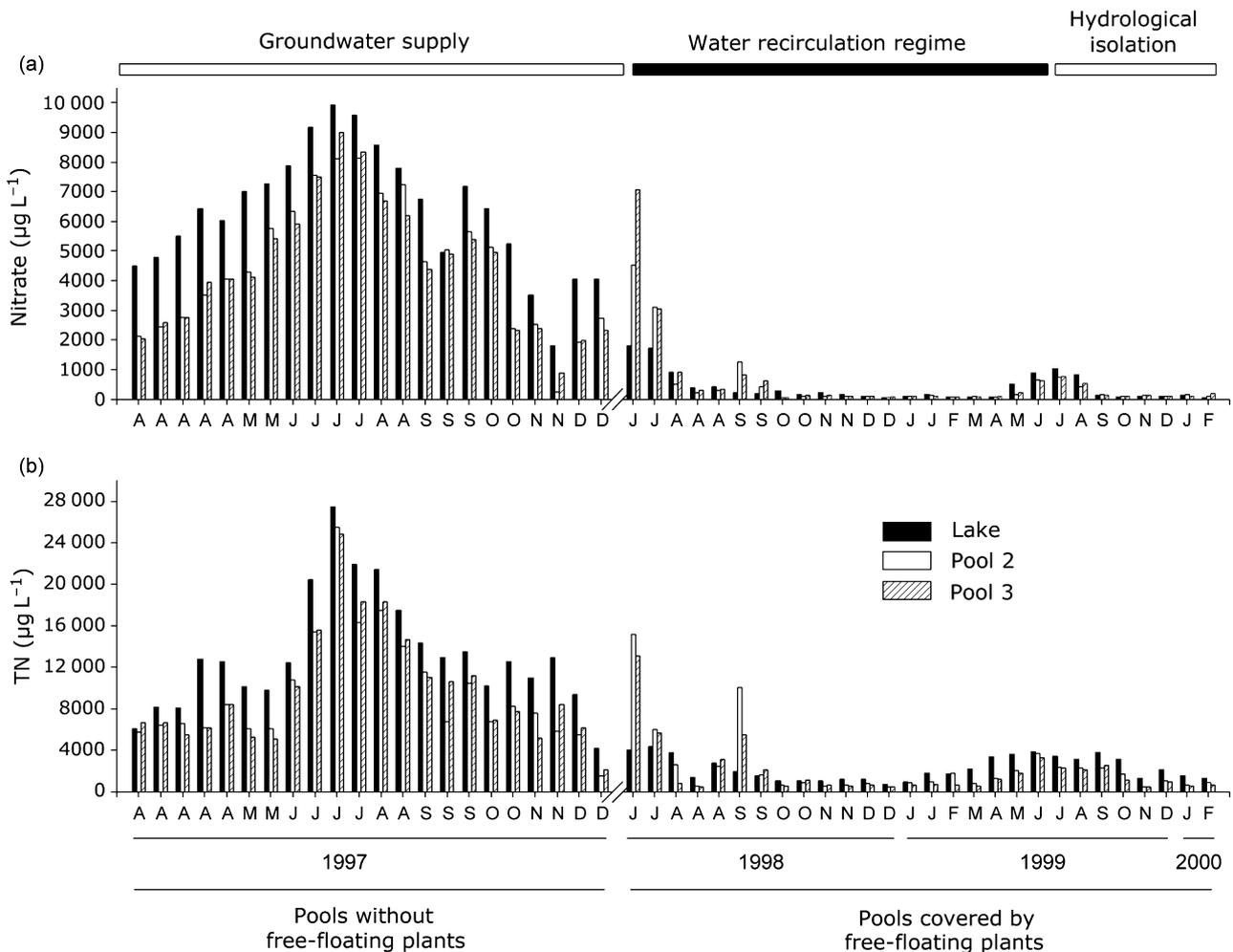


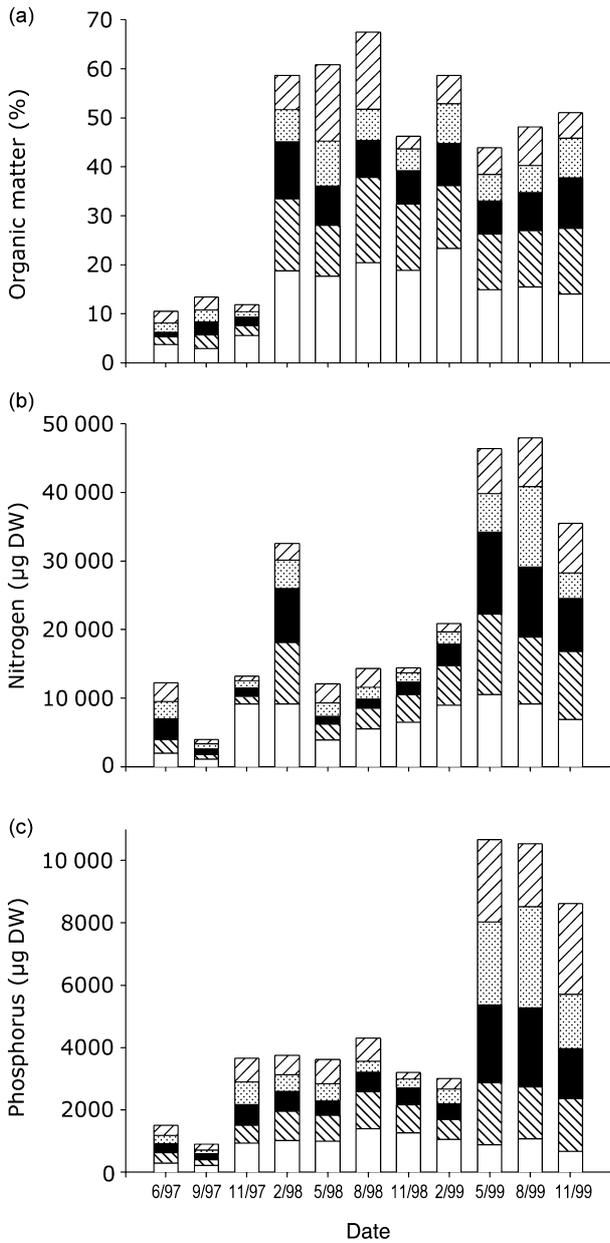
Fig. 5. Temporal variation of: (a) nitrate and (b) total nitrogen (TN) in the lake (surface and bottom average) and pools.

reviewed by Vymazal (1995) for subtropical regions. This species was easy to cultivate and harvest. Consistent with other results (DeBusk *et al.* 1981; Reddy & DeBusk 1984; Záková *et al.* 1994), *E. crassipes* was the most suitable macrophyte for removing nutrients in the water recirculation system. However, because *E. crassipes* is a low-temperature, sensitive plant, and *S. intermedia* can maintain high biomass production during winter, the latter appears to be an alternative and complementary species for subtropical systems. In spring and autumn, the other

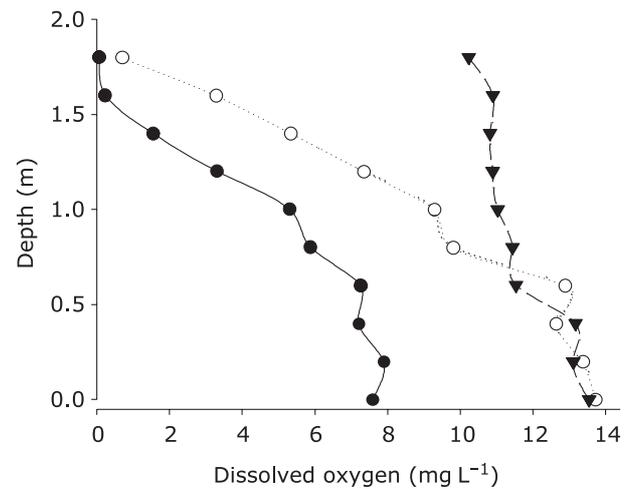
species, particularly the epiphytes and *R. nasturtium-aquaticum*, seem to efficiently complement the production of the community.

The absence of freezing events and the higher water temperatures during winter 1998 allowed the community to grow earlier and reach maximum biomass levels in December. In the second growing period, the lower temperatures registered in winter and spring 1999 might have slowed the growth of *E. crassipes*, allowing the other free-floating species to persist in the summer. A more diverse community assures all-year-round biomass production. As was found for *E. crassipes* (DeBusk *et al.* 1981), macrophyte monocultures are more susceptible to diseases and insect attacks. In this sense, the first introduced species, *P. stratiotes*, collapsed after a severe insect attack. Thus, diverse communities are more advantageous for water treatment.

The N and P concentrations for each species were within the range reviewed by Vymazal (1995). The nutrient standing stock and uptake, however, showed lower values. Water nutrient concentrations might limit biomass production, especially during the highest plant density phases, when nutrient consumption is at its maximum (Reddy *et al.* 1989, 1990). Balasooriya *et al.* (1984) suggested that *E. crassipes* growth is limited by SRP,  $\text{NO}_3^-$  and  $\text{NH}_4^+$  concentrations  $< 0.04$ ,  $0.03$  and  $0.21 \text{ mg L}^{-1}$ , respectively. The nutrient concentrations in the pools were often lower than these values, so *E. crassipes* growth could have been nutrient-limited. Roots that were 90 cm in length, and reaching the sediments, were observed, especially during summer, suggesting nutrient limitation (DeBusk & Reddy 1987; Kim & Kim 2000).



**Fig. 6.** Variation of organic matter and nutrient (nitrogen and phosphorus) content in the sediments, based on time and depth. □, 0–2 cm; ▨, 3–4 cm; ■, 5–6 cm; ▩, 7–8 cm; ▤, 9–10 cm.



**Fig. 7.** Dissolved oxygen profiles in the lake before (●, 10 January 2000 and ○, 7 February 2000) and after one week of groundwater supply (▼, 22 February 2000).

### Benefits of the water recirculation system with aquatic plants

The hydrophyte cover diminished water turbulence, facilitating sedimentation of particulate material and also the retainment of phytoplankton, SS and associated nutrients in the pools. Similar results were obtained by Poi de Neiff *et al.* (1994), Maine *et al.* (1999) and Kim and Kim (2000). In this sense, the root system acted as a filter, facilitating the decline, deposition and decomposition of the phytoplankton (Kim & Kim 2000). These mechanisms were also very effective in retaining allochthonous detritus originating from trees surrounding the lake.

By preventing gaseous exchange with the atmosphere, the decomposition of OM and the hydrophyte cover in the pools led to O<sub>2</sub> depletion. The resulting reduced chemical conditions promoted denitrification, nutrient release and the accumulation of OM in the sediments. The dissolved, inorganic nutrients that were released exceeded the assimilation capacity of the hydrophyte community. As a result, the NH<sub>4</sub><sup>+</sup>, SRP and NO<sub>3</sub><sup>-</sup> concentrations were higher in the pools. According to the criteria of Mitsch and Gosselink (1986), the pools acted as a transformer wetland, retaining the nutrient particulate fractions and exporting the dissolved, inorganic forms.

The hydrophyte harvests were efficient in removing the N and P associated with the water column but were insufficient for those nutrients associated with the sediments. The increased sediment OM and nutrient content, after elimination of the groundwater supply and during the water recirculation period, clearly indicates the inefficiency of the system designed to decrease the sediment-associated nutrient load. In addition to the groundwater-associated supply, other external nutrient sources could exist, one possibility being the abundant deciduous trees surrounding the lake. In agreement with the results of Carpenter and Adams (1977), our strategy had only a limited control of the internal load of the entire lake. The eutrophication process, however, could be attenuated with its ecological advantages. The algal biomass reduction was a result of the filtering capacity of the hydrophytes, not because of the reduction in nutrient concentrations. Accordingly, we concluded that the water recirculation system was an adequate measure for the restoration of this lake.

### Water recirculation system improvement

The harvest strategy should be different, based on the purposes of the system. To maintain a highly productive system, for example, a frequent harvest regimen would be more appropriate. The biomass should be kept within an optimum density range when biomass production is maximal (DeBusk & Reddy 1987). Based on our results

and those of Reddy and DeBusk (1984), the *E. crassipes* optimum density range was 500–1000 g DW m<sup>-2</sup>. Thus, harvests should be carried out after achievement of a biomass level of 1000 g DW m<sup>-2</sup>, leaving an average of no less than 500 g DW m<sup>-2</sup>, which allows for extension of the exponential growth phase. For *S. intermedia*, the optimum density range is 150–250 g DW m<sup>-2</sup>. A biweekly harvest frequency would maintain the biomass within that range during winter and spring. This strategy also would be most appropriate for controlling the internal P load as it depends mainly on biomass assimilation and growth rates (DeBusk & Reddy 1987). In contrast, the maximum N removal is reached when dense mats of macrophytes promote denitrification, thereby identifying infrequent harvest as the adequate control strategy (Reddy *et al.* 1989).

The considerable decrease in the internal N load, in contrast to that of P, resulted in important changes in the nutrient ratios in the lake during the water recirculation and hydrological isolation stages. In addition, the longer residence time favoured the establishment of populations of inedible cyanobacteria (Kruk 2001), compromising the success of the complementary biomanipulation measures.

The nitrate-rich ground water (Scasso *et al.* 2001) facilitated the decomposition of OM, prevented anoxia processes and the release of P from the sediments. Experimental groundwater supply during summer 2000 (Fig. 7) prevented an O<sub>2</sub> deficit at the bottom. In contrast, the operation of the recirculation system promoted the O<sub>2</sub> deficit at the lake bottom, promoting the release of P from the sediments in the warmest months. Based on these observations, the combined use of the water recirculation system and the groundwater supply in some periods of the year, especially in summer, could constitute a new and more effective management measure. The water recirculation system might promote the algal biomass removal (by filtration), whereas the groundwater injection would avoid anoxia at the bottom of the lake, shifting the TN/TP ratio and reducing the residence time to < 70 days, which are unfavourable conditions for cyanobacteria. Experiments combining underground water supply and water recirculation in summer would corroborate the previous statements.

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