Environ Monit Assess (2007) 132:377–393
DOI 10.1007/s10661-006-9541-x

Eutrophication and Sedimentation Patterns in Complete Exploitation of Water Resources Scenarios: An Example from Northwestern Semi-arid Mexico

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Received: 8 June 2006 / Accepted: 26 September 2006 / Published online: 14 December 2006

Abstract Water requirements to supply human needs lead water stakeholders to store more water during surplus periods to fulfil the demand during – not only – scarcity periods. At the reservoirs, mostly those in semi-arid regions, water level then fluctuates extremely between rises and downward during one single year. Besides of water management implications, changes on physical, chemical and biological dynamics of these drawdown and refilling are little known yet. This paper shows the results, throughout a year, on solids, nutrients (N and P), chlorophyll-*a*, and sedimentation changes on the dynamics, when the former policy was applied in a reservoir from the semi-arid Northwestern Mexico. Water level sinusoidal trend impinged changes on thermal stratification and mixing, modifying nutrient cycling and primary producer responses. According to nitrogen and phosphorus concentration as well as chlorophyll-*a*, reservoir was mesotrophic, becoming hypertrophic during drawdown. Nutrient concentrations were high (1.22 ± 0.70 mg N l^{-1} and 0.14 ± 0.12 mg P l^{-1}), increasing phosphorus and lowering N:P significantly throughout the study period, although no intensive agricultural, no urban development, neither industrial activities take place in the watershed. This suggests nutrient recycling complex mechanisms, including nutrient release from the sediment–water interface as the main nutrient pathway when shallowness, at the same time as mineralization, increases. Outflows controlled nitrogen and phosphorus availability on the ecosystem while organic matter depended on river inflows. As on other subtropical aquatic ecosystems, nitrogen limited primary productivity (Spearman correlation R=0.75) but chlorophyll-*a* seasonal pattern showed an irregular trend, prompting other no-nutrient related limitants. Shallowness induced a homogeneous temporal pattern on water quality. This observed temporal variability was mainly explained statistically by changes on solids (mineral and organic), chlorophyll-*a* and flows (62.3%). Annual sedimentation rates of total solids ranged from 11.73 to 16.29 kg m^{-2} $year^{-1}$ with organic matter comprising around 30%. N:P ratio on sedimentation rates were as high as could be expected in a resuspension dominated ecosystem, and spatially inverse related with N:P ratio on bottom sediments. Distance from river inlet into the reservoir reveals a marked spatial heterogeneity on solid and nitrogen sedimentation, showing

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the system dependence on river inflows and supporting resuspension as the main phosphorus pathway. Accretion rates (2.19±0.40 cm year^{-1}) were not related to hydrological variability but decreased with the distance to the river input. Total sediment accumulation (9,895 tons km^{-2} year^{-1}) denotes siltation as other serious environmental problem in reservoirs but possibly not related with operational procedures.

Keywords Nitrogen · Phosphorus · Chlorophyll-*a* · Sedimentation · Reservoir drawdown · Eutrophication · Semi-arid

1 Introduction

Eutrophication (as nutrient enrichment) as a serious environmental problem in lakes and reservoirs has been wide and extensively documented (de Jonge, Elliott, & Orive, 2002; Harper, 1992; Reynolds, 1992; Ryding & Rast, 1989; Skei et al., 2000 among others). In the last decades, most of the aquatic ecosystems of the entire world experienced a gradual eutrophication. Changes in land uses, including deforestation for agricultural or grazing activities, and growing wastewaters proceeding from a growing urban population are the main causes increasing the organic load (nitrogen and phosphorus) in rivers and lakes, modifying the biological responses of watersheds (de Anda, Shear, Maniak, & Riedel, 2001; Harper, 1992).

Spatial and temporal heterogeneity of limnological characteristics in man-made lakes is known to influence the ecological structure and functioning of these ecosystems (Nogueira, Henry, & Maricatto, 1999). Water-level variability changes the ecosystem development, affecting the nutrient flows, metabolism and decomposition and interactions between plants, addressing particularly the responses of the primary producers (Sánchez-Carrillo & Álvarez-Cobelas, 2001; Quintana, Comín, & Moreno-Amich, 1998; Wetzel, 1990), including the phytoplankton succession (Vinçon-Leite, Tassin, & Druart, 2002). Hydrological fluctuations are not significant in lakes and reservoirs of temperate areas while in arid and semi-arid areas both seasonal and annual variability changes the size and shape of the ecosystem which modifies the dependence from external to internal inputs, increasing the importance of resuspension of the accumulated nutrients in the sediments (i.e., internal load; Håkanson, 2004). In reservoirs, the requirement to store water in order to supply water demands and the operational procedures (i.e., management practices) to fulfil needs, isncreases temporal water level fluctuations (Straškraba & Tundisi, 1999). In semi-arid regions, water level rises when water surplus and it goes down when water is used without replenishment in the period of scarcity (Naselli-Flores, 2003). In this way, the ecological and dynamic relationships of the semi-arid reservoirs can be characteristic, inside only one cycle, of a deep thermal-stratified ecosystem to other extremely shallow, varying the capacity of nutrients recycled by the bacteria (Wetzel, 1990), modifying even the natural conditions of thermal stratification and the exchange and mixing processes (Arfi, 2003), and enhancing eutrophication (Naselli-Flores, 2003). Inflow and outflow reduction also increases the trophic status of lakes by means of retention time increments (Volohonsky, Shaham, & Gophen, 1992).

In tropical and subtropical lakes, trophic dynamics – relationships between nutrient limitation and phytoplankton biomass – is not as simple and linear as in temperate systems (Huszar, Caraco, Roland, & Cole, 2006). Furthermore, in those eutrophic shallow ecosystems, anoxia and warm hipolimnetic temperature can contribute to increase phosphorus releases significantly from the sediment–water interface (Ruley & Rush, 2004; Townsend, 1999). When no water inputs, phosphorus load in the ecosystem is maintained by means of increasing resuspension through different mechanism when water level descends (Eckert, Didenko, Uri, & Eldar, 2003; Volohonsky et al., 1992). However, little is known yet on the trophic effects of water drawdown and refilling in reservoirs, even the subtropical ones, particularly when it occurs in a single year. Literature reports how reservoir refilling cause an increase of nutrient loads (Scharf, 2002). When a drawdown occurs, larges areas are exposed to desiccation, organic mater oxidation and denitrification, including growth of terrestrial vegetation (Geraldes & Boavida, 1999). Refilling releases nutrients inducing a trophic upsurge in the reservoir. Except Naselli-Flores' (2003) study in a Sicilian reservoir, it is not well documented how quick changes, from drawdown to refilling of water bodies, impact on nutrient dynamics and sedimentation, including primary producer responses.

In this paper we present the results of a quick ~9 m drawdown followed by a partial refilling occurring during 1 year in a large reservoir located in the Nortwestern Mexico. Aims of this study were to observe the effects on nutrient, sediment and primary producers (as chlorophyll-*a*) in order to characterize the hydroecological impacts of water-level variation in warm semi-arid subtropical lakes and reservoirs due to the operational procedures.

2 Study Site

This study was carried out in the Álvaro Obregón reservoir, located in Northwestern Mexico (27°50′N and 109°53′W; Figure 1), within the Yaqui River basin (ca. 75,000 km^2). The reservoir, situated at the outlet of a 19,292 km^2 watershed, operates since 1952 receiving inflows from two sources: discharges from an upstream reservoir (in the dry season – winter to spring) and direct inflows from watershed runoff. Maximum water depth is 48.4 m (mean 28.2 m) when water storage reaches 3,227 hm^3 with an inundation area of 164.6 km^2 (see Table I). Relationship between water-level and inundation is close to 1:1. Morphometrically, the Alvaro Obregón reservoir is NE–SW elongated, conditioning the water flow circulation pattern. For the period 1964–2003, the National

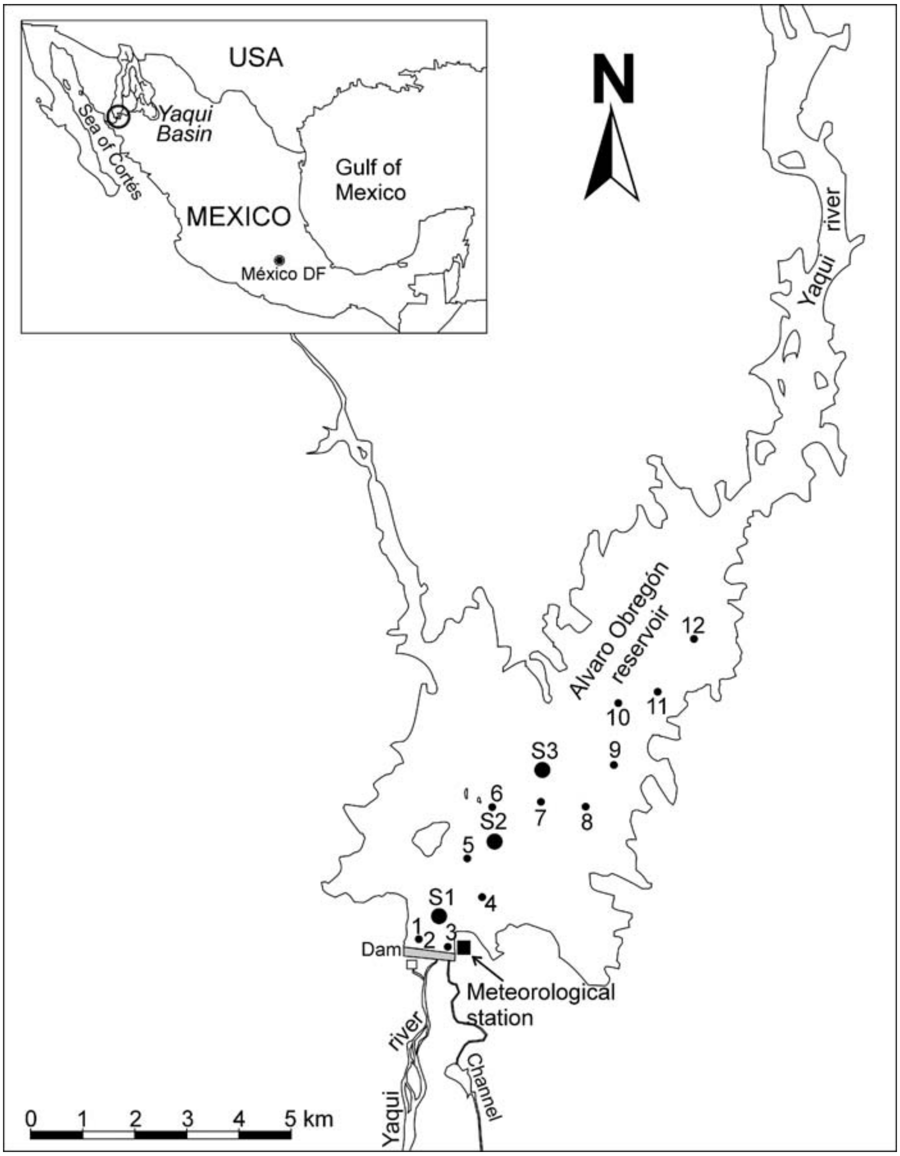

Figure 1 Location of the Alvaro Obregón reservoir. *S1*, *S2* and *S3* are the sampling sites for water and sediment traps.

Table I Morphometric characteristics in the Alvaro Obregón reservoir

Maximum area (*A*)	106.03 km^2
Maximum water storage (*V*)	2,989.17 hm^3
Mean depth (*z*)	28.19 m
Maximum depth (Zm)	105 m
Relative depth ratio (Zratio)	0.90 m
Maximum length (*l*)	13.56 km
Maximum effective length (Lf)	13.56 km
Maximum width (*b*)	5.84 km
Shoreline length (*L*)	109.12 km
Shoreline development (D_L)	0.000055
Shoreline development factor (*D*)	4.23
Average lake slope (*S*)	0.32
Volume development (Dv)	0.81
Dinamic sediment ratio (DSR)	365.27

Water Commission (CNA) estimates an average annual inflow of around 3,504 10^6 m^3 $year^{-1}$, even though the annual variability is very extreme (from 1,106 to 7,113 10^6 m^3 $year^{-1}$). Reservoir provides water supply mainly to a 250,000 ha irrigation area of intensive agriculture. Upstream of the reservoir, population density is low (<10 inhabitant km^{-2}) with non-mechanized (nor intensive) agriculture and cattle comprising the main economic activities. No industries are located in the area (INEGI, 2001).

The clime is semi-arid consisting of a warm, rainy monsoonal season from June to September, followed by a cool season from winter to the spring. Mean annual rainfall is 317 mm although very irregular year to year. Mean annual temperature is about 23.4°C. Average monthly temperatures are correlated to potential evaporation rates, which are normally higher than average monthly precipitation. Vegetation cover is composed dominantly by shrubs and deciduous broadleaf forests and some patches of pasturelands. Land cover in the Yaqui river basin is changing drastically in the last decades; for example, over the 20 years period, from 1973 to 1992, the forest areas were reduced in more than 52,000 ha (Navarro, 2003).

3 Materials and Methods

Sampling was conducted monthly from January to December of 2003 at the deepest area of the reservoir (S2, Figure 1). At the same time, temperature and oxygen profiles were carried out each month at 0.5 m intervals using an oxymeter YSI MOD 58, between 11:00 and 12:00 h. Water samples were collected from the surface (0.5 m), half depth (≈4–5 m), and near the bottom (≈10–14 m), using a Niskin bottle (according to temperature variation, i.e., thermocline). Occasionally, when differences on temperature profile were most marked and water depth increased, an additional water sample was taken at 8 m. Water samples were stored in acid-washed 2 l polyethylene bottles. The basic set-up for the sedimentation study consisted on cylindrical sediment traps (diameter–height ratio 1:10; according to Bloesch, 1996). Two traps were submerged vertically at 1 m depth in three sampling points (S1, at the end of the reservoir, close to the dam; S2, in the middle area of the reservoir, around 3 km before the dam; and S3, close to the inflow area, 5 km before the dam; Figure 1) and deployed every 30 days. No conservative substances were used in sediment traps, which were filled using distilled water. All samples were stored in the dark and conserved below 4°C during the trip to the laboratory where they were analyzed immediately for (APHA, 1998): total solids (drying to 110°C), organic matter and mineral matter (loss on ignition to 550°C), total organic carbon (calculated assuming that 50% of total organic matter is organic carbon; Serruya, 1976), total nitrogen (through Kjeldahl Nitrogen method) and total phosphorous (by the sulphomolybdate). Phytoplanktonic chlorophyll-*a* was measured spectrophotometrically after methanol extraction (Marker, Nusch, Rai, &, Riemann, 1980).

Finally, at the end of the observation period a bottom sediment sampling was conducted throughout permanent flooded areas of the reservoir (n= 14) analyzing water content (110°C), bulk density (paraffin method, Howard & Singer, 1981), organic matter content (according to Walkley–Black method; Sparks, 2002), and total nitrogen and phosphorus (following Kjeldahl and Bray methods, respectively; Sparks, 2002) in order to estimate relationships between sediment rates, accretion rates and nutrient dynamics.

Meteorological data (rainfall and air temperatures) were obtained from an automatic weather station situated close to the dam (Figure 1). Data on inflows, outflows, water level and evaporation were provided by National Water Commission (CNA) staff.

All statistical analyses were carried out using the Statistica v6.0 (Statsoft Inc., 2001). The Shapiro–Wilk W test was used in testing for normality. Simple regression and Spearman rank order non-parametric correlations were used to find relationships between variables. Non-parametric Wilcoxon paired test was used to find significant differences between sediment sampling points. Finally, multivariate exploratory techniques were performed in order to elucidate seasonal groups (cluster analysis) and control variables in the observed temporal variability (principal components) using correlation coefficients between all variables, in order to reduce magnitude differences.

4 Results

4.1 Meteorological and hydrological settings

During the observation period, air temperature oscillated from 43.1°C in summer (June) to −1.5°C in winter (December). Mean differences between day and night temperatures were around 20°C (32.6±5.3 and 14.2±7.5°C, respectively). Rainfall patterns was related to the North American Monsoon (June–September), accounting for 87% of the total precipitations (349 mm year^{-1}, Figure 2). Isolated minor storms appeared in winter and spring (Figure 2). Frequently, summer monsoonic storms occurred from 13:00 to 18:00 h (>75%). Wind blew dominantly from 181 to 225°, which corresponded to a fetch close to 2,800 m. Maximum wind velocity was up to 35 m s^{-1} (average 3.2±2.1 m s^{-1}) occurring at midday from 10:00 to 15:00.

Figure 2 shows the reservoir water balance during the study period. Inflows and outflows were just restricted to the first months of the year responding to the water demands from agriculture which forced the water managers to release water from the reservoir located upstream the Alvaro Obregón dam. Water-level drop from January to June with its highest rate at 1.4 m day^{-1}, recovering slightly after the monsoonal cycle (June–September) and lowering again until the end of the observation period. The annual variation on water level ranged 5 m (mean water depth in the deepest zone 22.8±2.5 m; Figure 2). The mean water retention time during 2003 was 1.05±0.90 year^{-1} and the mean hydraulic loading was 1.70±1.80 m month^{-1}. For the

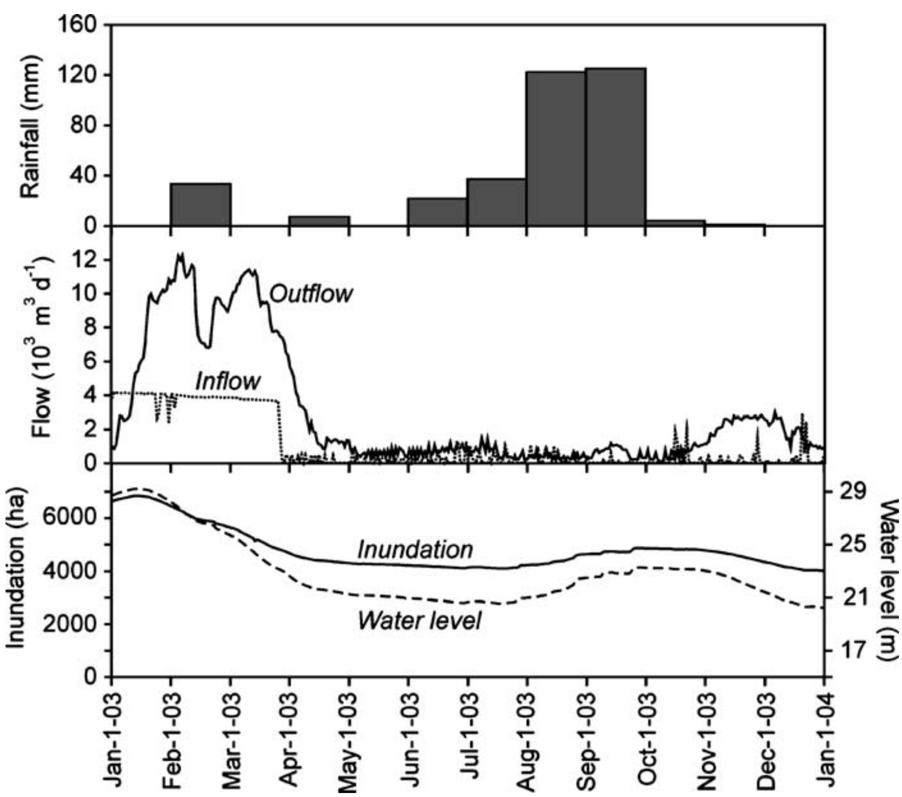

Figure 2 Meteological and hydrological conditions in the Alvaro Obregón reservoir during 2003.

entire period average water stored in the reservoir was 498.6±204.82 hm^3.

4.2 Oxygen and thermal variability

From May to June a thermal stratification was observed with a thermocline found from 5 to 9 m deep (Figure 3). Differences on water temperature between surface and bottom during this period were around 10°C (Figure 3). Water circulation began in September–October throughout April. An oxycline was detected between 8 and 9 m (Figure 3) coinciding with the thermocline depth. Below 9 m deep there was no oxygen during most of the year due to thermal stratification resulting in the imbalance between the organic matter oxidation and oxygen supply from surface waters (Figure 3). Oxygen concentration at the whole water profile increased in winter due to the water circulation. However, the highest values were registered in the surface from May to June (≈8 mg l^{-1}) but were not related with maxima of phytoplankton biomass neither with water inflows. Wind gusts were frequent at midday during summer months because of the stormy weather of the moonsonic period (see Section 4.1). It possibly could tend to induce an intensive atmospheric exchange due to the increased waves.

4.3 Solids, organic matter, nutrients and chlorophill-*a* dynamics

Total solids ranged from 126 to 340 mg l^{-1} (Figures 4 and 5), with its highest concentrations at the surface waters until June, when this behavior reversed to the bottom of the reservoir (Figure 5). In February appeared the highest concentrations for the whole study period (Figures 4 and 5). Average mineral matter in the water profile was 186.8±64.5 mg l^{-1}

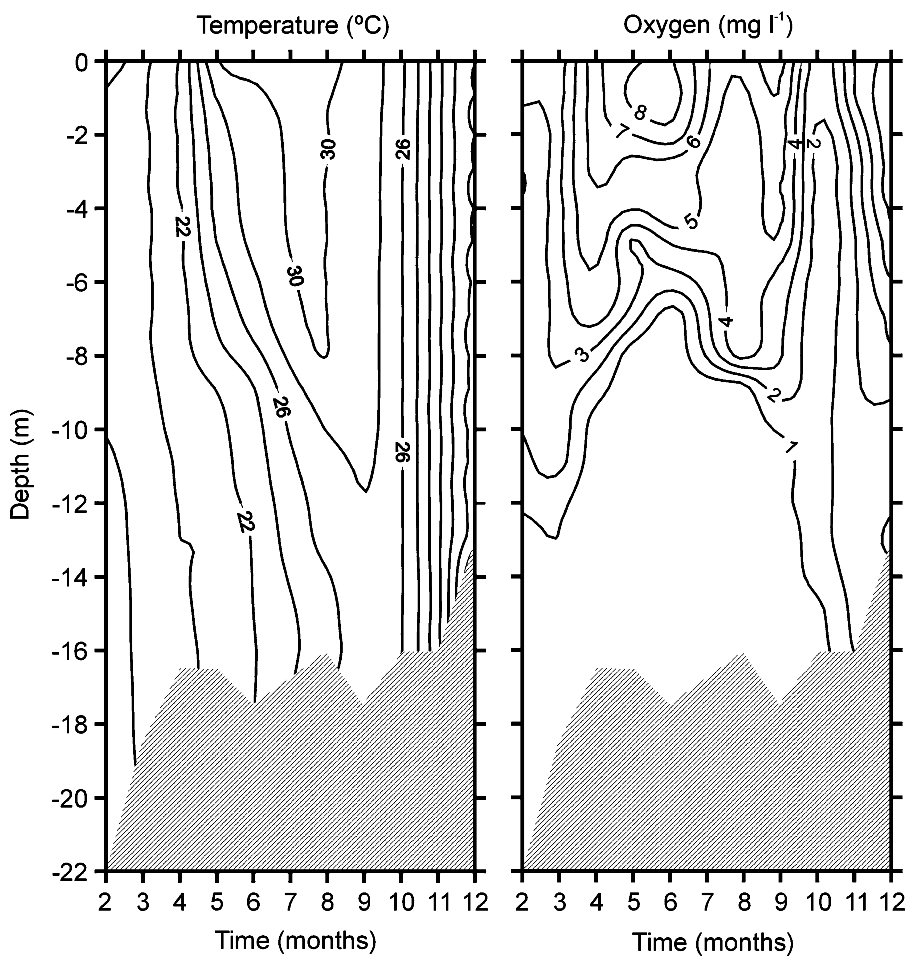

Figure 3 Temperature (°C) and oxygen (mg l^{-1}) isoplets at the Alvaro Obregón reservoir during the study period. *Grey areas* are bottom sediments indicating water level temporal variation at S2 sampling point.

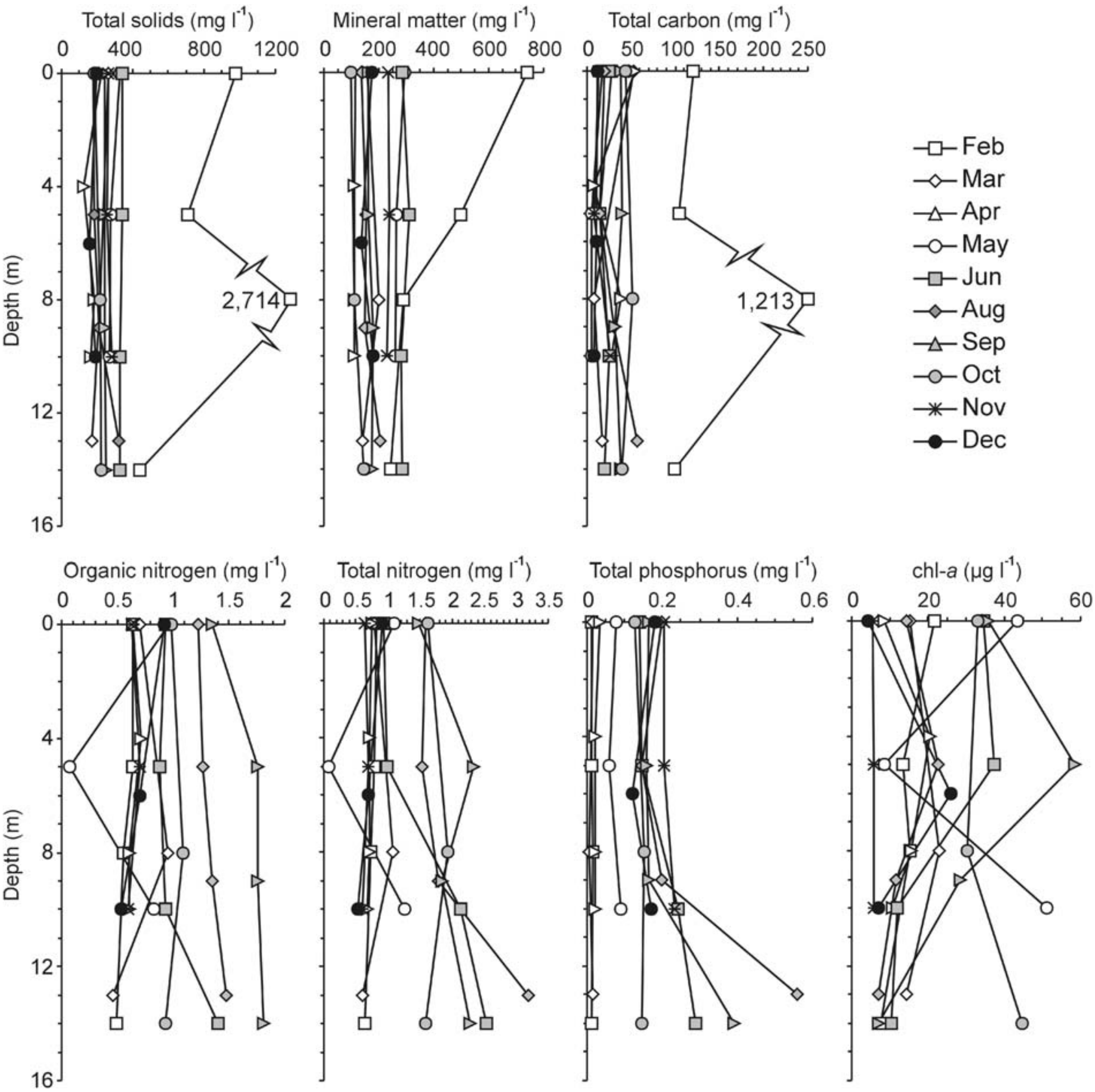

Figure 4 Time course of water content on total solids, mineral matter, organic matter, total carbon, organic nitrogen, total nitrogen, total phosphorus and chlorophyll-*a* (chl-*a*) during 2003. Sampling site corresponds to S2.

(Figure 5). Vertical gradients were negligible during spring, while from June to October maxima occurred in the deepest area (Figures 4 and 5). To the end of summer and the early winter, vertical gradient returned to be negligible, changing afterwards to a maximum at the surface (Figure 4). Both variables (solids and mineral matter) showed, graphically, an inverse relationship with water-level, coinciding the highest concentration with the periods of stratification and the lowest ones with the mixing periods (Figure 5). However, a moderately strong relationship between total solids and water level and inflows could be asserted (R^2=0.586 and 0.438, respectively, at $p<$ 0.05). Organic matter and total carbon appeared to have a changing vertical gradient during the whole study period: during February, May, August, and November it registered positive gradients with respect to depth with maxima near the bottom joined usually

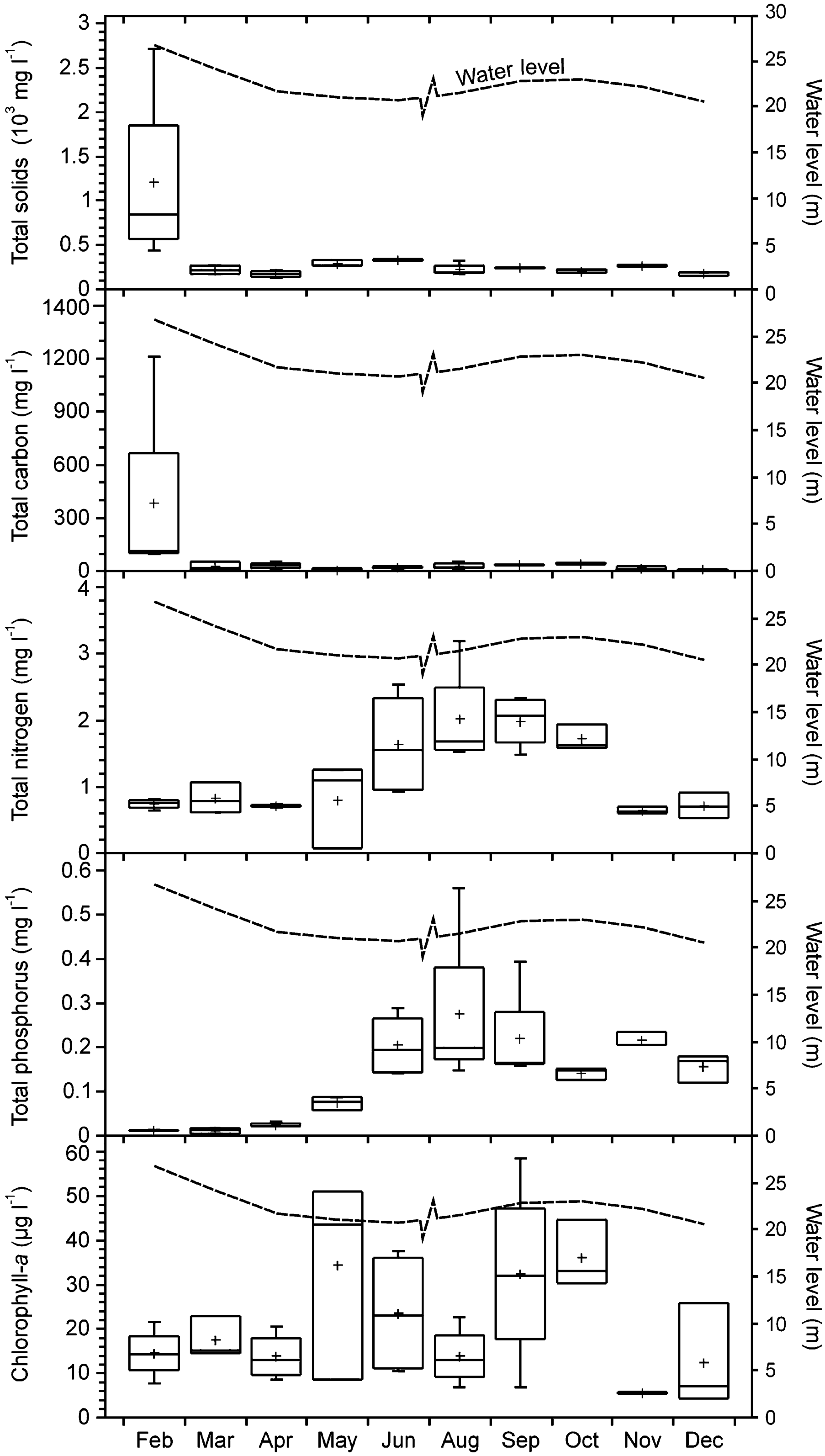

Figure 5 Box and whisker plots on total solids, total carbon, total nitrogen, total phosphorus and chlorophyll-*a* (chl-*a*) at S2 sampling site during 2003.

by a decreasing concentration in the half depth (Figure 4). Anyway, for the entire year, total organic matter was higher at the surface although more variable (287.0±752.1 and 89.8±71.6 mg l^{-1}, in the surface and the deepest water, respectively), while total carbon showed highest concentration close to the bottom (averaging 39.9±32.4, 26.6±31.5, 143.5±376.1 mg l^{-1}, at surface, 5 and 12 m depth, respectively). In a temporal basis, time courses of mean concentrations at the water profile for organic matter and total carbon concurred with the water level, with minima in May and maxima in September–October (Figure 5), and were explained significantly by water level (Spearman non-parametric correlations R=0.7212 for organic matter and R=0.7333 for total carbon at p<0.02). This pattern of concentration concurred also with the start of either, stratification or circulation periods (Figure 5). Inflows were related also with raises on organic matter concentrations in the reservoir (R^2=0.4625 at p=0.03), moderately though (for example, maxim occurred in September when there weren't any inflow).

During the sampling period total nitrogen ranged from 0.07 to 3.19 mg N l^{-1}, increasing with depth (Figures 4 and 5). At surface, values increased up to a maximum in August, September and October, while in the deepest area the maximum appeared before (from May to September, Figures 4 and 5). Seasonal trend of total nitrogen at the water column pointed out high concentration from May to November with maximum values during August, September and October that could be related to the beginning of the circulation period. (Figure 5). This pattern was dependent on organic nitrogen fraction, which comprised up to 58% of the total nitrogen (up to 100% at the end of the study period). Maximum concentration of organic nitrogen occurred in September, concurring with a felt on dissolved inorganic nitrogen. Exclusively, the nitrogen inorganic fraction caused the maximum observed on total nitrogen in June and August at 8 m depth (Figure 4). Averaging, minimal variation on organic nitrogen with depth were observed (0.90±0.25, 0.88±0.45 and 0.86±0.42 mg N l^{-1}, in surface, 4–8 and 12 m, respectively). Water level and nitrogen content relationship seemed to be inverse with lowest values of nitrogen registered when water level rose and maximum concentrations after minimum levels (Figure 5). However, no statistically significant relationships could be asserted (p>0.05). Only dissolved inorganic nitrogen exhibited an inverse significant correlation with outflows (R=−0.63, p=0.047). This could be related to the water management effects on the nitrogen by means of residence times since concentration increased in the reservoir when no inflows neither outflows were registered.

Seasonal trend on total phosphorus was clearer than those on total nitrogen. Increases, up to 10 times, in the phosphorus concentration for the whole water profile could be observed toward the end of year with highest values from June to September (Figures 4 and 5). Despite phosphorus presented a graphical inverse association with water level, a statistically significant relationship could not be established. Nevertheless, a moderately strong relationship between total phosphorus concentration and outflow (Spearman correlation, R=−0.6485 at p=0.043) was established. At the surface and deepest zones, two peaks appeared in August and November–December, while in the middle depth just one peak, in November, was registered (Figure 4). In average, total phosphorus concentration increased slightly with depth (0.12±0.08, 0.10±0.07 y 0.13±0.09 mg P l^{-1}; at the surface, 4–8 and 12 m, respectively).

Table II shows monthly N:P and C:N average ratios. Phosphorus supply/availability increased throughout the entire study period in both surface and deep water. N:P ratio declined over time following reciprocal-x functions as: $N:P_{surface}=-9.8642+152.341/month$ (R^2=0.889 at p<0.01) and

Table II Montly averages on N:P and C:N ratios at surface (0.5 m) and deep water (≈10–12 m) in the Alvaro Obregón reservoir during 2003

	N:P ratio		C:N ratio	
	Surface	Deep	Surface	Deep
February	63.1	58.2	147.0	154.4
March	56.4	35.9	67.1	28.4
April	22.4	29.6	72.6	40.8
May	14.3	14.0	15.9	4.0
June	6.6	8.8	29.6	7.9
August	7.9	5.7	12.7	18.0
September	8.9	5.8	25.9	16.8
October	12.9	10.7	26.6	25.3
November	3.1	2.6	21.2	44.5
December	5.1	3.1	12.7	15.7

$N:P_{deep}$=−8.81681+133.939/month (R^2=0.964 at p< 0.01). Spearmank correlation evidenced a significant relationship between N:P ratio at the surface and water-level (R=0.6364 at p=0.048). Mineralization – measured as C:N ratio – increased throughout the year following reciprocal-x functions ($C:N_{surface}$= −14.9301+297.039/month, R^2=0.8839 at p<0.01; $C:N_{deep}$=−14.7673+256.42/month, R^2=0.617 at p< 0.01). However, C:N ratios on surface and deep water did not have the same trend: while in both initially a strong decline occurred until May, at surface it continued through August increasing slightly before low value in December (Table II). On the contrary, at deep water C:N ratios increased until November (Table II), following closely the water level time course. This was sustained by a significant relationship between C:N ratio at deep water and water level (Spearman correlation R=0.7333 at p=0.016) and more strongly with outflows (Spearman correlation R =0.7576 at p=0.011). C:N ratio at surface also resulted significantly correlated with outflows (Spearman correlation R=0.6364 at p=0.048) but not with water level.

The response of primary producers (as chlorophyll-a) were very irregular during the entire study period (Figure 4). Maxima values appeared in May and September–October concurring with the beginning of the stratification and circulation periods, respectively. On the contrary, at 4–8 m depth this pattern was rather more variable although increasing chl-a toward highest concentration in October (Figure 5). This middle depth zone registered the highest values in average (21.68±14.15, 24.59±15.28 y 20.60± 15.69 μg l^{-1}, at surface, 4–8 y 12 m) and during most part of the year (Figure 5). Chlorophyll-a was not explained by the hydrological variability (p> 0.05). A significant relationship could be observed using monthly average values between dissolved

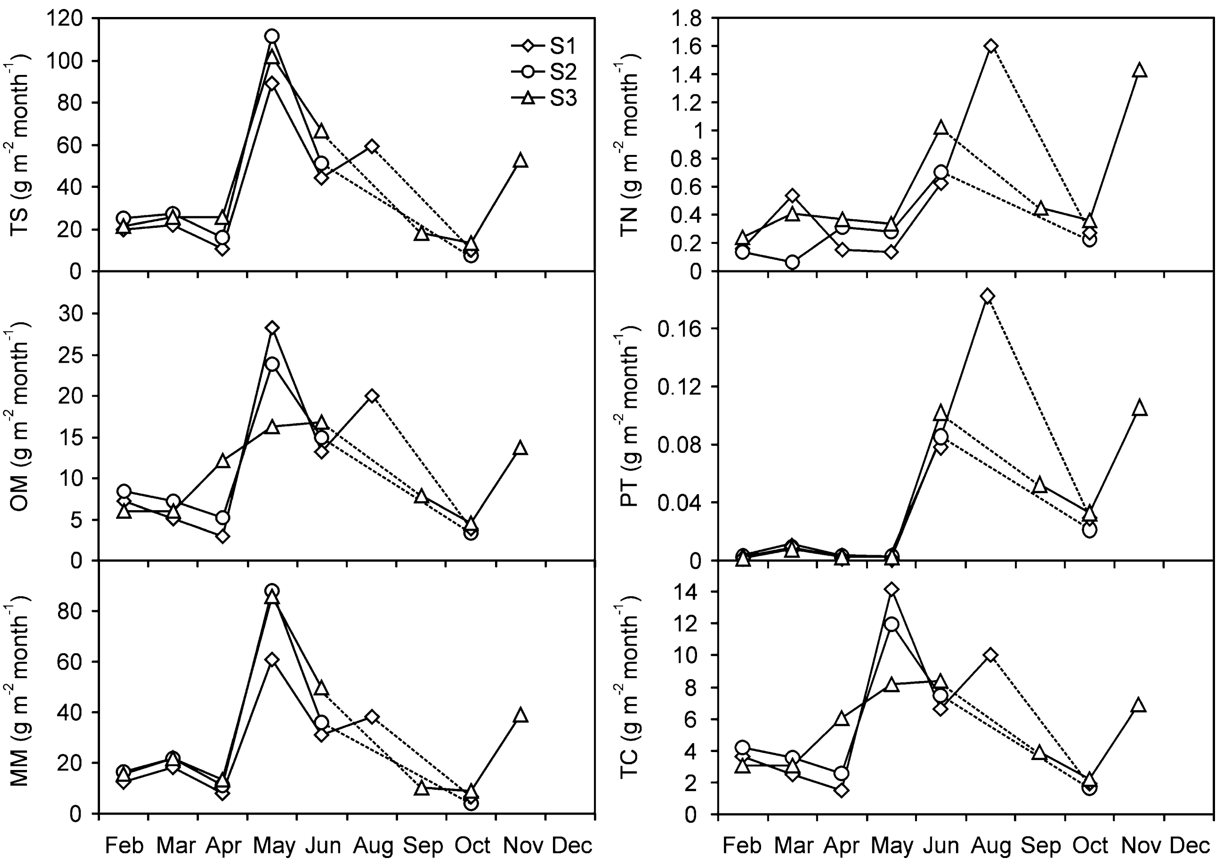

Figure 6 Trends on sedimentation rates of total solids (*TS*), mineral matter (*MM*), organic matter (*OM*), total carbon (*TC*), total nitrogen (*TN*), total phosphorus (*TP*) during 2003 in S1, S2 and S3 sampling points.

inorganic nitrogen, total nitrogen and chl-*a* (Spearman correlations: $R=0.7538$ at $p=0.0118$ and $R=0.6364$ at $p=0.0479$, respectively). Finally, no relationship appeared between the spring concentration on phosphorus and nitrogen and the summer chl-*a* concentrations, neither using whatever log TP, log TN or N:P ratios.

A cluster analysis using all data variables revealed significant differences on concentrations registered in February (mean Euclidean distance=1,286). May, June and November were related as months with low water level whilst October and April on a group and December, September, August and March in another one had not significant differences (Euclidian distances=13–88). However no causal hidrologically based relationships could be asserted. A principal component analysis exposed three factors explaining 98.65% of the seasonal variability on all measured variables. First factor was comprised by total solids, mineral and organic matter, total carbon, chl-*a*, inflows and outflows (explained variance=62.33%); nitrogen and phosphorus composed the second factor (explained cumulative variance=90.23%), and the other factor was explained by water-level.

4.4 Sedimentation

Throughout the study period seasonal patterns on total solids, mineral and organic matter sedimentation rates were similar in all sampling sites (Figure 6): low rates during the early year months, increasing in May and June – coinciding with the start of stratification, decreasing thereafter until the final of the rainy period (October) which was joined to the beginning of the circulation one. On average sedimentation rates were slightly higher in site S3 for total solids (40.69±30.84 g m^{-2} day^{-1}), while in site S1 for organic matter and total carbon sedimentation (11.50±9.57 g m^{-2} day^{-1}). Statistically, total solids sedimentation was significant different in the three sampling points (Wilcoxon test $p<0.05$) while mineral matter differential sedimentation only appeared at S1 ($p<0.05$). Annual total solids sedimentation rates ranged from 11.73 kg m^{-2} $year^{-1}$ in S1 to 16.29 kg m^{-2} $year^{-1}$ in S3, with organic matter fraction comprising from 32% (S1) to 25% (S3).

Total nitrogen sedimentation also concurred in all sampling sites with maximum in June (August in S1, located close to the dam; Figure 6) when water level was lowest. Organic nitrogen sedimentation comprised in average 84% of total nitrogen. Annual averages of nitrogen sedimentation were highest in S3 (0.58±0.53 g m^{-2} day^{-1}), followed by S1 and S2 (0.50±0.53 and 0.29±0.22 g m^{-2} day^{-1}, respectively). Total phosphorus sedimentation followed a same seasonal trend with lowest rates until the start of stratification, rising after in June–August, decreasing again until the start of the circulation period (October; Figure 6). Annual average of phosphorus sedimentation was similar on S1 and S3 (0.04±0.07 and 0.04±0.04 g m^{-2} day^{-1}, respectively), and lower in S2 (0.02±0.03 g m^{-2} day^{-1}). However, nutrient (nitrogen and phosphorus) sedimentation revealed a homogeneous spatial distribution since no significant differences were observed between sampling stations (Wilcoxon test $p>0.1$). Nitrogen sedimentation and nitrogen in the water column was related significantly at S1 sampling point (Spearman correlation $R=0.7857$ at $p=0.036$), while total phosphorus sedimentation rates at S3 demonstrated only a possibly casual relationship with total

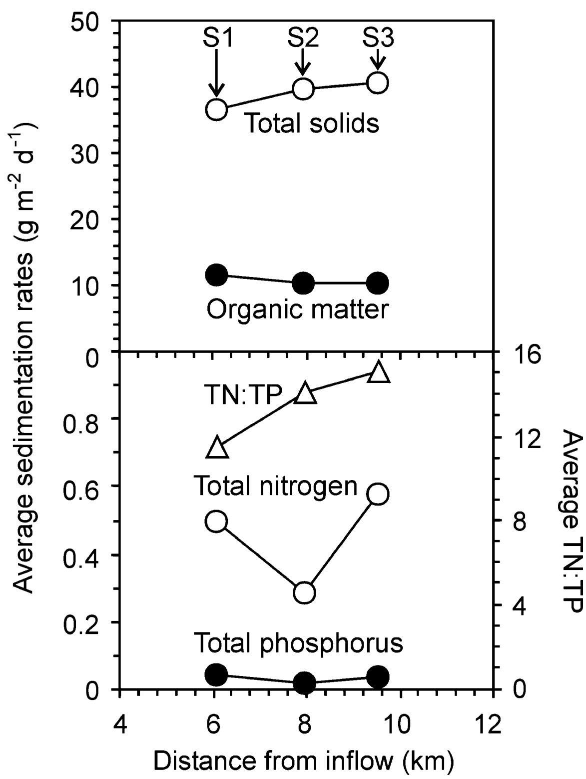

Figure 7 Relationships between average sedimentation rates on total solids, organic matter and mineral matter as well as total nitrogen, total phosphorus and N:P and the distance to the river inflow into the Alvaro Obregon reservoir.

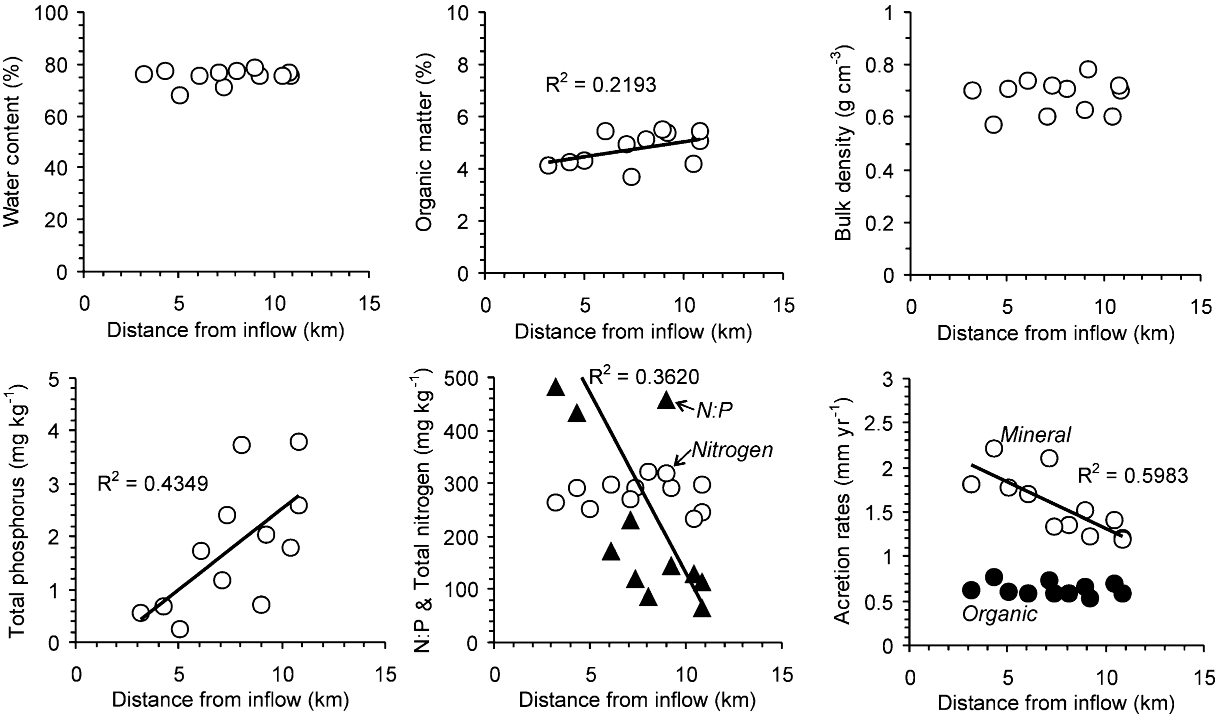

Figure 8 Distance from river inlet into the Alvaro Obregon reservoir as spatial condition of heterogeneity on bottom sediments for: water content, organic matter, bulk density, total phosphorus, total nitrogen, N:P ratios, and mineral and organic accretion rates (cm year^{-1}).

phosphorus contained in the water profile at S1 (R=0.7619 at p=0.028). Just organic – and total carbon – and total nitrogen sedimentations at S2 were conditioned by the reservoir hydrological management (i.e., inflows–organic matter; Spearman correlation R =0.8214 at p=0.023; water-level-nitrogen: Spearman correlation R=−0.8857 at p=0.019), althoug inversely. Chlorophyll-*a* concentration did not significantly explained any sedimentation rates (i.e., organic, carbon or nutrient). In an average annual basis, at least graphically, total solid sedimentation rates appeared conditioned by the distance from the river inflow into the reservoir (Figure 7). Inversely, organic matter sedimentation increased with this distance, displaying, as above mentioned, relationship with inflows occurring not by means of organic inputs proceeding from the watershed but in an indirect way. Figure 7 also shows how nutrient sedimentation does not have a clear spatial pattern respect to the distance from the river outlet. Possibly, the fall on sedimentation rates at S2 were responses to its maximum water depth, considering the significant relatioship between nitrogen sedimentation and water level. However, total nitrogen:total phosphorus sedimentation ratios depicted a clear positive dependence on river inflow distance (Figure 7). Data from bottom sediments demonstrated these findings (Figure 8): phosphorus (not nitrogen) like organic sediment content resulted significantly conditioned by the distance from the inflows. Nitrogen sediment content averaged 2.81± 0.28 g kg^{-1}, with the highest values close to S2, where paradoxically the lowest sedimentation rates were registered (Figure 8), while mean phosphorus sediment content was 1.78±1.19 mg kg^{-1}. N:P ratios of sediments also exihited a inverse dependence on the river discharge (Figure 8; average N:P=292.96± 290.56). A physical property of sediment such as bulk density (mean±SD=0.68±0.07 g cm^{-3}), which could be spatially affected by the sediment river differential accumulation into the reservoir, resulted not statistically conditioned (Figure 8). Finally, showing its dependence on river inflows, mineral matter accretion rates decreased significantly towards the dam (Figure 8).

The amount of sediments deposited during this year was 0.014 ton m^{-2} year^{-1}, from which 79% was mineral matter, 20% organic matter, 0.9% total nitrogen and 0.1% total phosphorus. Mean accretion rate in the reservoir was 2.19±0.40 cm year^{-1}, with around 30% corresponding to organic sediments (0.63±0.07 cm year^{-1}). According to the mean inundated area for this period (4,793 ha) a volume around 1.05 hm^3 of sediments were accumulated (0.31 hm^3 of organic sediments).

5 Discussion

Following Hutchinson (1957) thermal lake classification, the Alvaro Obregon reservoir is a warm monomictic lake with summer stratification and winter circulation. In spite of scarce inflows registered during 2003, meteorologically, this period can be considered as normal for the Northwestern Mexico (Gochis, Leal, Shuttleworth, Watts, & Garatuza-Payan, 2003). As pointed out by Naselli-Flores (2003), a disruption in stratification duration occurs when system changes into a shallow water body. Then thermocline could be expected temporally extended during high inundation episodes – when depth becomes close to 50 m. This could be considered as the first effect of water management.

The second effect is an enhanced eutrophication as water level goes down. According to the OECD standards (Ryding & Rast 1989), reservoir changes from mesotrophic to eutrophic (as averages on total phosphorus and total nitrogen) or from eutrophic to hypertrophic (based on chlorophyll-*a* concentrations), after reducing the trophic status when refilling. Hence, the progressive and conspicuous depth decrease would interfere with nutrient and phytoplankton dynamics (Naselli-Flores, 2003). Barica and Mur (1980), found that the occurrence of hypertrophy in subtropical lakes is due to the shallowness, to the limited water circulation, to the imbalanced nutrients and to the anoxic conditions, similar to those observed during our water drawdown.

Water level regulates seston (i.e., suspended particulate matter) content in the water column. Mineral matter, which would be expected to depend just on river inflows, increases as water depth drops as well as after refilling. This could be explained through a strong dynamic relationship with the beginning of both stratification and circulation periods, but possibly it responds to an increasing concentration because water drop is more rapid than particle settling velocities. Anyway, it could also respond to an enhanced sediment resuspension when shallowness.

Outflows exert a strong control on nutrient dynamics. Dissolved inorganic nitrogen and total phosphorus increases when no outflows occur. This operational variable regulates then nutrient availability at the system through water retention time. Unlike most worldwide reservoirs, in the Alvaro Obregón high nutrient loadings are not expected to come from the watershed because its land uses and socioeconomic characteristics (see the study site section; INEGI, 2001). Whilst nitrogen concentrations at the beginning and the end of the observation period are within the same range, phosphorus increases significantly. Traditionally, nitrogen limitation is considered more common in tropical than in temperate systems (Downing & MacCauley, 1992; Lewis, 2000), possibly due to greater P supply by chemical rock weathering and greater internal N loss at higher temperatures (Lewis, 2002, Settacharnwit, Buckney, & Lim, 2003). In our system, P – also N – inputs from external sources were restricted to sporadic episodes. Besides, increases on P content were not correlated to inflows. When allocthonous inputs drop, phosphorus concentration in an ecosystem must be maintained by means of increasing resuspension – internal loading – through low water level (Eckert et al., 2003), controlling then strongly the phosphorus cycle (Ruley & Rush, 2004). A large mechanism of phosphorus resuspension can be expected in an ecosystem, as cited by different authors (e.g., Gin, Zhang, Chan, & Chou, 2001; Håkanson, 2004; Naghavi & Adrian, 1993; Peeters, Piepke, & Gloor, 1997; Petterson, 1998; Ramm & Scheps, 1997; Reddy, Fisher, & Ivanoff, 1996; Reddy, O'Connor, & Schelske, 1999; Ruley & Rusch, 2004; Sondergaard, Jensen, & Jeppesen, 1999). In shallow eutrophic lakes, internal loading can provide phosphorus concentrations up to 20 times greater that of external loading (Krogerus & Ekholm, 2003). Townsend (1999) reported, in a dry tropic Australian reservoir, how anoxia and warm hipolimnetic temperature can contribute to increase phosphorus releases significantly from the sediment–water interface, depending on phosphorus sediment concentration. However, we cannot expect any phosphorus release from sediments since P sediment concentration at Alvaro Obregon reservoir is lower than those reported in the former study for a temperate reservoir which did not experience resuspension.

Otherwise, N:P ratios at surface as well as deep waters confirms the increase of P availability throughout the study. Ratio is controlled at least in surface by water level, which could confirm, indirectly even though, the occurrence of phosphorus resuspension. Likewise, phosphorus resuspension could be responsible of the lack of statistical significance between phosphorus water content and phosphorus sedimentation inasmuch as nitrogen sedimentation responded to that correlation. Possibly, it would explain also the absence of control by hydrology on phosphorus sedimentation (nitrogen was related inversely to water level). Our TN:TP ratio from trapped sediments (average 13.5±1.8) is close to those reported by P. Nõges, Järvet, Tuvikene, and T. Nõges (1998) in the shallow eutrophic Lake Võrtsjärv in Estonia (≈12), where resuspension accounts for around 50% of gross nitrogen sedimentation and only 20% is buried in the sediments (denitrification=80%). In contrast, TN:TP ratio from bottom sediments in the Alvaro Obregon reservoir was high (293±291), being able to note a most effective phosphorus resuspension or a low denitrification rate. Nevertheless, more studies addressed to this complex topic are needed to support our indirect remarks in order to confirm the role of water management on resuspension and P cycling.

Water level decreasing exposes larges areas to desiccation increasing organic mater oxidation and denitrification (Geraldes & Boavida, 1999). C:N ratios confirm how mineralization is enhanced by water drawdown mainly at deep waters.

The lack of a seasonal pattern of primary producers is a common characteristic in tropical and subtropical lakes (Lewis, 2000). In these regions, the more favorable growing conditions during most part of the year leads to a higher metabolism level (i.e., high productivity; Thornton, 1980). Our chl-*a* data shows a closed relationship with the beginning of both thermal stratification and mixing. Although Huszar et al. (2006) using data from 192 lakes from tropical and subtropical regions rejects the view that nitrogen limitation is necessarily the norm in these ecosystems, considering our peculiar conditions, nitrogen (total and dissolved inorganic nitrogen) seem to be confirmed as the limiting nutrient at this warm subtropical reservoir. As observed by Naselli-Flores (2003), a nutrient depletion in the epilimnion due to phytoplankton uptake did not take place, likely because the increasing internal loading by means of circulation induced by low water level. Phytoplankton growth should be enhanced by the continuous supply of nutrients in summer period. However, the fall on chl-*a* concentration concurs with high nutrient availability. It could respond likely most to light limitation as biomass increases more than zooplankton grazing (Huszar et al., 2006; Naselli-Flores, 2003). Even so, since nutrients availability is controlled by outflows and they were negligible, the former rather appears not as main limiting of primary producers. Meteorology also could exert a limiting factor as reported by the study of Vinçon-Leite et al. (2002), who evidenced how short meteorological extreme episodes can affect significantly the plankton communities, including successions.

Despite possible internal loading influences on nutrient and primary producer dynamics, a strong spatial zonation within the ecosystem is marked by the river inflows. Solids sedimentation as well as TN:TP ratio on settling seston are clearly influenced by river discharges when they occur as reported in other riverine-dependent ecosystems (Reddy, DeLaune, DeBusk, & Koch, 1993, Sánchez-Carrillo, Alvarez-Cobelas, & Angeler, 2001). This spatial pattern results clearer in bottom sediments concentration on TP and TN:TP ratio. Both relationships with the distance from inflow suggests a less important source of phosphorus from watershed by weathering (Lewis, 2000) as could be expect during humid periods. TN:TP ratios on settling seston and bottom sediments respect to the distance from inflow concurs in a inversely way: where highest phosphorus settling, the lowest phosphorus in sediments appear. This finding again could confirm the importance of resuspension on P cycling in the ecosystem.

Sedimentation rates (namely accretion rates) were high during the study period, highlighting a serious siltation problem as water storage is reduced to a rate around 10^6 m^3 $year^{-1}$. While mineral matter accretion reveals an obvious dependence on river inflows, organic matter sedimentation shows a strong dependence on water inputs and strangely not on primary productivity. It concurs on sedimentation and possibly is independent of water operational procedures.

The annual cycle of water drawdown and refilling impinges significant temporal differences on ecosystem responses. Months when shallowness took place behaved differently than those deepness ones. Also occurred with those months which suffered important hydrodynamic changes as the beginning of stratification and circulation periods, which were temporal managed through operational decisions. Temporal differences were most marked by solids, chl-*a* or inflows/outflows than by nutrients or water level. However, nutrient dynamics changes play an important role on trophic status of the reservoir.

Siltation is other global problem of the semi-arid and arid reservoirs (Verstraeten, Poesen, de Vente, & Koninckx, 2003; WCD, 2000). Palau (2002) pointed out in Spain (mostly semi-arid) a 0.5% annual losses of water storage capacity due to sediment accumulation (mean of ≈2,500 tons km^{-2} year^{-1}, maximum ≈4,000 tons km^{-2} year^{-1}). In Latin America, Africa and Asia, averages of sedimentation rates of 1,104, 259 and 293 tons km^{-2} year^{-1}, respectively, have been cited (White, 1993). Comparatively, our reported sedimentation rates in the reservoir (9,895 tons km^{-2} year^{-1}; 0.3% annual decrease in water storage capacity), in spite of low water inflows, is extraordinarily high.

In conclusion, our results show, as in other semi-arid regions, how complete exploitation of water resources strongly interferes with the chemical, physical and biological dynamics of the aquatic systems and with their seasonal patterns, increasing eutrophication. As water level changes, nutrient trends are modified as well as limiting nutrients also vary its dependence from external load and nutrient recycling is increased. The enhanced system complexity and operating mechanisms result unclear in order to develop management strategies for water quality improvement. Eutrophication in drylands is more critical as compared to temperate zones because of the scarcity of water availability. Environmental conditions of warm subtropic regions enhance the nutrient cycling, denitrification, and primary productivity as shallowness increases. Water drawdowns for irrigation or hydroelectric power need to be analyzed carefully in order to avoid the several ecological related changes, inducing a strong water quality decline which increases the treatment costs to meet the water quality requirements for different uses.

Acknowledgements This research was supported by CONACYT (contract R-35164-T). We are very grateful to Vicente Amezaga and to "El Anzuelo Fishing Club" by the use of their installations during the sampling. Thanks also to Rafael Angulo for your help on water and sediment analyses.

References

APHA (1998). *Standard methods for the examination of water and wastewater* (20th ed.) Washington: American Public Health Association.

Arfi, R. (2003). The effects of climate and hydrology on the trophic status of Sélingué Reservoir, West Africa. *Lakes and Reservoirs Research and Management, 8*, 247–257.

Barica, J., & Mur, L. R. (1980). Hypertrophic ecosystems. *Development on Hydrobiology, 2*, 9–11.

Bloesch, J. (1996). Towards a new generation of sediment traps and a better measurement/understanding a setting particulate flux in lakes and oceans: A hydrodynamic protocol. *Aquatic Science, 58*, 283–296.

de Anda, J., Shear, H., Maniak, U., & Riedel, G. (2001). Phosphates in Lake Chapala, Mexico. *Lakes and Reservoirs: Research and Management, 6*, 313–321.

de Jonge, V. N., Elliott, M., & Orive, E. (2002). Causes, historical development, effects and future challenges of a common environmental problem: Eutrophication. *Hydrobiologia, 475/476*, 1–19.

Downing, J. A., & MacCauley, E. (1992) The nitrogen : Phosphorus relationship in lakes. *Limnology and Oceanography, 37*, 936–945.

Eckert, W., Didenko, J., Uri, E., & Eldar, D. (2003). Spatial and temporal variability of particulate phosphorus fractions in seston and sediments of Lake Kinneret under changing loading scenario. *Hydrobiologia, 494*, 223–229.

Geraldes, A. M., & Boavida, M. J. (1999). Limnological comparison of a new reservoir with one almost 40 years old which had been totally emptied and refilled. *Lakes and Reservoirs Research and Management, 4*, 15–22.

Gin, K. Y. H., Zhang, Q. Y., Chan, E. S., & Chou, L. M. (2001). Three-dimensional ecological-eutrophication model for Singapore. *Journal of Environmental Engineering, 127*, 928–937.

Gochis, D. J., Leal, J. C., Shuttleworth, W. J., Watts, C. J., & Garatuza-Payan, J. (2003). Preliminary diagnostics from a new event-based precipitation monitoring system in support of the North American monsoon experiment. *Journal of Hydrometeorology, 4*, 974–981.

Håkanson, L. (2004). Internal loading: A new solution to an old problem in aquatic sciences. *Lakes and Reservoirs: Research and Management, 9*, 3–23.

Harper, D. (1992). *Eutrophication of freshwater: Principles, problems and restoration*. London: Chapman & Hall.

Howard, R. F., & Singer, M. J. (1981). Measuring forest soil bulk density using irregular hole, paraffin clod, and air permeability. *Forest Science, 27*, 316–322.

Huszar, V. L. M., Caraco, N. F., Roland, F., & Cole, J. (2006). Nutrient–chorophyll relationships in tropical–subtropical lakes: Do temperate models it? *Biogeochemistry, 79*, 239–250.

Hutchinson, G. E. (1957). *A treatise on Limnology, vol I. Geography, Physics and Chemistry*. New York: Wiley.

INEGI (2001). *Núcleos agrarios de Sonora: tabulados básicos por municipio 1992–2001*. México DF: Instituto Nacional de Estadística, Geografía e Informática.

Krogerus, K., & Ekholm, P. (2003). Phosphorus in settling matter and bottom sediments in lakes loaded by agriculture. *Hydrobiologia, 429*, 15–18.

Lewis, W. M., Jr. (2000). Basis for the protection and management of tropical lakes. *Lakes and Reservoirs Research and Management, 5*, 35–48.

Lewis, W. M. Jr. (2002). Causes for the high frequency of nitrogen limitation in tropical lakes. *Verhandlungen Internationale Vereinigung für theoretische und angewandte Limnologie, 28*, 210–213.

Marker, A. F. H., Nusch, E. A., Rai, H., & Riemann, B. (1980). The measurement of photosynthetic pigments in freshwaters and standarisation of methods: conclusions and recommendations. *Archiv für Hydrobiologie, 14*, 91–106.

Naghavi, B., & Adrian, D. D. (1993). Transient response of lake systems to phosphorus. *Journal of Environmental and System, 229*, 137–149.

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

Navarro, J. M. (2003). *Clasificación de la cobertura vegetal y uso del suelo en la cuenca del Río Yaqui y detección de cambios temporales usando datos del satélite LANDSAT MSS*. Master dissertation, Instituto Tecnológico de Sonora.

Nogueira, M. G., Henry, R., & Maricatto, F. E. (1999). Spatial and temporal heterogeneity in the Jurumirim Reservoir, São Paulo, Brazil. *Lakes and Reservoirs: Research and Management, 4*, 107–120.

Nõges, P., Järvet, A., Tuvikene, L., & Nõges; T. (1998). The budgets of nitrogen and phosphorus in shallow eutrophic Lake Võrtsjärv (Estonia). *Hydrobiologia, 363*, 219–227.

Palau, Y. A. (2002). *La sedimentación en embalses: medidas preventivas y correctoras*. Madrid: Dirección de Medio Ambiente y Calidad – Endesa Servicios.

Peeters, F., Piepke, G., & Gloor, M. (1997). A diffusion model for the development of a boundary layer in lakes. *Aquatic Science, 59*, 95–114.

Petterson, K. (1998). Mechanisms for internal loading of phosphorus in lakes. *Hydrobiology, 373–374*, 21–25.

Quintana, X. D., Comín, F. A., & Moreno-Amich, R. (1998). Nutrient and plankton dynamics in a Mediterranean salt marsh dominated by incidents of flooding. Part 2: Response of a zooplanktoon comunity to disturbances. *Journal of Plankton Research, 20*, 2109–2127.

Ramm, K., & Scheps, V. (1997). Phosphorus balance of a polytrophic shallow lake with the consideration of phosphorus release. *Hydrobiologia, 342–343*, 43–53.

Reddy, C. N., DeLaune, R. D., DeBusk, W. F., & Koch, M. S. (1993). Long-term nutrient accumulation rates in the Everglades. *Soil Science Society of American Journal, 57*, 1147–1155.

Reddy, K. R., Fisher, M. M., & Ivanoff, D. (1996). Resuspension and diffusive flux of nitrogen and phosphorus in a hypertrophic lake. *Journal of Environmental Quality, 25*, 363–371.

Reddy, K. R., O'Connor, G. A., & Schelske, C. L. (Eds.) (1999). *Phosphorus biogeochemistry in subtrophical ecosystems*. Boca Raton: Lewis.

Reynolds, C. S. (1992). Eutrophication and the management of plankton algae: What Vollenweider couldn't tell us. In D. W. Sutcliffe & J. G. Jones (Eds.), *Eutrophication: Research and application to water supply* (pp. 4–29). Ambleside: Freshwater Biological Association.

Ruley, J. E., & Rusch, K. A. (2004). Development of a simplified phosphorus management model for a shallow, subtrophical, urban hypereutrophic lake. *Ecological Engineering, 22*, 77–98.

Ryding, S. O., & Rast, W. (1989). *The control of eutrophication of lakes and reservoirs*. New Jersey: Parthenon.

Sánchez-Carrillo, S., & Alvarez-Cobelas, M. (2001). Nutrient dynamics and eutrophication patterns in a semi-arid wetland: The effects of a fluctuating hydrology. *Water Air and Soil Pollution, 131*, 97–118.

Sánchez-Carrillo, S., Alvarez-Cobelas, M., & Angeler, D. G. (2001). Sedimentation in the semi-arid freshwater wetland Las Tablas de Daimiel (Spain). *Wetlands, 21*, 112–124.

Scharf, W. (2002). Refilling, ageing and water quality management of Brucher Reservoir. *Lakes and Reservoirs: Research and Management, 7*, 13–23.

Serruya, C. (1976). Rates of sedimentation and resuspension in Lake Kinneret. In H. L. Golterman (Ed.), *Interactions between Sediments and Freshwater* (pp. 48–56). The Hague: Dr. W. Junk.

Settacharnwit, S., Buckney, R. T., & Lim, R. P. (2003). The nutrient status of Nong Han, a shallow tropical lake in north-eastern Thailand: Spatial and temporal variations. *Lakes and Reservoirs: Research and Management, 8*, 189–200.

Skei, J., Larsson, P., Rosenberg, R., Jonsson, P., Olsson, M., & Broman, D. (2000). Eutrophication and contaminants in aquatic ecosystems. *Ambio, 29*, 184–194.

Sondergaard, M., Jensen, J. P., & Jeppesen, E. (1999). Internal phosphorus loading in shallow danish lakes. *Hydrobiologia, 408–409*, 145–152.

Sparks, D. L. (Ed.) (2002). Methods of Soil Analysis Part 3: Chemical Methods. Madison: Soil Science Society of America Book Series, no. 5.

Statsoft Inc. (2001). *STATISTICA v6.1 for windows.* Tulsa: Statsoft Incorporation.

Straškraba, M., & Tundisi, J. G. (1999). Reservoir ecosystem functioning: theory and applications. In J. G. Tundisi & M. Straškraba (Eds.), *Theoretical reservoir ecology and its applications* (pp. 565–597). Leiden: Backhuys.

Thornton, J. A. (1980). Comparison of the summer phosphorous loading, to three Zimbabwean water supply reservoirs of varying trophic states. *Water SA, 6*, 163–170.

Townsend, S. A. (1999). The seasonal pattern of dissolved oxygen, and hypolimnetic deoxygenation, in two tropical Australian reservoirs. *Lakes and Reservoirs: Reseach and Management, 4*, 41–53.

Verstraeten, G., Poesen, J., de Vente, J., & Koninckx, X. (2003). Sediment yield variability in Spain: A quantitative and semiqualitative analysis using reservoir sedimentation rates. *Geomorphology, 50*, 327–348.

Vinçon-Leite, B., Tassin, B., & Druart, J. C. (2002). Phytoplankton variability in Lake Bourget: Phytoplankton dynamics and meteorology. *Lakes and Reservoirs*: *Research and Management, 7*, 93–102.

Volohonsky, H., Shaham, G., & Gophen, M. (1992). The impact of water inflow reduction on the trophic status of lakes. *Ecological Modelling, 62*, 135–147.

WCD (2000). *Dams and development. A new framework for* *decision making. Report of the World Commission on Dams*. London: Earthscan.

Wetzel, R. G. (1990). Land–water interfaces: Metabolic and limnological regulators. *Verhandlungen Internationale Vereinigung für theoretische und angewandte Limnologie, 24*, 6–24.

White, P. (1993). Environmental problems in the peat moorlands of the Southern Pennines: Reservoir sedimentation and the discolouration of water supplies. In J. C. Currie & A. T. Pepper (Eds.), *Water and the environment* (pp. 246–263). Chichester: Ellis Horwood.