J Paleolimnol (2019) 62:337–357
https://doi.org/10.1007/s10933-019-00090-4

ORIGINAL PAPER

Assessing the degree of ecological change and baselines for reservoirs: challenges and implications for management

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Received: 13 August 2018 / Accepted: 9 August 2019 / Published online: 10 September 2019

Abstract Tropical reservoirs are sensitive to eutrophication but long-term impacts of impoundment on their productivity and biota are poorly understood. Here, we employ a palaeolimnological approach to assess whether ecological baselines can be defined for reservoirs, and examine the challenges and management implications. We studied the environmental history of five reservoirs in Brazil with different productivities, using sediment records covering the period since reservoir construction (~ 50–90 years). Our main goals, based on the analysis of organic geochemistry (TOC, TN, TP, C:N), stable isotopes (δ^{13}C, δ^{15}N), and diatoms, were to reconstruct and compare the magnitude of environmental changes, to determine the conditions prior to any eutrophication and discuss the implications for reservoir management. We inferred that Ribeirão do Campo reservoir has remained oligotrophic since its construction, Itupararanga and Paineiras have both remained mesotrophic with an improvement in water quality around 1970, and Salto Grande has been eutrophic since its construction in 1949. In Rio Grande reservoir, which was originally oligotrophic, eutrophication began in the 1950s, with a slight improvement in water quality after its separation from Billings Reservoir, followed by a subsequent decline in quality since ~ 2001. We found that the studied reservoirs have unique environmental histories and there are clearly challenges associated with defining ecological baselines for reservoirs against which the extent of degradation can be assessed. Nonetheless, when the data from all reservoirs were compared, a coherent pattern in the diatom assemblages emerged, reflecting the trophic gradient. The diatom assemblages prior to enrichment were composed of two groups. The oligotrophic baseline was characterized by several benthic species with low abundances, mainly *Eunotia* and *Brachysira*, while in three reservoirs the early assemblages were characterized by planktonic taxa associated with mesotrophic conditions, namely planktonic species *Aulacoseira ambigua*, *Aulacoseira tenella*, *Discostella stelligera*, and *Spicaticriba rudis*. This work provides information on the baseline conditions, the natural

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Electronic supplementary material The online version of this article (https://doi.org/10.1007/s10933-019-00090-4) contains supplementary material, which is available to authorized users.

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variability of non-degraded reservoirs, and the degree of ecological change in degraded ecosystems. This study contributes to an improved understanding of the timing and extent of eutrophication in these systems and provides information to help better inform the management of tropical reservoirs vulnerable to anthropogenic pressures.

Keywords Diatoms · Eutrophication · Stable isotopes · Squared chord distance

Introduction

Tropical environments are particularly vulnerable to eutrophication. Owing to high temperatures, stratification events are likely to persist over the year and consequently phosphorus release from sediments is intensified, increasing system resilience to restorative strategies (Bicudo et al. 2007). Over 70% of Brazil's rivers have been dammed to create reservoirs (Kelman et al. 2002) and hundreds more are under consideration to be built (Latrubesse et al. 2017). Many of these ecosystems have experienced extensive environmental degradation since the 1960s, when the population growth, combined with poor wastewater infrastructure, exceeded the provision of adequate sanitation (Beyruth 2000). Eutrophication with occurrence of algal blooms is common in these systems and, in some cases, algicides have been applied to control algae in reservoirs used for water supply (Capobianco and Whately 2002). Given this fast-changing landscape and warmer tropical temperatures, it is essential to define restoration targets for reservoirs in Brazil and other tropical climate zones.

Paleolimnology has played a key role in understanding lake response to eutrophication over decadal to centennial timescales (Davidson and Jeppesen 2013). Owing to the high temporal resolution of sediment records from most lowland lakes, ecological and chemical change can be inferred over timeframes that are meaningful to lake managers. Species assemblage data, as well as estimates of past total phosphorous (TP) concentrations using diatom-TP transfer functions, have been widely employed to determine pre-enrichment baseline or reference conditions which can, in turn, be used to inform restoration targets (Bjerring et al. 2008; Battarbee and Bennion 2011). The deviation of current conditions from those of the baseline can also be employed by managers to assess ecological status and to evaluate the degree of recovery and thus whether restoration efforts have been successful (Bennion et al. 2015). Management strategies informed by paleolimnological studies include sediment removal to reduce nutrients, and maximize restoration potential through exposure of dormant macrophyte oospores, fish management (Sayer et al. 2012) and development of guidelines to control both phosphorous and nitrogen sources in the watershed (Adams et al. 2014).

Most of our understanding of ecological baselines and restoration targets from palaeoecological studies centres on natural lakes in temperate regions. Paleolimnological studies in reservoirs are fewer and did not take off to any great extent until the 1990s (Donar et al. 1996; Hall et al. 1999). Despite the paucity of reservoir sediment studies, several advances have been made. Importantly, concerns about the accuracy and quality of reservoir sediment history, mainly regarding changes in the sedimentation rates, have been overcome (Shotbolt et al. 2006; Filstrup et al. 2010; Winston et al. 2014). Furthermore, several proxies have been employed to provide interpretations of ecosystem changes in response to anthropogenic impacts in reservoirs. Diatoms are a well-known proxy, sensitive to changes in productivity (Woodbridge et al. 2014; Schroeder et al. 2016), and geochemical markers can also be particularly valuable to trace the history of trophic changes (Filstrup et al. 2010; Kumar et al. 2016). A combination of both geochemical and biological proxies has been recently used with success to determine environmental change in reservoirs in North and South America (Winston et al. 2014; Fontana et al. 2014).

To date, only a few paleolimnological studies have addressed the issue of eutrophication in tropical reservoirs. Long-term changes in hydrodynamics and spatial gradients were assessed with diatoms (Liu et al. 2012) and with Cladocera (Xu et al. 2017) in Liuxihe Reservoir in southern China. In Brazil, Costa-Böddeker et al. (2012) and Fontana et al. (2014) studied the anthropogenic impacts in urban reservoirs, both concluding that cultural eutrophication began in the mid-1970s and intensified over the 1990s. More recently, Wengrat et al. (2018) evaluated patterns of spatial and temporal biodiversity in seven tropical reservoirs in Brazil, concluding that eutrophication leads to homogenization of the diatom assemblage, with significant loss of species over time. Despite the increased interest in reservoir paleolimnology (Tibby et al. 2010; Liu et al. 2012; Elchyshyn et al. 2018), few studies focus on implications for management, and the magnitude of environmental change in reservoirs is poorly understood relative to natural waterbodies.

The present study aimed to reconstruct and compare the magnitude of environmental change in five tropical reservoirs in Brazil along a trophic gradient from oligotrophic to eutrophic. We aimed to determine the conditions prior to any eutrophication, assess the degree of ecological change and to consider the implications for reservoir management. Paleolimnological techniques were employed to explore both temporal and spatial variation in the five reservoirs. We analyzed geochemistry and diatom assemblages in dated sediment cores, covering a period of 50–100 years, to determine if observed shifts were associated with changes in trophic status and if the conditions prevailing prior to enrichment in impacted sites were similar to those seen in oligotrophic reservoirs in the current population. In addition, in the oligotrophic and mesotrophic systems, evidence for natural variability in the diatom assemblages was examined. We expected similar diatom assemblages to be present at the base of each core (i.e. prior to impact) and for diatom changes to reflect a response to eutrophication.

Study sites

The reservoirs are located in a tropical climate (Cepagri 2015) in São Paulo State, Brazil, in four drainage basins. Five reservoirs were selected considering their age, nutrient gradient, and the absence of sediment removal by dredgers (Table 1). The surrounding areas are variously dominated by rainforest flora, agriculture or are densely populated, for example São Paulo Metropolitan Region, with nearly 20 million inhabitants (IBGE 2014). In order to cover the period prior to eutrophication, which according to previous paleolimnological studies started around the 1970s in São Paulo (Costa-Böddeker et al. 2012; Fontana et al. 2014), we sampled only reservoirs constructed before the 1960s. In shallow reservoirs, diatom response to eutrophication can be complicated by the influence of habitat availability such as macrophytes and periphyton (Wengrat et al. 2018). To avoid this, only sites with an average water column depth of > 7 m were included.

Table 1 Main characteristics of the five study sites and cores [Z max, maximum water depth; trophic state based on Cunha et al. (2013) for tropical reservoirs]

Reservoir	Constructed year	Coordinates of core sites	Coring date (month/year)	Area (km^2)	Z max (m)	Water depth (m)	Core length (cm)[a]	Code	Trophic state	TP (μg L^{-1})	TN (μg L^{-1})	Chl-a (μg L^{-1})
Ribeirão do campo	1958	23°38′34″S 45°49′55″W	11/13	1.6	26	10	26	RC13	Oligotrophic	4.4	418.6	1.8
Rio Grande	1927	23°45′50″S 46°31′23″W	11/12	16.1	15	11.3	36	RG12	Mesotrophic–eutrophic	23.0	1078.7	10.4
Salto Grande	1949	22°43′05″S 47°16′02″W	07/13	9.3	19	8	28	SG13	Eutrophic	112.7	2050.4	125.3
Itupararanga	1914	23°37′18″S 47°23′53″W	10/13	24.4	20	15	46	IT13	Mesotrophic	15.9	1735.8	4.0
Paineiras	1912	23°50′35″S 47°38′26″W	12/13	2.6	12	9	36	PI13	Mesotrophic	17.4	285.8	2.8

Limnological data from the core sites [mean values from the water column, methodology published in Wengrat et al. (2018)]

TP total phosphorous, *TN* total nitrogen, *Chl-a* chlorophyll-a

[a] Length of the core profile from the reservoir phase

Oligotrophic reservoir (Ribeirão do Campo)

Constructed in 1958, this reservoir is the site most protected from anthropogenic impacts in our study. Located in Alto Tietê headwater basin, the reservoir is surrounded by native forest and the watershed is part of an environmental preservation area of the Atlantic Forest. Ribeirão do Campo is oligotrophic, with weakly acidic waters and low conductivity (Santana et al. 2017).

Mesotrophic reservoirs (Paineiras and Itupararanga)

Paineiras was constructed in 1912, and the surface water is mesotrophic (Table 1). Native forest, pastureland, and Eucalyptus plantations are the dominant forms of land use (Ferreira 2011). Itupararanga reservoir was constructed in 1914 and is surrounded by agricultural fields and native forest. Although it receives some eutrophic water from tributaries, the water quality is considered mesotrophic close to the dam (Pedrazzi et al. 2013).

Eutrophic reservoirs (Salto Grande and Rio Grande)

Salto Grande reservoir was constructed in 1949. Located in a densely populated area, industrialized and with intensive agriculture, the water quality is highly degraded and cyanobacterial blooms are frequent (Espíndola et al. 2004). Rio Grande reservoir was isolated from Billings reservoir (1927) in 1982 in an attempt to preserve its water quality. The contamination comes mainly from sewage and diffuse pollution from the densely populated and industrialized surrounding area (Capobianco and Whately 2002). The reservoir's water quality is eutrophic with algal blooms in the upstream region and mesotrophic at the sampling site of this study as a result of algicide application (Wengrat and Bicudo 2011). High concentrations of copper sulfate ($CuSO_4$) used as algicide have been detected in sediments (Franklin et al. 2016).

Methods

Core sampling

Sediment cores were collected by divers in 2012–2013 (Table 1). The approximate sampling site was chosen based on previous research on the reservoirs and local expertise. The exact coring location was selected following a bathymetric survey (Fig. 1). One core per reservoir was collected using an acrylic tube, 7.6 cm in diameter and 150 cm long. Water above the surface sediment was removed with a thin hose before sectioning. The cores were sliced in the field at 1-cm intervals (except RG12, sliced at 2-cm intervals). Subsamples were subsequently used for physical (dry mass and water content), dating, chemical (TOC, TN, TP), isotopic ($\delta^{15}N$ and $\delta^{13}C$), and diatom analyses.

Chronology

The cores were dated radiometrically (^{210}Pb, ^{226}Ra, ^{137}Cs) by gamma spectrometry at the Inorganic Marine Chemistry Laboratory (University of São Paulo), using an EGandG ORTEC low-background gamma spectrometer (GMX25190P model) according to Ferreira et al. (2016). The following activity peaks were detected: ^{210}Pb (46.52 keV), ^{226}Ra (609.31 keV), and ^{137}Cs (661.67 keV). The activity of ^{137}Cs was analyzed as an auxiliary parameter for unsupported ^{210}Pb modeling and as a historical marker of 1963 (Ferreira et al. 2016). Two models were used to calculate the sedimentation rates, CIC (Constant Initial Concentration) and CRS (Constant Rate of Supply), which were validated with ^{137}Cs and historical events for each reservoir (Appleby and Oldfield 1978). Samples were selected every two centimeters (0–2 cm, 2–4 cm, 4–6 cm, etc.), and approximately 10 g of each dried sediment sample was homogenized and placed into individual air-sealed polyethylene containers.

Sediment chemistry

Total organic carbon (TOC), total nitrogen (TN), and the stable isotopes ($\delta^{13}C$, $\delta^{15}N$) were analyzed using an element analyzer (ANCA-GSL) interfaced to a PDZ Europa 20–20 isotope ratio mass spectrometer at the University of California, Davis, Stable Isotope Facility. Delta values are expressed relative to international standards V-PDB (Vienna PeeDee Belemnite) for carbon and Air for nitrogen (UC DAVIS 2015). Total phosphorus (TP) was analyzed by the colorimetric method (Valderrama 1981), after acid digestion with nitric and hydrochloric acid (HNO_3 + HCl) (Andersen 1976). Chemical and isotopic analyses were performed on samples at 2-cm intervals for all core profiles.

Diatoms

Diatom samples were cleaned using hydrogen peroxide (H_2O_2) and hydrochloric acid (HCl) according to Battarbee et al. (2001). Permanent slides were mounted using Naphrax®. Counts were made using a Zeiss® Microscope (Axio Imager A2) at 1000 × magnification. At least 400 valves were counted per slide according to Battarbee et al. (2001) until reaching an efficiency of 90% (Pappas and Stoermer 1996). Species abundances were calculated as percentages of the total counts in each subsample. Taxonomy and nomenclature followed classic literature and recent reviews (Round et al. 1990; Metzeltin and Lange-Bertalot 2007; Spaulding et al. 2010). Diatom analysis was carried out on every 2nd or 3rd subsample and according to changes in the lithology for each core.

Numerical analysis

To visualize the major change in sediment chemistry, stratigraphic diagrams were produced using the C2 program, version 1.3 (Juggins 2003). The main changes in organic matter were characterized by C:N ratio and stable isotope composition ($\delta^{13}C$, $\delta^{15}N$) (Meyers 2003; Costanzo et al. 2003). To identify the major zones in the diatom records from each core, constrained incremental sum of squares (CONISS) was carried out using TILIAGRAPH 1.7.16 (Grimm 1991) and the stratigraphic changes were plotted for the abundant diatom species ($\geq$ 2%). To explore and compare assemblage-level change across reservoirs and time, we performed a principal coordinate analysis (PCoA, Gower 1966) using Bray–Curtis distance. The degree of dissimilarity between the sample prior to enrichment and every sample analyzed up to the core top was calculated using the squared chord distance (SCD) dissimilarity measure (Overpeck et al. 1985). To select the diatom reference sample, we selected the lowermost sample in each core, excluding the samples which correspond to the construction phase of the reservoir, in order to avoid the period of ecosystem instability. The SCD values range from 0 to 2, 0 being indicative of a pair of perfectly similar subsamples and a value of 2 being perfectly dissimilar. Given that reservoirs experience greater hydrological changes than most natural lakes, we use less stringent criteria for defining the class ranges for our study sites: scores less than 0.45 indicate minimal changes, between 0.45 and 0.6 indicate slight changes, between 0.6 and 0.93 indicate moderate changes, and greater than 0.93 indicate major changes (Bennion et al. 2011).

Results

Core chronologies

The dating results permitted generation of a chronology for four cores, PI13, IT13, SG13, and RG12. The CIC dating model was considered appropriate for the cores from the mesotrophic reservoirs (PI13 and IT13), both with two decay curves (Fig. 2), while the CRS model was more appropriate for the eutrophic sites, SG13 and RG12 (Fig. 3). In all the profiles, it was possible to observe the decay of unsupported (excess) ^{210}Pb until it achieved radioactive equilibrium, as well as the ^{137}Cs peak, which was used as an auxiliary evaluation method for the sedimentation rates (ESM1-4). As ^{210}Pb cannot reach radioactive equilibrium in young reservoirs (Winston et al. 2014), subsamples from the pre-dam phase were used for calculating PI13, IT13, SG13 and RG12 chronologies (Figs. 2, 3). Geochemistry profiles showed abrupt changes for all sites during reservoir formation and confirmed the period of dam construction (Fig. 4). The Rio Grande sample (RG12) had low levels of ^{210}Pb in the upper sediment, however, it was possible to derive a chronology based on the ^{210}Pb model with the remaining samples of the core and via ^{137}Cs activity, whose record was good with a clear maximum peak at 14 cm sediment depth, most likely corresponding to 1963 (Ferreira et al. 2016). For the fifth core, Ribeirão do Campo (RC13), there was a marked and synchronous change in several proxies (diatoms, lithology, density, total phosphorus) at 26 cm (ESM5, 6). The composition of the sediment changes from light color, composed of sand and no diatoms to brown sediments, a muddy appearance and plentiful diatoms, suggesting that reservoir construction probably occurred around 26 cm. There was no major change in the proxies (diatoms, geochemical and stable isotopes) throughout this core and no zones were identified after reservoir construction (26 cm).

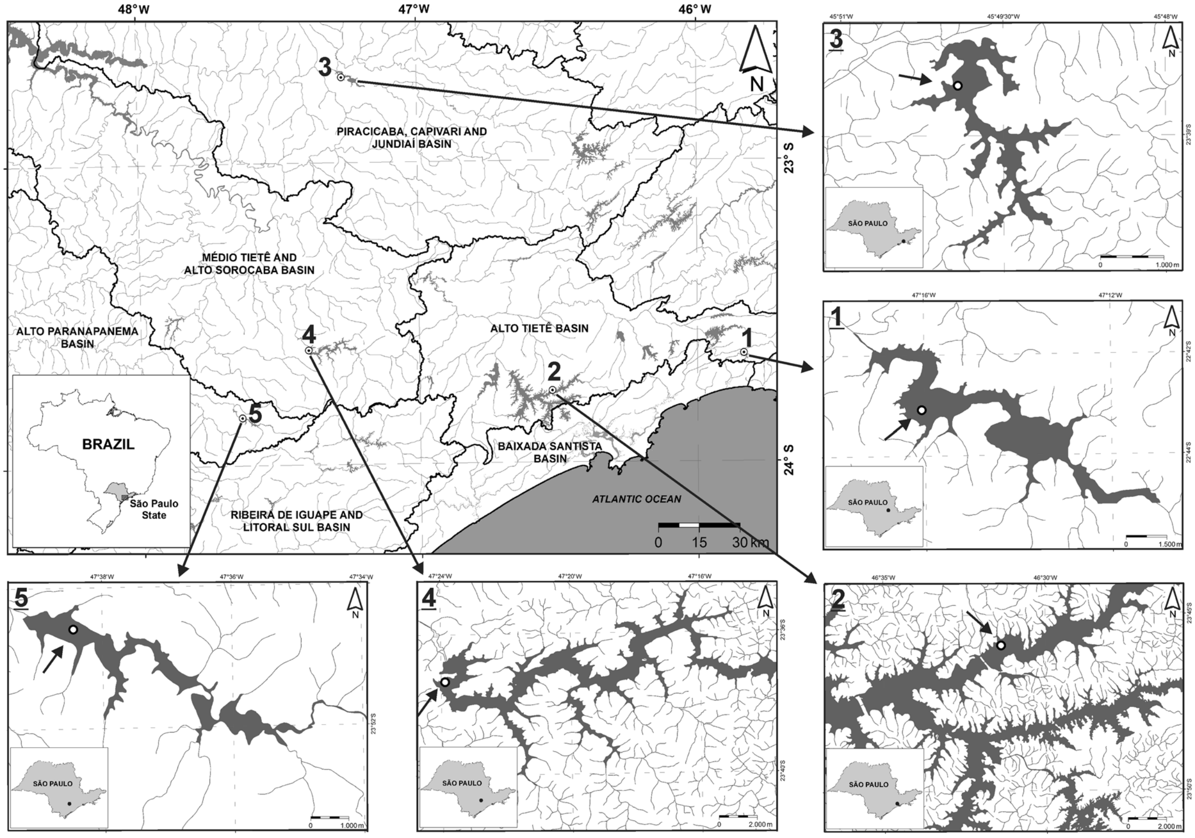


Fig. 1 Location of the reservoirs and river basins: 1 Ribeirão do Campo reservoir, 2 Rio Grande reservoir, 3 Salto Grande reservoir, 4 Itupararanga reservoir and 5 Paineiras reservoir. Smaller arrows indicate the location of sampling sites for each reservoir

Physical and chemical profiles

Differences along the geochemistry profiles were found in the mesotrophic (IT13, PI13) and eutrophic reservoirs (SG13, RG12) but not in the oligotrophic reservoir (RC13). The results are presented according to reservoir productivity (Fig. 4).

Oligotrophic reservoir

There was no notable change in the physical and geochemical profiles throughout this core. Of all the cores, the highest values of TOC ($\sim$ 8.6%) and C:N ratio ($\sim$ 15), as well as the lowest values of the stable isotopes δ^{13}C (– 30‰) and δ^{15}N ($\sim$ 2.5‰), were observed in RC13 (Fig. 4a).

Mesotrophic reservoirs

The geochemical profiles of the two mesotrophic reservoirs share some similarities, with sharp variations during reservoir construction and with stabilization towards the top. For Paineiras reservoir (PI13, Fig. 4b), at the bottom of the core ($\sim$ 1914–1920), the geochemical values were differentiated from the rest of the profile, particularly the TOC, TN, C:N and N:P ratio markers which were higher, and TP which was lower, than the values up to the surface, where all the values stabilized (34–0 cm). In the Itupararanga reservoir (IT13, Fig. 4c), a sharp decrease in TOC, TN, C:N ratio, and δ^{13}C occurred from 46 to 35 cm ($\sim$ 1920–1938). From 35 cm to the top, the most stable values were found; the exception being the isotopes (δ^{13}C, δ^{15}N), which displayed some minor variations, with a slight peak in TOC, TN, and TP at 15 cm, as well as a gradual decrease in δ^{15}N starting at 10 cm and continuing upward.

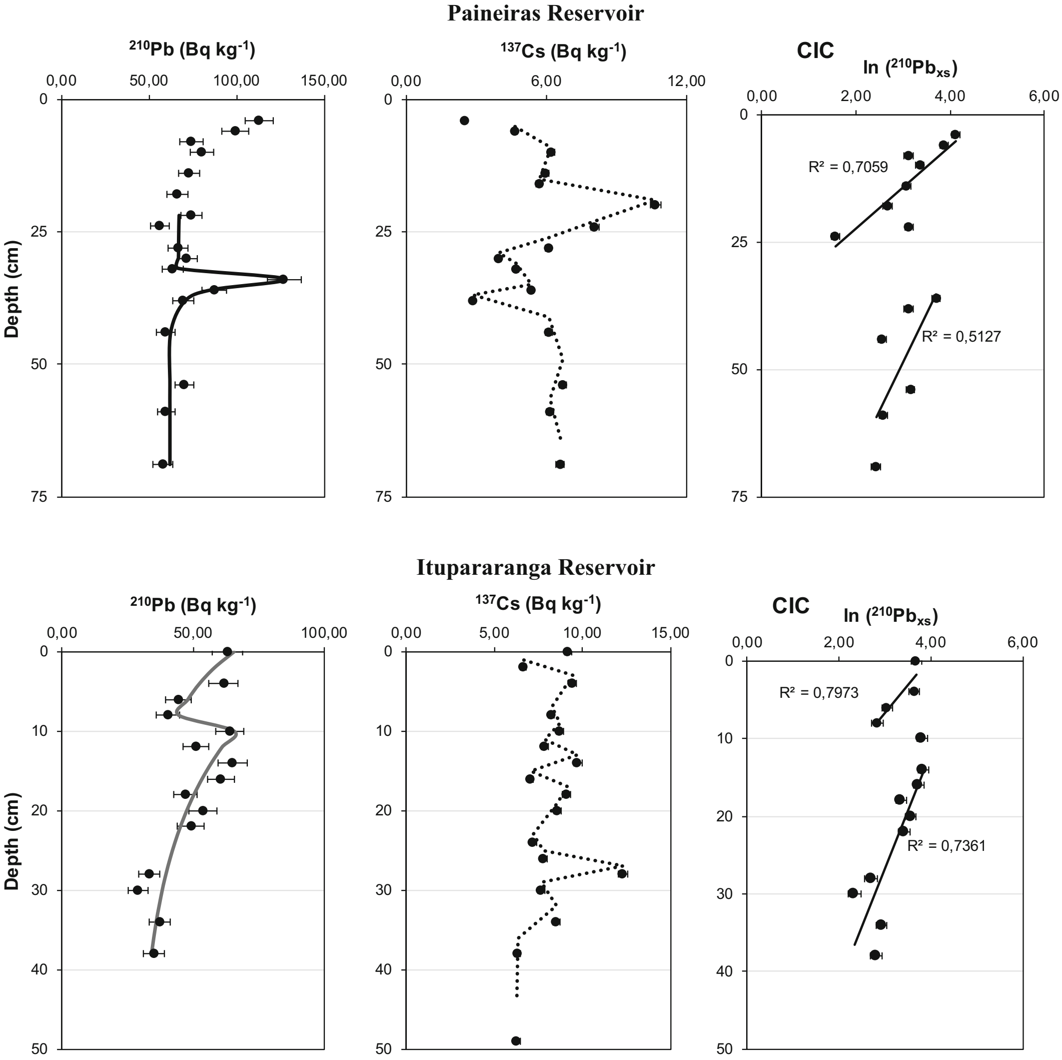


Fig. 2 ^{210}Pb and ^{137}Cs activities (Bq kg^{-1}) and decay model (CIC) versus depth (cm) from the mesotrophic cores Paineiras PI13 and Itupararanga IT13

Eutrophic reservoirs

Salto Grande (SG13, Fig. 4d): The values of TP (1.8–2.2 mg g^{-1}) at the bottom of the core were higher than those of the oligotrophic and mesotrophic reservoirs (0.1–0.9 mg g^{-1}). There was a sharp decrease in TOC, TN and N:P ratio, as well as an increase in TP, C:N ratio, and δ^{13}C during the construction phase (~ 1950). Toward the top (~ 1963–2012), TP and δ^{15}N increased (up to 2.6 mg^{-1} and 13.7, respectively). The C:N ratio remained around 10 throughout the core.

Rio Grande core (RG12, Fig. 4e): After reservoir construction, a marked decline occurred in the values of TOC, TN, and C:N ratio, which continue declining until stabilization around (~ 1950), except C:N ratio, which stabilized only in the top. TP and δ^{15}N increased along the profile, reaching their maximum values between 1964 and 1970, followed by δ^{13}C. Between (1976 and 1992), there was a reversal in the TP and δ^{15}N values, with a decrease towards 1992, when TP and δ^{13}C values stabilized. TOC and TN slightly increased with a notably sharp increase in δ^{15}N during recent years.

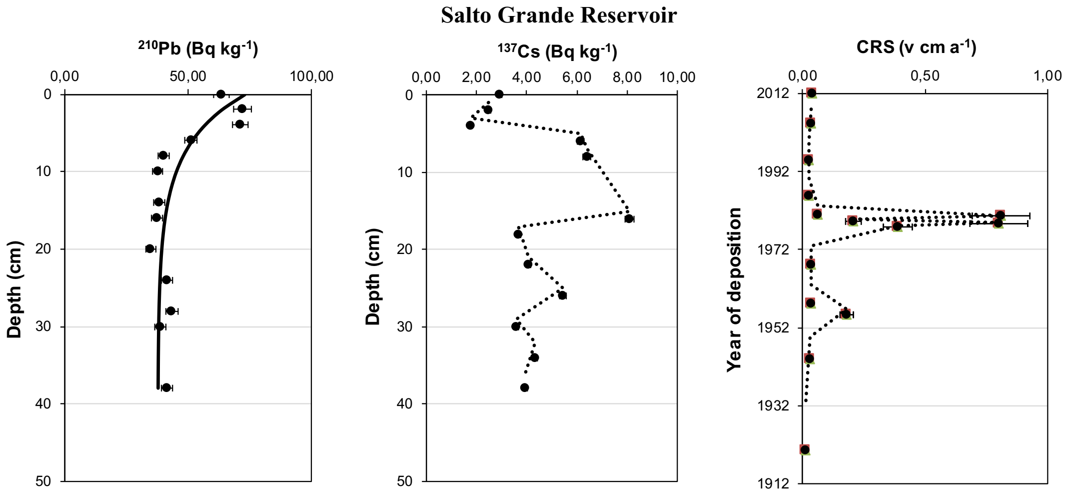
Diatoms

Oligotrophic reservoir

A total of 137 species were identified in core RC13 (Ribeirão do Campo), with 23 taxa occurring at a relative abundance ≥ 2% (Fig. 5). The diatom assemblages were relatively stable throughout the core and no zones were identified by the cluster analysis. The core was dominated by benthic species (∼ 80%). The most abundant taxa (15–25%) were present throughout the whole core, namely *Brachysira brebissoni* Ross, *Encyonopsis sanctipaulensis* Wengrat et al. and *Frustulia crassinervia* (Brébisson) Lange-Bertalot. Toward the top (above 10 cm), some new species appeared or increased (9–16%), such as *Kobayasiella subtilissima* (Cleve) Lange-Bertalot and *Stenopterobia delicatissima* (Lewis) Brébisson ex Van Heurck. The SCD score of 0.70 between the core bottom and top samples reflects a moderate change in the assemblage structure over the time period.

Mesotrophic reservoirs

In core PI13 from Paineiras, a total of 148 species were identified, with only 14 taxa occurring at ≥ 2% (Fig. 6a). The core was dominated by planktonic species (∼ 90%). Cluster analysis identified two



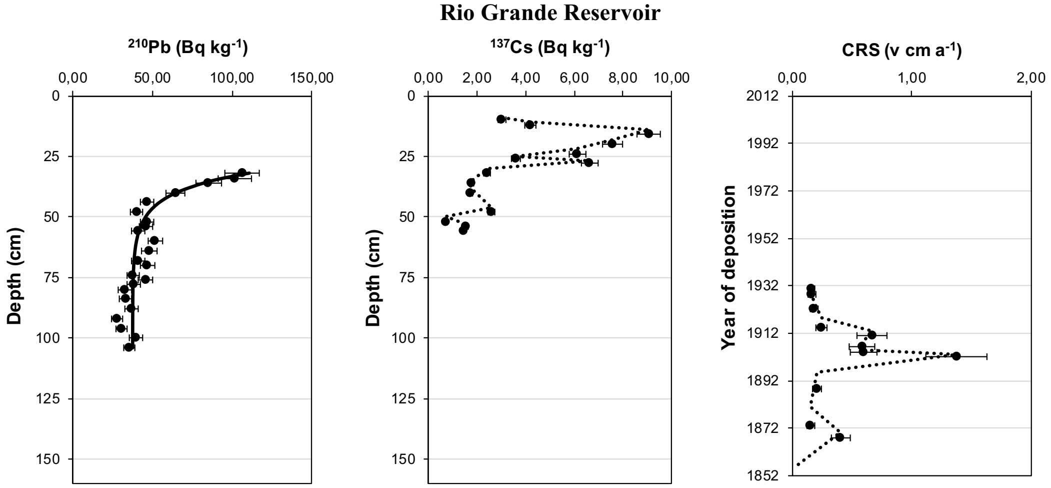
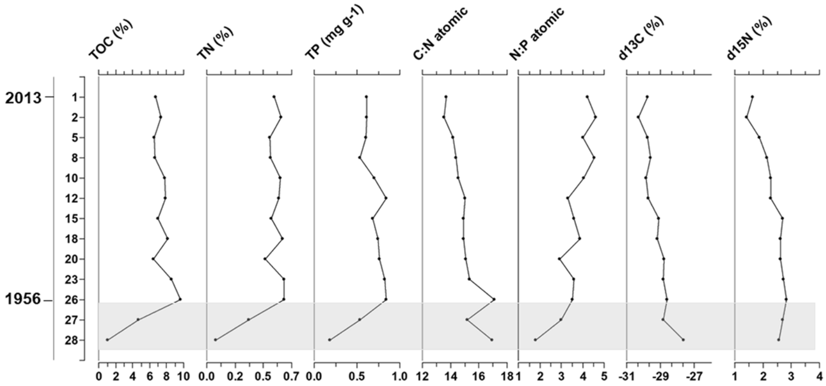
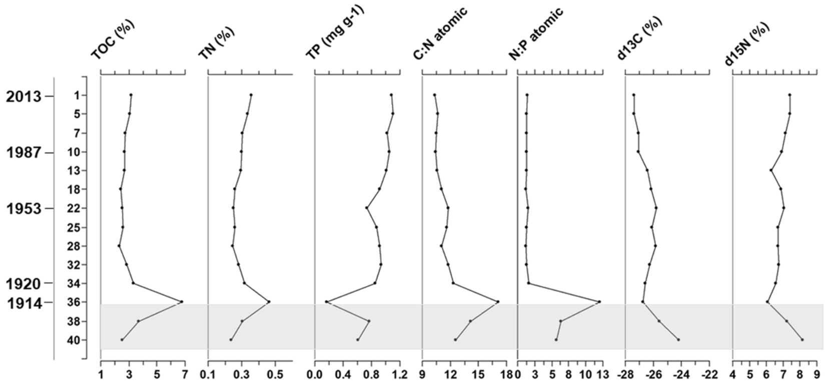


Fig. 3 ^{210}Pb and ^{137}Cs activities (Bq kg^{-1}) versus depth (cm) and decay model (CRS) of sedimentation rate from the eutrophic cores Salto Grande (SG13) and Rio Grande (RG12)

(a) **Oligotrophic reservoir (Ribeirão do Campo)**



(b) **Mesotrophic reservoir (Paineiras)**



(c) **Mesotrophic reservoir (Itupararanga)**

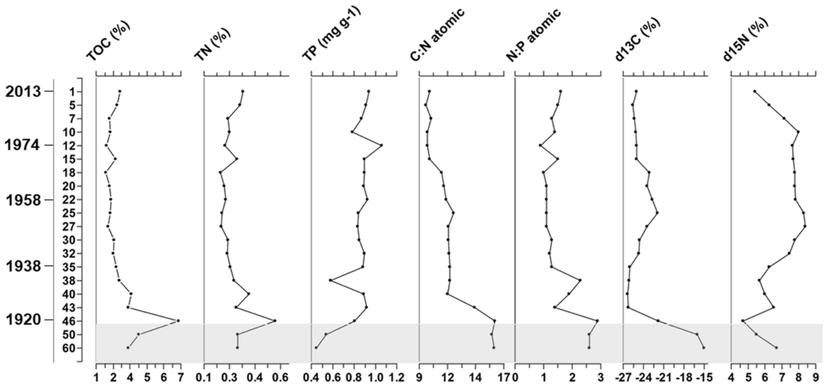
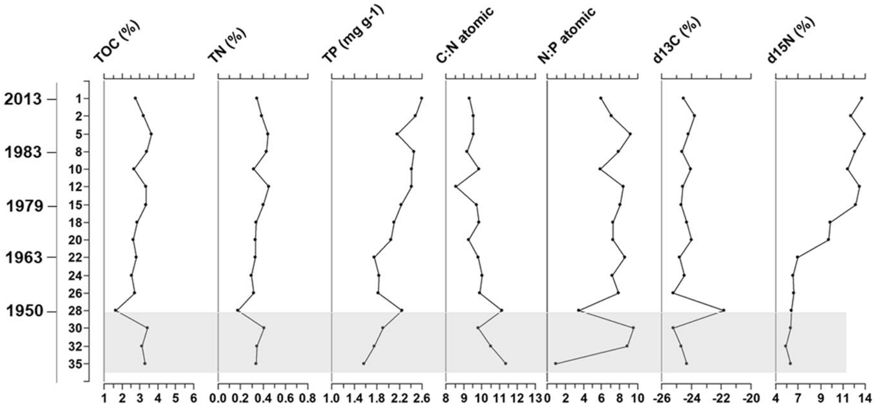


Fig. 4 Total organic carbon (TOC), total nitrogen (TN), total phosphorus (TP), C:N ratio, N:P ratio, δ^{13}C and δ^{15}N versus depth in the cores: **a** Ribeirão do Campo (RC13), **b** Paineiras (PI13), **c** Itupararanga (IT13), **d** Santo Grande (SG13) and **e** Rio Grande (RG12). The grey shadow represents the geochemical values before the formation of each reservoir. Note the abrupt changes during dam constructions

(d) Eutrophic reservoir (Salto Grande)



(e) Meso-eutrophic reservoir (Rio Grande)

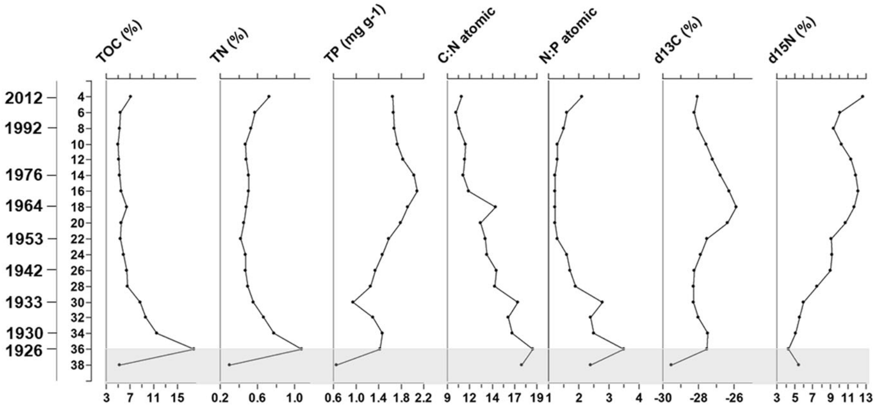
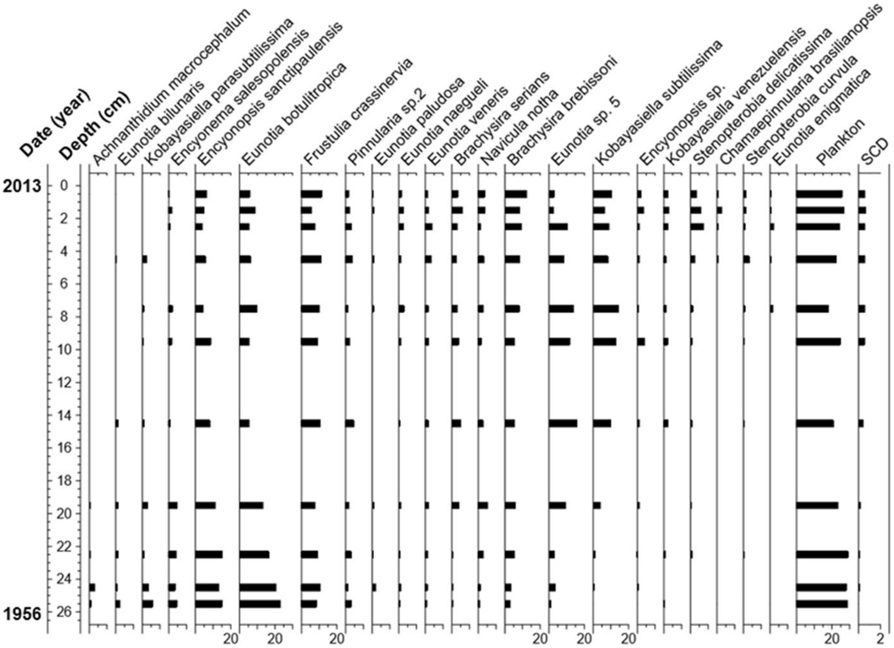


Fig. 4 continued

zones. In Zone 1 (~ 1914–1976), there was a high abundance of *Aulacoseira ambigua* (Grunow) Simonsen (49%), *Aulacoseira granulata* (Ehrenberg) Simonsen (24%) *Aulacoseira granulata tenella* (Nygaard) Simonsen (39%) and *Discostella stelligera* (Cleve and Grunow) Houk & Klee (~ 25%). In zone 2 (~ 1976–2013), there was a large decrease in *A. ambigua* and *A. granulata* (2.6–4.5%), and conversely an increase in *D. stelligera* (~ 60%). The relative abundances of *A. pusilla* (Meister) Tuji & Houki and *Spicaticriba rudis* (Tremarin et al.) Tuji et al., although < 10%, also increased in this zone. The SCD score of 0.78 between core bottom and top samples indicated moderate change in the assemblage composition.

A total of 140 species were identified in core IT13 from Itupararanga, with only 17 taxa occurring at ≥ 2% (Fig. 6b). The whole core was dominated by planktonic species (~ 90%). Cluster analysis identified two zones, one of them divided into two subzones. In zone 1 (~ 1920–1954), *A. ambigua* was dominant (87%), while *D. stelligera* and *A. tenella* occurred but in relatively low abundance. In subzone 2a (~ 1954–1968), several taxa appeared, most notably *S. rudis, A. granulata* and *A. ambigua* var. *japonica* Tuji & D.M.Williams, while *A. ambigua*

Fig. 5 Diatom stratigraphy of the oligotrophic reservoir (Ribeirão do Campo, core RC13)

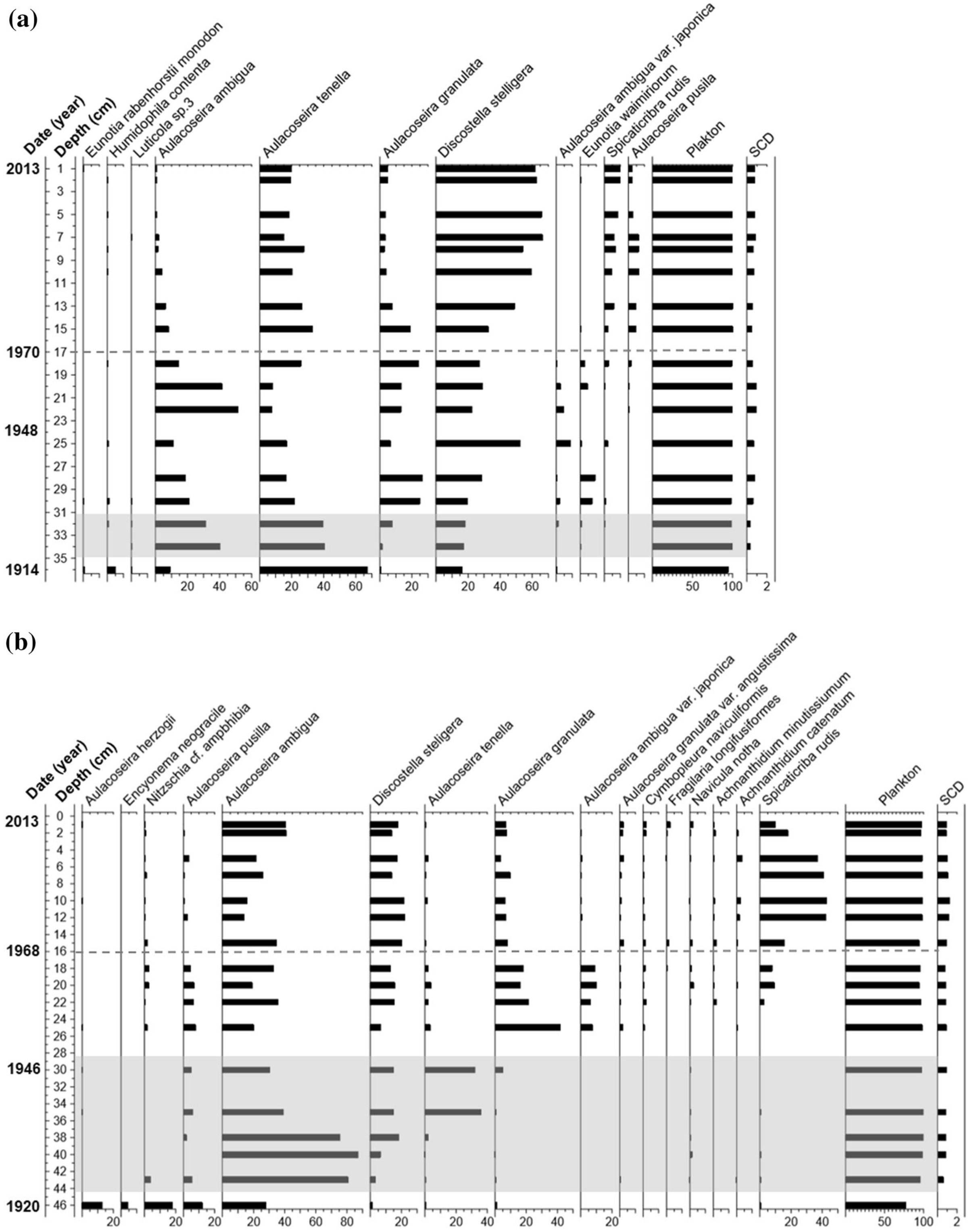


decreased. In subzone 2b (∼ 1968–2013), *A. granulata* declined to less than 10% and there was an increase in the abundance of *S. rudis* (40%) and *A. ambigua* (∼ 40%). The SCD score of 0.94 between the core bottom and top samples reflected a large change in the structure of the assemblage.

Eutrophic reservoirs

A total of 118 species were identified in core SG13 from Salto Grande, with only 15 taxa occurring at ≥ 2% (Fig. 7a). The core was dominated by planktonic species (∼ 90%) and, while there were some shifts in the diatom assemblages, these were insufficient to allow identification of zones. *Aulacoseira granulata* was dominant (∼ 60%) in a large part of the core, especially at the base of the core to 8 cm (∼ 1983), declining up to the top when *A. ambigua* var. *japonica* increased to a relative abundance of ∼ 70%. Other abundant species included *Cyclotella meneghiniana* Kützing (20%), *A. granulata* var. *angustissima* (Otto Müller) Simonsen (20%) and *A. ambigua* (25%). The SCD score of 0.61 between core bottom and top samples reflects a moderate change in the assemblage structure.

A total of 138 species were identified in core RG12 from Rio Grande, with 33 taxa occurring at ≥ 2% (Fig. 7b). Cluster analysis identified three zones, two of which contained subzones. Subzone 1a (34–36 cm; ∼ 1926–1931) was relatively diverse and included *A. ambigua* and several species from the genus *Eunotia* Ehrenberg. This zone contained the largest percentage of benthic species (∼ 30%) in the core. The subzone 1b (22–34 cm; ∼ 1931–1953) was dominated by planktonic taxa, the most abundant being *A. ambigua* (65%) with lesser contributions from *A. tenella*, *D. stelligera* and *A. ambigua* f. *japonica*. In zone 2 (12–22 cm; ∼ 1953–1982), *A. pusilla*, *C. meneghiniana*, *Fragilaria* cf. *fusa*, and *Achnanthidium tropicocatenatum* Marquardt et al. increased markedly, while *A. ambigua* declined. Subzone 3a (12–8 cm; ∼ 1982–2001) was marked by dominance of *A. tropicocatenatum* (47%), increase in *D. stelligera* (∼ 30%) and the near disappearance of the principal species from zone 2. In subzone 3b (8–0 cm; ∼ 2001–2012), *A. tropicocatenatum* remained abundant and other species from the genus *Achnanthidium* Kützing, such as *A. saprophilum* (Kobayasi & Mayama) Round & Bukhtiyarova and *A. minutissiumum* (Kützing) Czarnecki increased. Conversely *D. stelligera* decreased in zone 3b. The SCD score of 1.67 between core bottom and top samples revealed a large change in the assemblage composition over the period represented by the core.


(a)
Date (year)
Depth (cm)
Eunotia rabenhorstii monodon
Humidophila contenta
Luticola sp.3
Aulacoseira ambigua
Aulacoseira tenella
Aulacoseira granulata
Discostella stelligera
Aulacoseira ambigua var. japonica
Eunotia waimiriorum
Spicaticribra rudis
Aulacoseira pusila
Plakton
SCD
2013
1970
1948
1914
1 3 5 7 9 11 13 15 17 19 21 23 25 27 29 31 33 35
20 40 60
20 40 60
20
20 40 60
50 100
2
(b)
Date (year)
Depth (cm)
Aulacoseira herzogii
Encyonema neogracile
Nitzschia cf. amphibia
Aulacoseira pusilla
Aulacoseira ambigua
Discostella steligera
Aulacoseira tenella
Aulacoseira granulata
Aulacoseira ambigua var. japonica
Aulacoseira granulata var. angustissima
Cymbopleura naviculiformis
Fragilaria longifusiformes
Navicula notha
Achnanthidium minutissiumum
Achnanthidium catenatum
Spicaticriba rudis
Plankton
SCD
2013
1968
1946
1920
0 2 4 6 8 10 12 14 16 18 20 22 24 26 28 30 32 34 36 38 40 42 44 46
20
20
20
20 40 60 80
20
20 40
20 40
20
20 40
50 100
2

◀**Fig. 6** Diatom stratigraphy of the mesotrophic reservoirs **a** Paineiras (PI13) and **b** Itupararanga (IT13). Horizontal lines define diatom zones 1, 2 and subzones 2a, 2b. The grey shadow highlights the common diatom assemblage for the mesotrophic baseline. The dashed line shows the assemblage changes after 1970

Ordination analysis

The PCoA analysis revealed four main clusters of samples (Fig. 8, table S6). Axis 1 (31.4% variance) distinctly separated the samples from the oligotrophic

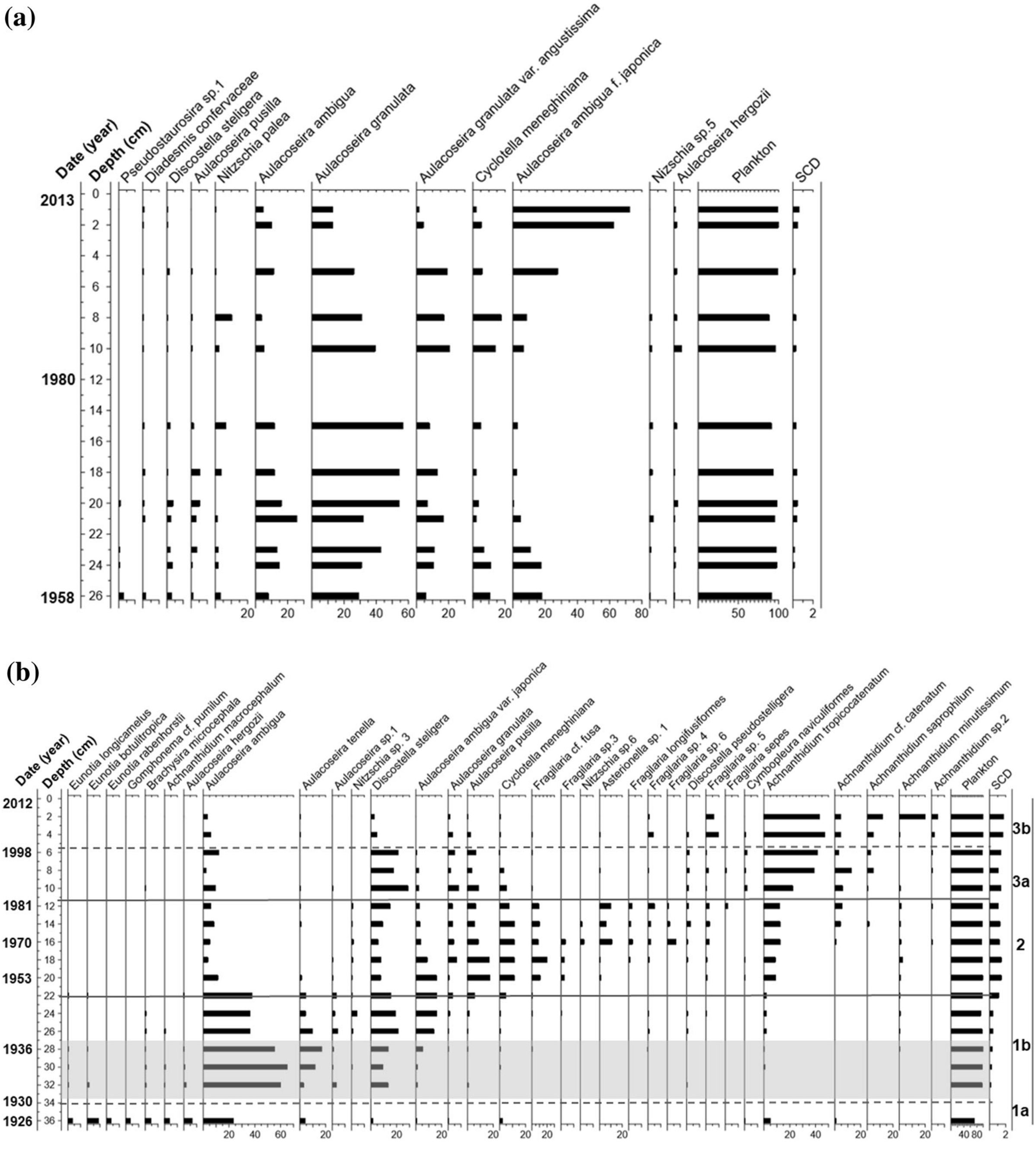


Fig. 7 Diatom stratigraphy of the eutrophic reservoirs **a** Salto Grande (SG13) and **b** Rio Grande (RG12). Horizontal lines define diatom zones 1, 2, 3 and subzones 1a, 1b, 3a, 3b. The grey shadow highlights the mesotrophic baseline

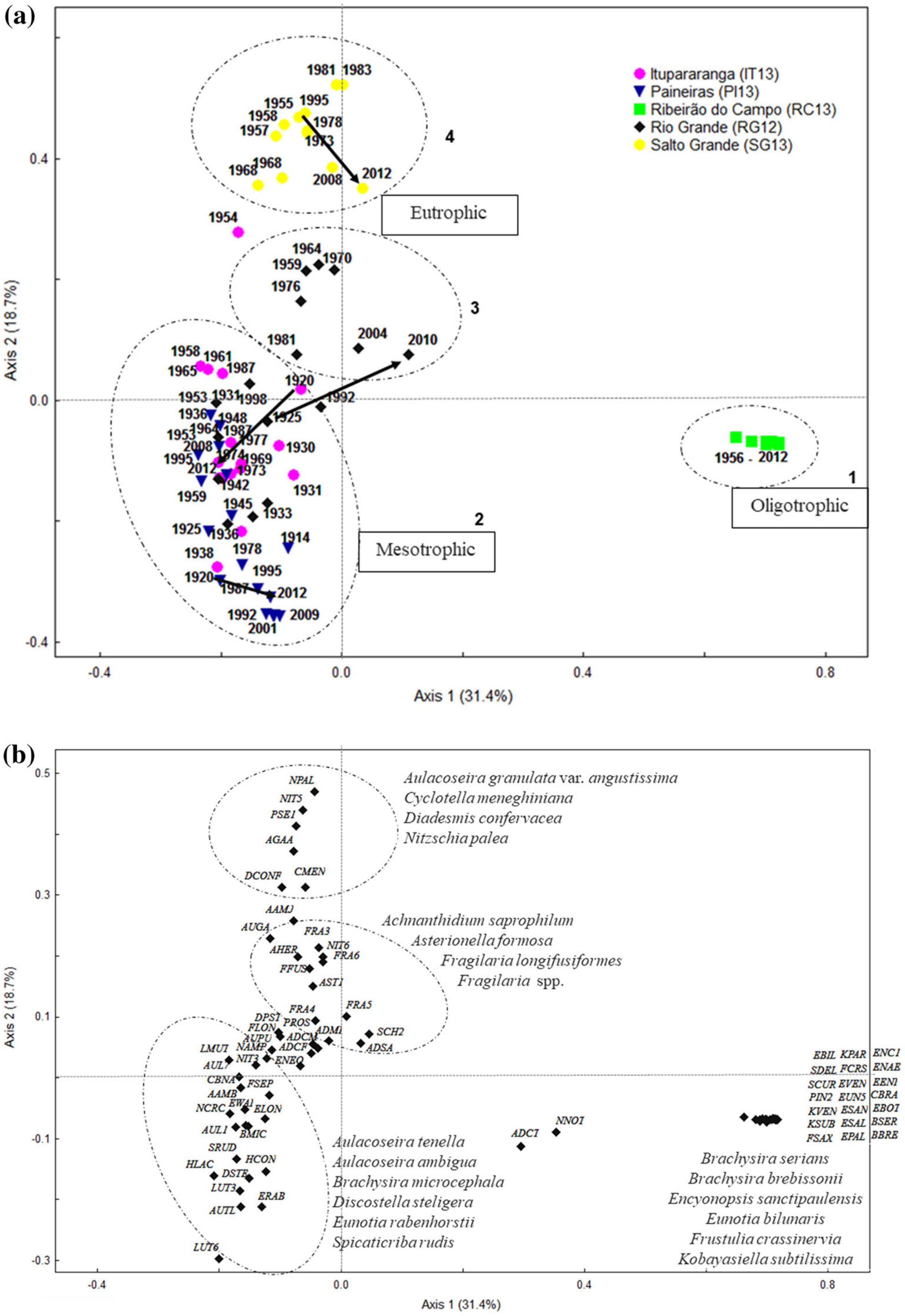


Fig. 8 Principal coordinate analysis (PCoA) of diatom assemblages (≥ 2%) for all five reservoirs (IT13, PI13, RC13, RG12, SG13). Group 1 (oligotrophic), group 2 (mesotrophic), group 3 and 4 (eutrophic). **a** Sample scores. **b** Species scores. Explanation of codes and full names for taxa are listed in ESM7

reservoir (group 1) from those of the other reservoirs. Axis 2 (18.7% variance) separated the eutrophic group 4 (Salto Grande Reservoir) and group 3 (zone 2 and subzone 3b of Rio Grande Reservoir from 1959 to 2012), with positive axis scores, from mesotrophic group 2 comprised of samples from Rio Grande (zone 1 and subzone 3a, Paineiras, and Itupararanga), with negative axis scores.

Diatom taxa associated with group 1 (oligotrophic), included *Brachysira brebissoni*, *Frustulia crassinervia*, *Encyonopsis sanctipaulensis* and species of the genus *Eunotia*. In group 2 (mesotrophic), the most common species were *A. tenella*, *A. ambigua*, *B. microcephala* (Grunow) Compère, *D. stelligera*, *E. rabenhorstii* Cleve & Grunow, and *S. rudis*. The main species associated with group 3 (meso-eutrophic), were *Asterionella formosa* Hassal, species of *Fragilaria* Lyngbye, and *Achnanthidium saprophilum*. In group 4 (eutrophic), the most common species were *A. granulata* var. *angustissima*, *C. meneghiniana*, *Diadesmis confervacea* Kützing and *Nitzschia palea* (Kützing) Smith.

Discussion

Reconstruction of the environmental history of reservoirs with different productivities revealed that the nature and extent of any changes reflected the origin of the reservoir, and the use and management of its drainage basin. Overall, the changes were reservoir-specific, with the exception of the two mesotrophic sites which bore some similarities. The use of organic geochemistry and stable isotopes allowed inference of the changes in the organic source material, while the diatoms provided an assessment of ecological change, enabling inference of trophic status across space and time.

Oligotrophic reservoir

Environmental conditions have remained relatively stable throughout the history of Ribeirão do Campo. The geochemical proxies change little across the sediment profile. High values of carbon, nitrogen, C:N ratio, and the stable isotope $\delta^{13}C$ indicate that the predominant source of organic matter was from C3 plants (Meyers 2003), possibly deriving from the Atlantic Forest, which has predominated in the drainage basin since its construction. The lowest values of $\delta^{15}N$ of all five sites were found in this reservoir, in agreement with its low contribution reported for oligotrophic environments (Torres et al. 2012), which in turn are mainly derived from the biological fixing of atmospheric nitrogen (Martinelli et al. 2009). The values of all geochemical markers (TOC, TP, C:N, and the stable isotopes ^{13}C and ^{15}N) were comparable to base levels of the Rio Grande and the Guarapiranga reservoirs, both in the same drainage basin, and which characterized the phase of flooding of the Atlantic Forest vegetation (Fontana et al. 2014). The diatom assemblage consisted of benthic taxa, typically associated with slightly acidic and oligotrophic waters, such as *Eunotia*, *Brachysira*, *Encyonopsis sanctipaulensis*, and *Frustulia crassinervia* (van Dam et al. 1994; Taylor et al. 2007). Although some species appeared toward the top, or increased in abundance, such as *Stenopterobia delicatissima* and *Chamaepinnularia brasilianopsis*, these species are also good indicators of acid waters and oligotrophic conditions (Metzeltin and Lange-Bertalot 2007; Ferreira and Bicudo 2017). As shown by PCoA, the diatom composition of this reservoir differed the most from the other cores. There are no macrophytes covering the reservoir but the species present suggest good light penetration in the water column, which fosters the growth of benthic species (Spaulding et al. 2010). An SCD score of 0.7 indicates moderate change in the diatom assemblage structure from the core base to the top. This value was greater than those reported for minimally impacted temperate lakes in Europe, which have SCD values typically less than 0.4 (Bennion et al. 2011). This larger-than-expected shift in a reservoir dominated by oligotrophic species is likely due to the greater dynamic, primarily via hydraulic forcing, that exists in reservoirs, as well as the high variability and biodiversity of tropical ecosystems, particularly where rivers are a source of species. Overall, the multi-proxy study indicates high ecological quality of this reservoir since its creation in 1958, enabling it to be considered a reference site for the other reservoirs, particularly those in the same drainage basin.

Mesotrophic reservoirs

While the stratigraphic changes in the two mesotrophic reservoirs shared some common features, the geochemical and diatom data exhibited rather contrasting trends. Organic geochemical markers show sharp variations during the reservoir constructions, which are in line with the ontogeny observed in reservoirs: the phase immediately after construction and flooding can present instabilities and greater productivity, which varies according to the region flooded (Hall et al. 1999). In contrast, the diatoms did not suggest this initial formation zone in the reservoirs. Toward the top, the geochemical values became relatively more stable and comparable to transitional zone 2 (mesotrophic conditions) in Guarapiranga (Fontana et al. 2014). Paineiras and Itupararanga were both dominated by planktonic diatom species throughout the cores. Three species were well-represented in both reservoirs, namely *A. ambigua*, *A. granulata*, and *D. stelligera*. These species are reported to have a broad distribution along the trophic gradient (van Dam et al. 1994; Stenger-Kovacs et al. 2007), albeit with *A. ambigua* and *D. stelligera* having a preference for mesotrophic waters, and *A. granulata* having a preference for eutrophic waters (Siver and Kling 1997; Houk 2003; Bicudo et al. 2016). Both reservoirs show a similar baseline, composed mainly of *A. ambigua*, *A. tenella* and *D.* stelligera, which have been reported in mesotrophic to oligotrophic and slightly acidic to neutral waters (Siver and Kling 1997; Bicudo et al. 2016). The increase of *A. granulata*, and the appearance of *A. ambigua* f. *japonica*, characteristic of eutrophic environments (Tuji and Williams 2007), along with the rise of δ^{15}N (Itupararanga reservoir), suggest a more productive environment between 1945 and 1970 for Paineiras, and 1952–1968 for Itupararanga reservoir. Decline of *Aulacoseira* Thwaites taxa and the emergence of species preferring oligotrophic and mesotrophic environments (*A. pusilla* and *S. rudis*) indicates an improvement in water quality starting in ~ 1970 for both reservoirs, with some recovery in the last four decades. Although we classified both of the currently mesotrophic reservoirs as having remained mesotrophic over time, the SCD scores of 0.78 and 0.94 suggest moderate floristic change. The emergence and decline of oligotrophic and eutrophic species through time, indicates a fluctuation over the water quality. These results could reflect biomass stabilization after impoundment (Hall et al. 1999) and changes in the catchment, especially owing to agriculture. The challenges associated with defining baseline samples to assess the extent of degradation in reservoirs will be discussed below.

Eutrophic reservoirs

Salto Grande has been described as eutrophic since the first limnological studies of the reservoir in 1969 (Tucci et al. 2004), and the results of this paleolimnological study indicate that it has been eutrophic since its construction in 1959. The rapid growth in urbanization and cultivation in the basin after 1875 influenced the water quality of the Atibaia River, the main inflowing river to the reservoir (Espíndola et al. 2004), which would certainly have contributed nutrients to the site. The TP levels, C:N ratio, δ^{13}C and δ^{15}N observed since its initial phase correspond to the eutrophic zone of Guarapiranga Reservoir (Fontana et al. 2014). The sharp increase in TP and δ^{15}N, beginning in ~ 1963 (22 cm), indicates a progressive rise in untreated urban and industrial effluent entering the reservoir as these are generally enriched with δ^{15}N (Savage 2005; Costanzo et al. 2003). The values of TP have increased since the initial construction phase and were the highest of all the reservoirs in this study. Although sedimentary P concentrations must be interpreted with caution, given mobility of P in sediments (Ginn et al. 2012), this increasing trend does suggest that a process of eutrophication began shortly after the reservoir's creation. The diatom assemblages were dominated throughout the core by planktonic species of *A. granulata* (var. *granulata* and var. *angustissima*) which are typically associated with eutrophic waters (Siver and Kling 1997; Zalat and Vildary 2007; Bicudo et al. 2016). There was a large contribution throughout the core from two others eutrophic taxa: *A. ambigua* f. *japonica* (Tuji and Williams 2007), which reached 80% toward the top, and *C. meneghiniana*, which can tolerate domestic and industrial effluent and is associated with elevated TP and hypereutrophic conditions (Costa-Böddeker et al. 2012). The diatom data therefore indicate that eutrophic conditions have prevailed since the initial phase of the reservoir. The SCD score for this reservoir was 0.61, even less than the oligotrophic reservoir (Ribeirão do Campo, 0.70), indicating the low assemblage variability over the time period represented by the core. Eutrophication normally leads to a simplification of biological communities (Jeppesen et al. 2000; Wengrat et al. 2018), acting as an environmental filter, in which only the tolerant species can persist (Chase 2007) and hence this could explain the low species turnover.

Rio Grande (Billings reservoir until 1982) exhibited large changes throughout the core with three zones and four subzones. In zone 1a (~ 1926–1931), the construction phase, the geochemistry was dominated by organic material of plant origin (C3 plants) while the C:N ratio and the δ^{13}C signals indicated vascular plants rich in cellulose and poor in proteins (Meyers 2003). These conditions were most probably fostered by the flooding of the local catchment (Atlantic Forest) during the reservoir impoundment, similarly to the Guarapiranga reservoir located in the same drainage basin (Fontana et al. 2014). In addition, low levels of δ^{15}N are commonly found in oligotrophic and mesotrophic environments (Torres et al. 2012). The initial diatom assemblages were relatively diverse with a large number of species in low abundance, primarily belonging to the genera *Achnanthidium*, *Eunotia*, *Brachysira*, and *Gomphonema*. Such genera are benthic, indicative of oligotrophic and slightly acidic environments (Metzeltin and Lange-Bertalot 2007) and adhere to substrata via mucilage (Spaulding et al. 2010). Zone 1b (~ 1930–1953), saw a gradual increase of δ^{15}N and TP along with a decline in C:N ratio indicating an input of nutrients to the reservoir, most likely supplied by the main body of the Billings reservoir, which began receiving polluted water from the Rio Pinheiros in 1940 (Capobianco and Whately 2002). Benthic diatoms were replaced by planktonic species, *A. tenella*, *A. ambigua*, and *D. stelligera*, characteristic of mesotrophic waters and observed in the stabilization phase of the mesotrophic reservoirs in this study. Zone 2 (~ 1953–1982) was considered eutrophic as a result of the direct influence of human activity causing rapid eutrophication after the 1950s. The highest values of TP in the history of the reservoir occurred in this zone, followed by a decrease in C:N ratio and enrichment of δ^{15}N. Such trends indicate elevated algal productivity (Meyers 2003) as well as untreated effluent rich in nitrogen and phosphorous (Costanzo et al. 2003). Generally, bacteria preferentially metabolize ^{14}N as it is lighter, causing enrichment of δ^{15}N (Mayer et al. 2002). This phase was characterized by eutrophic diatom species such as *C. meneghiniana* and *A. granulata*. This zone extended to 1982, when the Rio Grande branch was completely isolated from the Billings reservoir by a dam, in an effort to control the eutrophication problems (Capobianco and Whately 2002). In subzone 3a (~ 1982–2001), after isolation, both geochemical and biological markers indicated an improvement in the trophic status of Rio Grande, principally in the fall of δ^{15}N, rise of *D. stelligera* (mesotrophic), decline of *C. meneghiniana*, and disappearance of various diatom species associated with the previous (eutrophic) phase. These changes influenced the SCD scores, which declined slightly in this phase, indicating that the diatom assemblage was more similar to that of the core base. From 2001 (zone 3b), *Achnanthidium* species (*A. tropicocatenatum*, *A. saprophilum*, *Achnanthidium* sp.) became dominant during a phase marked by the application of algicides (Franklin et al. 2016). The species shifts coincided with the sharp enrichment of δ^{15}N, indicating a decline in the ecological quality of the reservoir. The growth of slum dwellings leading to an increase in sewage inputs near the Rio Grande dam (Capobianco and Whately 2002) and high nutrient inputs from hypereutrophic tributaries (Wengrat and Bicudo 2011) are both likely sources of elevated nutrients to the reservoir over the last decade.

Only the Rio Grande reservoir exhibited large changes in the geochemistry and the diatom assemblages, the latter confirmed by the high SCD score of 1.67 between the core bottom and top samples. These environmental changes reflect the location of the reservoir in the urban center of São Paulo and the management that has been carried out at the reservoir. The paleolimnological study identified three significant but distinct processes: (1) eutrophication primarily influenced by the main body of Billings Reservoir and, more recently, by the region's urbanization upstream of Rio Grande Reservoir, (2) management of the water supply via the separation of the Rio Grande branch in the 1980s, thereby forming a new reservoir with improved water quality and (3) the management via application of algicides, which has likely influenced the species in the upper core. The eutrophication of Rio Grande Reservoir, already evidenced in the 1950s, shows that the process began ~ 20 years earlier than previously reported for the other two reservoirs in the region (Costa-Böddeker et al. 2012; Fontana et al. 2014).

Management implications

The potential of palaeolimnological data for identifying reference conditions and establishing restoration targets for lakes has been well recognized (Bennion et al. 2011). However, adopting a similar approach for reservoirs requires some caution and this study suggests that the concept of reference conditions in the sense of Bennion et al. (2011) cannot be easily applied to reservoirs for a number of reasons. Firstly, given the young age of reservoirs (usually $\leq$ 100 years), it is not possible to guarantee the absence of nutrient inputs related to human activities at the time of impoundment. Additionally, owing to the nature of reservoir creation, the amount of flooded biomass of terrestrial material can vary greatly between reservoirs, which impacts directly on the productivity of the initial phase. However, the fact that reservoirs do not have easily definable reference conditions does not mean that baselines to inform recovery strategies cannot be determined. Palaeolimnological approaches can, therefore, make a valuable contribution to reservoir management which is of particular importance in the Brazilian context where rivers are increasingly being impounded.

We determined that the studied reservoirs have unique environmental histories due to the construction history and the nature and extent of hydrological and environmental changes in the catchment, which reflect the use and management of the drainage basin, the greater water level fluctuations and the direct influence of the inflowing rivers. This contrasts with the common patterns seen in temperate reservoirs after impoundment, related to changes in aquatic productivity and community composition (Hall et al. 1999). In their synthesis, Hall et al. (1999) report an increase in phytoplankton production for the first 5 to 20 years, followed by a variable period (3–30 years) of reduced production and water quality improvement. However, in some cases, temperate reservoirs did not conform to this pattern and phytoplankton standing crops were observed to decline after impoundment (Hall et al. 1999). Some studies in Brazilian reservoirs ($\geq$ 1960) have reported changes in biota with impoundment, such as an increase in floating macrophytes and marked change in fish assemblages (Thomaz and Bini 1999; Agostinho et al. 2016). Nevertheless, long-term impacts of inundation on tropical reservoir productivity and biota remain poorly understood.

There are clearly challenges associated with defining baselines for reservoirs against which the extent of degradation can be assessed. Nonetheless, when the data from all five Brazilian reservoirs were compared, a coherent pattern in the diatom assemblages emerged, reflecting the trophic gradient. The diatom assemblages prior to eutrophication in the three mesotrophic environments were characterized by abundant planktonic species and the co-occurrence of three species (*Aulacoseira tenella*, *A. ambigua*, and *D. stelligera*), sometimes with *Sicaticriba rudis*. For the pre-enrichment assemblage of the oligotrophic environments, the species found were not the same among sites, although some similarities emerged related to common genera (e.g. *Eunotia*, *Brachysira* and *Frustulia*) and higher diversity of benthic species with low abundances. Finally, eutrophic environments were characterized by abundant planktonic species from three main taxa (*C. meneghiniana*, *A. granulata* var. *granulata* and *A. granulata* var. *angustissima*).

We offer the following recommendations for the use of palaeolimnological studies in the management of reservoirs based on the findings of our study, some of which will have relevance to reservoirs elsewhere:

- Baseline/reference conditions: The baseline conditions of a reservoir are difficult to define and given the marked variation in the geochemical data during initial flooding, the first few years of a reservoir's life are too unstable to act as reference conditions. Based on the evidence here, the baseline conditions are best represented by a period some years (5 to 10 years) after reservoir construction.
- Assessing the degree of ecological change and recovery: The squared chord distance (SCD) provides a valuable tool to measure the degree of ecological change, and to assess how far the ecological conditions have deviated from those of the baseline. Moreover, the SCD score can be used to assess whether an ecosystem is recovering or moving to another state following management. For example, after Rio Grande was isolated from the contaminated waters of Billings Reservoir (in an attempt to improve water quality) the SCD scores declined, indicating that the diatom assemblage was more similar to the baseline conditions and some degree of recovery had thus taken place.

- Natural variability: The higher than expected natural variability in the diatom assemblages of the oligotrophic reservoir suggests that not all ecological changes are driven by anthropogenic factors and this is an important consideration when addressing management questions. Further palaeolimnological studies of oligotrophic systems are recommended to assess the degree of natural variability.
- Management at the river basin scale: The recognition that reservoirs may have been enriched since their creation, such as Salto Grande Reservoir, has important implications for the setting of restoration goals and brings new knowledge to reservoir management. The high-water quality of the inflowing rivers and the control of the area to be flooded, reducing the biomass of terrestrial material, may be key to ensuring the ecological quality of a reservoir. The protection of the wider drainage basin from unsustainable urbanization and agriculture is of utmost importance to management strategies.
- Evaluating past productivity: The strength of combining geochemical and diatom data to infer the trophic history of reservoirs is illustrated in this study and we advocate the use of several indicators. The addition of zooplankton, invertebrates and pigment (e.g. chlorophyll-a and cyanobacteria toxins) analysis would provide further information and we encourage similar studies to adopt a multi-indicator approach to better understand wider ecosystem changes in reservoirs since their construction.

This study makes a valuable contribution to better understanding the nature of conditions prior to enrichment and the extent of ecological change in tropical waters. The identification of diatom assemblages indicative of a range of positions along the eutrophication gradient, and of a common assemblage prior to the eutrophication phase in three of the studied reservoirs, provides a valuable tool for monitoring and managing tropical reservoirs. Measures of diatom turnover were shown to be useful in evaluating the degree of change in the diatom assemblage between a sample prior to eutrophication and the present day. The use of such an approach for measuring recovery has great potential and could be applied to reservoirs undergoing management elsewhere to assess success of restoration strategies.

Acknowledgements This study was funded by FAPESP (n° 2009/03950-4) and CNPq (n° 140550/2012-7) as part of SW thesis at the Instituto de Botânica, São Paulo, Brazil; Acquased Project (FAPESP n° 2009/53898–9) and CNPq for DCB (n° 310404/2016). The authors also thank MSc. William de Queiroz (Universidade de Guarulhos) for providing the illustration of the study area, to all the students and technicians from the Institute of Botany involved in field and laboratory work, as well as SABESP for their support with sampling.

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