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**Abstract**

Human activities and the unplanned occupation of land in aquatic basins can drive cultural eutrophication, which leads to the degradation of water quality, an increase in the concentrations of nutrients, and the depletion of biodiversity. This chapter will describe the basic concepts of trophic status, ecological changes, and alterations to the composition of freshwater biodiversity, as well as current threats and global perspectives of these issues. Additionally, selected case studies from Brazilian freshwater using benthic macroinvertebrates in urban reservoirs and cyanobacteria in semiarid lentic ecosystems are discussed.

**Keywords**

Eutrophication · Human alteration · Biodiversity loss

**5.1 Introduction**

Throughout the last few centuries, human use of freshwater resources for a variety of purposes has resulted in the degradation of aquatic ecosystems (Tundisi and Matsumura-Tundisi 2003). Increasing human populations and the expansion of industrial and agricultural activities have been important driving factors for the rapid deterioration of freshwater ecosystems. These impacts have resulted in water bodies with poor water quality and limited potential uses (Straskraba and Tundisi 1999).

The unplanned human occupation of aquatic basins can deteriorate water quality and limit the quantity and availability of freshwater resources, although the scale of this impact can vary with the economic and social organization of

a particular region (Tundisi et al. 1991). Human activities can also often result in a considerable increase in nutrient concentrations in aquatic ecosystems, especially nitrogen and phosphorus, which can lead to the process of cultural eutrophication (Callisto et al. 2004).

Cultural eutrophication, which follows a clear sequence of cause and effect, can disrupt the homeostasis of lake ecosystems and result in an imbalance between the production, consumption, and decomposition of organic matter (Esteves 2011). Thus, algae and macrophyte blooms are often observed in eutrophic systems and can be responsible for the diminished multiple uses of lentic ecosystems. Additionally, a bloom of primary production is one of the main difficulties in the treatment of water for human consumption or industrial utilization (Tundisi et al. 1993). These ecological changes, including altered physical and chemical characteristics, can lead to changes in biological communities and a drastic loss of freshwater biodiversity (Barbosa et al. 1998; Maberly et al. 2002; Callisto et al. 2004, 2005).

Water quality biomonitoring programs commonly use benthic macroinvertebrates as bioindicators of environmental pollution, in addition to utilizing a suite of chemical, microbiological, and toxicological tests of water quality (Valdovinos and Figueroa 2000; Callisto et al. 2001). Benthic

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macroinvertebrate communities have frequently been used in assessments of human impacts on freshwater (Rosenberg 1992; Junqueira et al. 2000; Brooks et al. 2001; Fleituch et al. 2002). The use of bioindicators to assess water quality is based on the response of the organisms to changes in their environment, including perturbations of human or natural origins, and reflects their lifestyle and sedimentary behavior (Bonada et al. 2006).

There are many benefits of using benthic macroinvertebrates for biomonitoring programs: (a) they are easily collected and identified at the family or genera levels; (b) many taxa are sedentary and possess long life cycles, making them capable of registering cumulative effects on their habitats, and (c) they are sensitive to physical and chemical changes in ecosystems, and their responses to these alterations are detectable and measurable (Barbour et al. 1996).

There are a variety of benthic organisms that are frequently highly abundant in reservoirs; these organisms include Chironomidae, Oligochaeta, Bivalvia, Hirudinea, and Chaoboridae (Takeda et al. 1990; Brooks et al. 2001; Martins-Montanholi and Takeda 2001; Santos et al. 2006). Chironomidae larvae exhibit a high diversity of ecological features and are capable of surviving in many different environmental conditions, where some species are tolerant or sensitive to low oxygen concentrations, extremes of temperature, pH, salinity, and trophic (Prat et al. 1992; Henry 1993; Cranston 1995).

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## 5.2 Basic Concepts

### 5.2.1 Lake Aging

Lake aging is a natural process that occurs on a geologic timescale and results in the lake being filled with materials that are eroded and transported by tributaries, deposited from the atmosphere and produced in the lake. Differences in runoff and watershed characteristics cause lakes to fill-in at different rates. The didactic natural succession is from lake to pond, pond to marsh, marsh to meadow, and meadow to dry land.

### 5.2.2 Brazilian Lacustrine Ecosystems

In Brazil, most lacustrine ecosystems are characterized as lagoons owing to their geomorphological origin related to the meanders of large rivers, coastal lagoons near the sea in the “restinga” formation, or in the floodplains of Amazonia and the Pantanal. Some deep lakes can be found in the Minas Gerais State in southeastern Brazil that originated in the curves of the Doce River (Barbosa et al. 1999).

### 5.2.3 Anthropogenic (Cultural) Eutrophication

Human land use, deforestation, and untreated sewage disposal are common causes of eutrophication in freshwater. Originally, eutrophication was known as the increased productivity of a lake as it ages. Often, an increased nutrient supply from human activities results in an increase in the biological production of the lake. Although the increased production may increase the rate of lake filling, it is incorrect to define eutrophication as lake aging. A lake does not die when it reaches a state of high productivity, but rather when it has been completely filled in and no longer exists. Lake filling results from production that occurs within the lake, which may increase with eutrophication, and from the deposition of organic and inorganic materials from outside the lake, which is not related to lake eutrophication.

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## 5.3 Too Much of a Good Thing

Natural eutrophication is a fairly slow and gradual process that typically occurs over a period of many centuries as natural disturbances cause an imbalance between production and consumption within the lake, and the lake slowly becomes overfertilized. Although natural eutrophication is not rare in nature, it does not occur frequently or quickly. However, cultural eutrophication has become so common that the word “eutrophication” has come to mean a very harmful increase and acceleration of nutrient concentrations within a water body. The situation is as if something receives too much fertilizer or has too much of a good thing.

### 5.3.1 What is Meant by Trophic State?

In temperate regions, the term “trophic” originally referred to the nutrient status of an ecosystem. Esteves (1988) discussed the misuse of the lake typology approach in tropical regions and argued that it is necessary to consider the naturally high nutrient concentrations in tropical lakes. His paper, titled “Considerations on the application of typology of temperate lakes to tropical lakes,” stressed the inadequacy of its application. The metabolic patterns of tropical lakes are completely different than temperate lacustrine ecosystems. When the basic indicators used in the typology of temperate lakes are applied to tropical lakes, the resulting classification may cause the same ecosystem to be placed into different categories.

Thienemann (1913) was the first author to use the distribution of benthic chironomid larva to classify lakes in Germany and, later, to propose his typology. In his studies of volcanic lakes in the Eifel region, he discovered a different

distribution of *Tanytarsus* and *Chironomus* genera in lakes. The *Tanytarsus* genus was found to occur at a higher density in lakes with oxygen that was well distributed throughout the water column during the summer period of thermal stratification. On the other hand, the *Chironomus* genus was abundant in deeper waters with lower oxygen levels during the same stratified period. Thienemann called the former category of lakes “alpine lakes” because of their similarity to lakes in the Alps, and the latter category “Baltic lakes” because they were similar to lakes in the Baltic zone in northern Germany. As Esteves (1988) described, the Eifel lakes may be considered to be the origin of all modern typology.

Almost one decade later, Naumann (1921) used phytoplankton and concentrations of phosphorus, nitrogen, and organic matter in the water and sediments to classify Swedish lakes. This researcher used the same classification scheme that was used for soils during that period and proposed the term “oligotrophic” to describe lakes with low nutrient concentrations of P, N, and organic matter and low densities of algal populations. Additionally, Naumann applied the term “eutrophic,” which was typically used to describe fertile soils, to lakes with high concentrations of these characteristics. Further studies by Thienemann (1928) and Naumann (1930) classified the alpine lakes as oligotrophic and the Baltic lakes as eutrophic.

Most tropical lakes are shallow and show circulation patterns that are different from those in temperate lakes. These patterns, coupled with the higher temperatures, result in a unique metabolism in tropical lakes. In general, lakes are divided into three trophic categories: oligotrophic, mesotrophic, and eutrophic. The prototypic oligotrophic lake is a large deep lake with crystal clear water and a rocky or sandy shoreline. Both planktonic and rooted plant growth are sparse and the lake can support a coldwater fishery. A eutrophic lake is typically shallow with a soft and mucky bottom. Rooted plant growth is abundant along the shore and into the lake and algal blooms are not unusual. Water clarity is generally poor and the water often has a tea color. If the lake is deep enough to thermally stratify, the bottom waters are often devoid of oxygen. A mesotrophic lake has an intermediate trophic state with characteristics between the other two.

## 5.4 Current Threats

Although the response of any given water body to environmental changes will be unique, as some lakes are relatively resistant to change and others are more sensitive, one or more of the following factors can potentially reduce biodiversity in a eutrophic ecosystem (<http://www.strategy.sebio-diversity.org.uk/pages/eutrophic-lakes.html>):

- *Climate change:* A substantial change to the supply of water to an aquatic ecosystem would alter the character-

istics of that water body, whereas an increase in temperature would produce wide-ranging effects, including accelerated plant growth.

- *Pollution:* Pollutants such as organic and inorganic fertilizers and nitrogen-rich gases can enter a water body from both point sources and diffuse sources. Increasing concentrations of these pollutants can cause nutrient enrichment (eutrophication) that can damage plant and animal communities. In general, pollution inputs from diffuse sources are greater than inputs from point sources.
- *Changes in land cover:* These changes can potentially result in the release of nutrients from soil that can then cause enrichment of water bodies. In addition to causing an increase in pollution, the long-term effects of such land use changes can also include siltation, which can smother fish spawning sites and damage aquatic vegetation. These problems are exacerbated by the removal of vegetation and reed swamps at the edges of a water body, which can act as nutrient sinks and effective barriers against particulate matter.
- *Water extraction:* Water may be extracted, either directly from a standing water body or from surface feeders or aquifers, for use as a potable supply or in industry or irrigation applications. Removing water from the system can decrease water levels and result in increased retention times and reduced flushing rates. This may exacerbate nutrient enrichment, cause deterioration of marginal vegetation through drawdown and cause shallow lakes to dry out. For coastal sites, a reduction in the throughput of fresh water could increase the salinity of a water body.
- *Fishing:* The introduction of fish, the removal of predators, and the manipulation of existing fish stocks for recreational fishing can lead to the loss of natural fish populations and may affect plant and invertebrate communities. Heavy stocking of bottom-feeding fish such as carp (*Cyprinus carpio*) can increase turbidity levels and accelerate the release of nutrients from sediments. This effect has been shown to cause major enrichment problems in some eutrophic water bodies.
- *Recreation:* The use of standing water for recreational and sporting purposes may disturb existing bird populations. Marginal vegetation may be negatively impacted by trampling and the action of boat hulls and propellers can destroy aquatic plants and stir up sediments, which can contribute to enrichment and algal growth. Additionally, the construction of marinas and other leisure facilities may destroy valuable habitat and can lead to increased pollution.
- *Nonnative plants and animals:* The release of nonnative plants and animals can be very damaging to aquatic ecosystems. As an example, the introduced signal crayfish (*Pacifastacus leniusculus*) has destabilized the biota of some water bodies by consuming large amounts of

aquatic vegetation and it has eliminated many populations of native crayfish by spreading crayfish plague.

## 5.5 International Perspective

Cultural eutrophication, which is the excessive growth of plants caused by anthropogenic nutrient enrichment, is recognized as the primary problem facing most surface waters worldwide (Smith and Schindler 2009). The main driving factors of cultural eutrophication are anthropogenic changes to land use in the catchments and nutrient inputs from untreated sewages (Tundisi and Matsumura-Tundisi 2003). For example, clearing forested catchments causes the long-term loss of nutrients from the landscape, and these nutrients can end up in the local water bodies. Applications of manure or commercial fertilizer further increase terrestrial nutrient exports. Fertilized soils can become nutrient saturated and these nutrients can leak into receiving waters for decades after the external nutrient additions are reduced or discontinued. The main potential effects of cultural eutrophication caused by excessive inputs of phosphorus and nitrogen into fresh and coastal waters are as follows:

- Increased biomass of phytoplankton and macrophyte vegetation.
- Increased biomass of consumer species.
- Shifts to bloom-forming algal species that might be toxic or inedible.
- Increased biomass of benthic and epiphytic algae.
- Changes in species composition of macrophyte vegetation.
- Increased incidence of fish kills.
- Reductions in species diversity.
- Reductions in harvestable fish and shellfish biomass.
- Decreases in water transparency.
- Taste, odor, and drinking water treatment problems.
- Oxygen depletion.
- Decreases in the perceived esthetic value of the water body.

## 5.6 Eutrophication and Infectious Disease Risk

There is a direct link between eutrophication and disease risk. Water-related diseases are a major cause of human morbidity and mortality worldwide, and recent evidence suggests that diseases can have a major impact among aquatic organisms. Clearly, biological waste disposal activities, such as manure applications to cropland, can simultaneously increase the loading of phosphorus, nitrogen, and potentially hazardous coliform bacteria to surface waters. However, enhanced nu-

trient loading may be sufficient on its own to influence the abundance, composition, virulence, and survival of pathogens that are already present in aquatic ecosystems. Another potential nutrient-pathogen interaction involves changes in food quality. For example, if eutrophication influences the nutrient content of food consumed by host organisms, then changes in host nutrition could alter host-pathogen dynamics and ultimately result in infection. These potential links between pathogens and nutrient availability could also be important for managing human health in streams and rivers with significant bathing-related activity. Recreational use of waters can spread fecal-oral viruses (e.g., enteroviruses, hepatitis A viruses, rotaviruses and others) that cause a broad range of gastrointestinal, respiratory, eye, nose, ear, and skin infections.

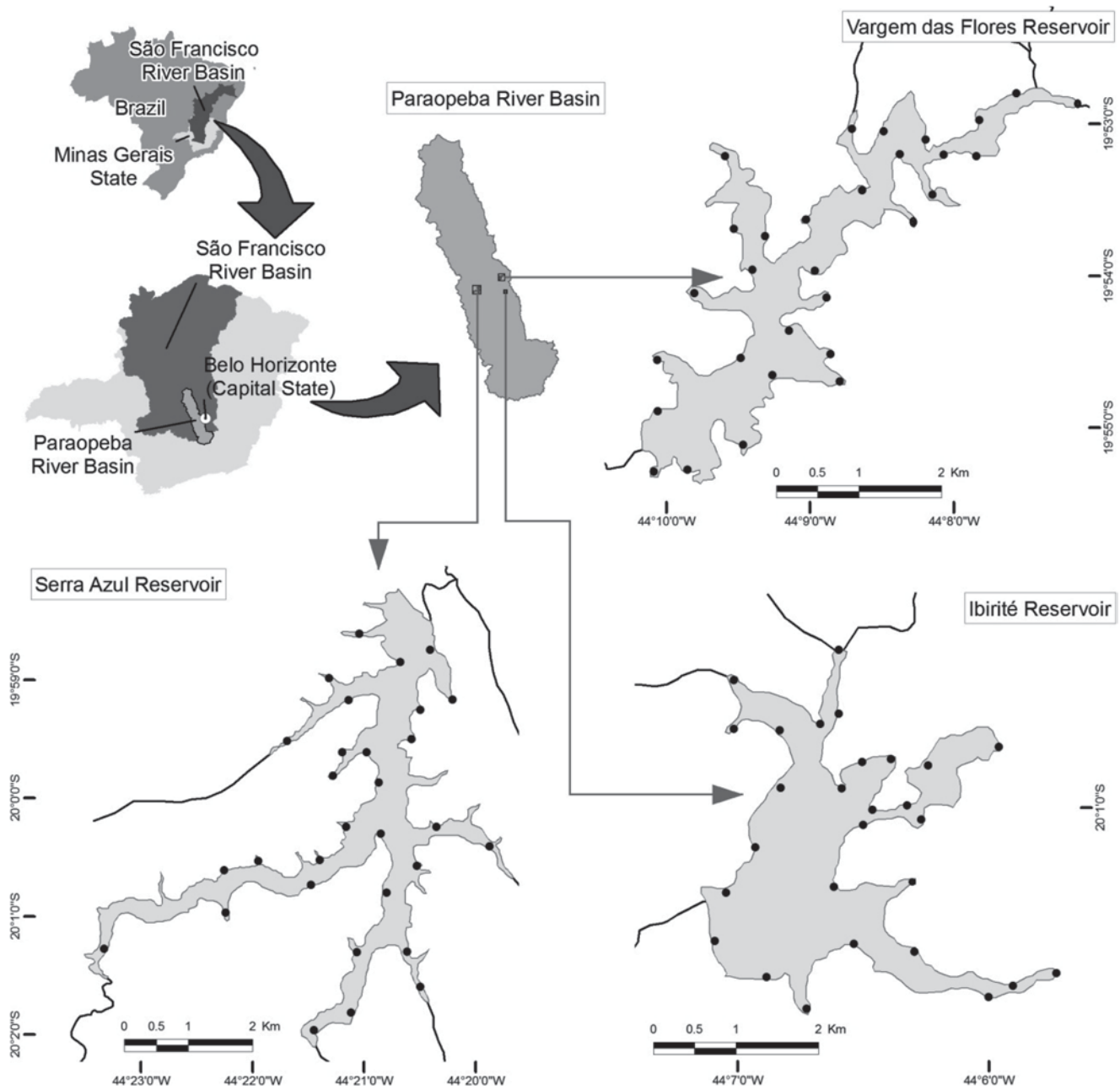
## 5.7 Eutrophication of Brazilian Freshwaters

### 5.7.1 A Case Study of Reservoirs

The Ibitité reservoir (19°07'00"–20°02'30"S, 44°07'30"–44°05'00"W) was built in 1968 at an altitude of 773 m a.s.l. This reservoir has an area of 2.8 km<sup>2</sup>, a water volume of 15,423,000 m<sup>3</sup>, and an average depth of 16 m. Most of the hydrographic basin of the Ibitité Reservoir spans the municipalities of Ibitité (148,535 inhabitants) and Sarzedo (23,282 inhabitants). The landscape of the reservoir basin is dominated by Eucalyptus plantations, a large condominium complex, small farms, and several industrial plants (Pinto-Coelho et al. 2010; Fig. 5.1).

The Vargem das Flores reservoir (19°53'30"–19°55'25"S, 44°07'22"–44°10'59"W) was built in 1971 and is situated at 838 m a.s.l. The reservoir has a water surface of 5.5 km<sup>2</sup>, a water volume of 44,000,000 m<sup>3</sup>, and a maximum depth of 18 m. The maximum level sill spillway is 838.64 m with a hydraulic retention time of 365 days. Approximately 12.3 ha of the Vargem das Flores reservoir basin was designated as an environmentally protected area (EPA) by law 16.197/06 (Minas Gerais, Brazil; Fig. 5.1).

The Serra Azul reservoir (19°54'09"–20°00'52"S, 44°23'16"–44°30'20"W), which has been operating for approximately 30 years, is located at an altitude of 760 m a.s.l., has a water surface of 8.9 km<sup>2</sup>, a water volume of 93,000,000 m<sup>3</sup>, and a maximum depth of 40 m. The maximum level sill spillway is 760 m with a hydraulic retention time of 351 days. This reservoir, together with the Vargem das Flores reservoir, provides the primary drinking water supply to the metropolitan region of the State's capital (ca. 4.8 million people). The protection area of the Basin is 27,200 ha, and the territory's domain Companhia de Saneamento de Minas Gerais (COPASA) is 3,200 ha (COPASA 2000; Fig. 5.1).



**Fig 5.1** Location of the Vargem das Flores, Serra Azul, and Ibitité reservoirs in the catchment of the Paraopeba River, Minas Gerais, Brazil and the distribution of the sampling sites (*black dots*) within the reservoirs

### 5.7.2 Macroinvertebrate Sampling

Macroinvertebrate samples were collected from 90 sites in the littoral zone of the three reservoirs. Samples were collected quarterly over a period of 2 years (in March, June, September, and December of 2008 and 2009) with an Ekman-Birge dredge (0.0225 m<sup>2</sup>). The material collected was fixed in 70% formaldehyde and subsequently identified to the family or genus level, as was the case of Chironomidae (Peterson 1960; Merritt and Cummins 1996; Mugnai et al. 2010; Trivinho-Strixino 2011).

### 5.7.3 Abiotic Data

The following physical and chemical parameters of the surface water were measured using a YIS model Multiprobe for each sampling occasion at each sampling site: electrical conductivity, turbidity, total nitrogen (TN), and pH. Additionally, groundwater samples were collected with a Van Dorn-type cylinder for subsequent measurement of total phosphorus (TP) and orthophosphates (PO<sub>4</sub>) for use in the trophic state index (TSI), following the “Standard Methods for the Examination of Water and Wastewater” (APHA 1992). The

concentration of chlorophyll *a* (Chla) was obtained according to Golterman et al. (1978) and the transparency was estimated using a Secchi disc (S). Additionally, large seasonal variations of temperature and rainfall were evaluated during the sample period. The average monthly values of temperature and precipitation were calculated for all the sampling periods based on data from the Brazilian National Institute of Meteorology (INMET) for the metropolitan region of Belo Horizonte in 2008 and 2009.

The Carlson (1977) TSI, modified by Toledo et al. (1983), was calculated for each of the sites. Each index is composed of subindices, which are then weighted to obtain a final value of the trophic status. The TSI is calculated with the following formula:

$$TSI = TSI(S) + 2 \times \left[ \frac{TSI(TP) + TSI(PO_4) + TSI(Chla)}{7} \right]$$

where the subindices are defined as follows:

$$TSI(S) = 10 \times \left( 6 - \left( \frac{0.64 + \ln S}{\ln 2} \right) \right),$$

$$TSI(TP) = 10 \times \left( 6 - \left( \frac{\ln \left( \frac{80.32}{TP} \right)}{\ln 2} \right) \right),$$

$$TSI(PO_4) = 10 \times \left( 6 - \left( \frac{\ln \left( \frac{21.67}{PO_4} \right)}{\ln 2} \right) \right),$$

$$TSI(Chla) = 10 \times \left( 6 - \frac{2.04 - 0.695 \ln Chla}{\ln 2} \right).$$

Based on this calculation, TSI values ranging from 0 to 44 correspond to oligotrophic waters; values from 44 to 54 are mesotrophic waters, and a value of greater than 54 are eutrophic waters.

#### 5.7.4 Macroinvertebrate Sampling

In total, 14,425 organisms, belonging to 47 taxa (4 Mollusca, 2 Annelida, and 41 Arthropoda), were collected from the 90 sampling sites during the 2-year observation period. Of the total number of organisms sampled, 24% were Diptera larvae, where *Chironomus* (8%), *Tanyptus* (4%), and *Coelotanyptus* (4%) were the most representative genera.

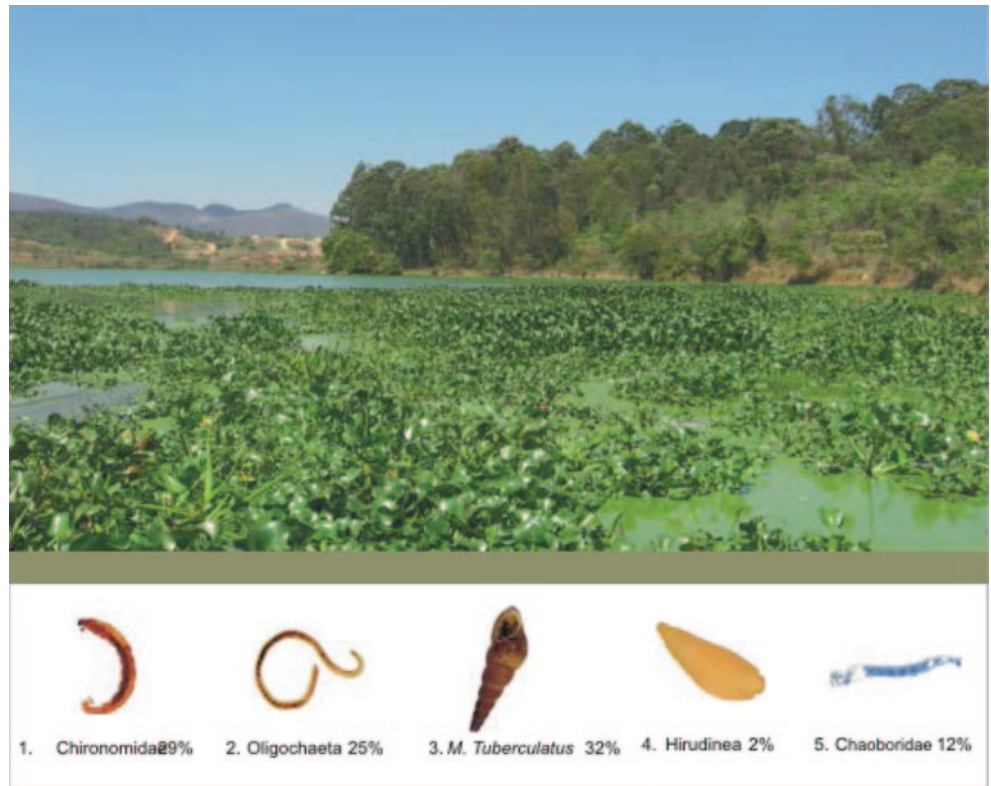
The lowest numbers of organisms were found in the oligotrophic Serra Azul reservoir (2,438 organisms), followed by the Vargem das Flores reservoir (mesotrophic, 5,033 organisms) and Ibirité reservoir (eutrophic, 6,954 organisms). Following the construction of a reservoir, great changes can be expected in the physical and chemical characteristics of the water and in the functional and structural composition of aquatic communities, including a reduction of the total number of species and the establishment of exotic species (Yanling et al. 2009).

The colonization of highly modified new habitats, as in the case of reservoirs, is typically undertaken by highly resistant species that are adapted to stagnant waters, as well as generalist species that are small in size, have long life cycles, and have high rates of sexual maturation (Rueda et al. 2006; Ruse 2010). In our reservoirs, even in the selected sites with oligotrophic conditions, the observed taxa richness (51 taxa, 59% Diptera) was lower than on the river in the same drainage basin where 63 taxa were recorded, with Ephemeroptera, Plecoptera, and Trichoptera (EPT) representing 16% of the total individuals (A. Lessa, unpublished data). In our study, the presence of the exotic species *Melanoides tuberculatus* (Müller 1774; Thiaridae, Gastropoda) was recorded at sites with oligotrophic characteristics. Since it was first recorded in Brazil in 1967, this African-Asian species has extensively invaded tropical freshwater ecosystems and settled in various types of substrates (Dudgeon 1989; Clementes et al. 2006). The densities of *M. tuberculatus* in disturbed habitats are likely to increase and may surpass the level of 10,000 ind m<sup>-2</sup> (Santos and Eskinazi-Sant'Anna 2010). Additionally, differences were found in the taxa composition of oligotrophic, eutrophic, and mesotrophic sites. Oligochaeta, including the above-mentioned *M. tuberculatus* and *Chironomus*, represented 60% of the total individuals in the more disturbed sites, while they only accounted for 7% of the total individuals in the oligotrophic sites (Fig. 5.2). In the Ibirité eutrophic reservoir, chironomids represented 29% of the samples.

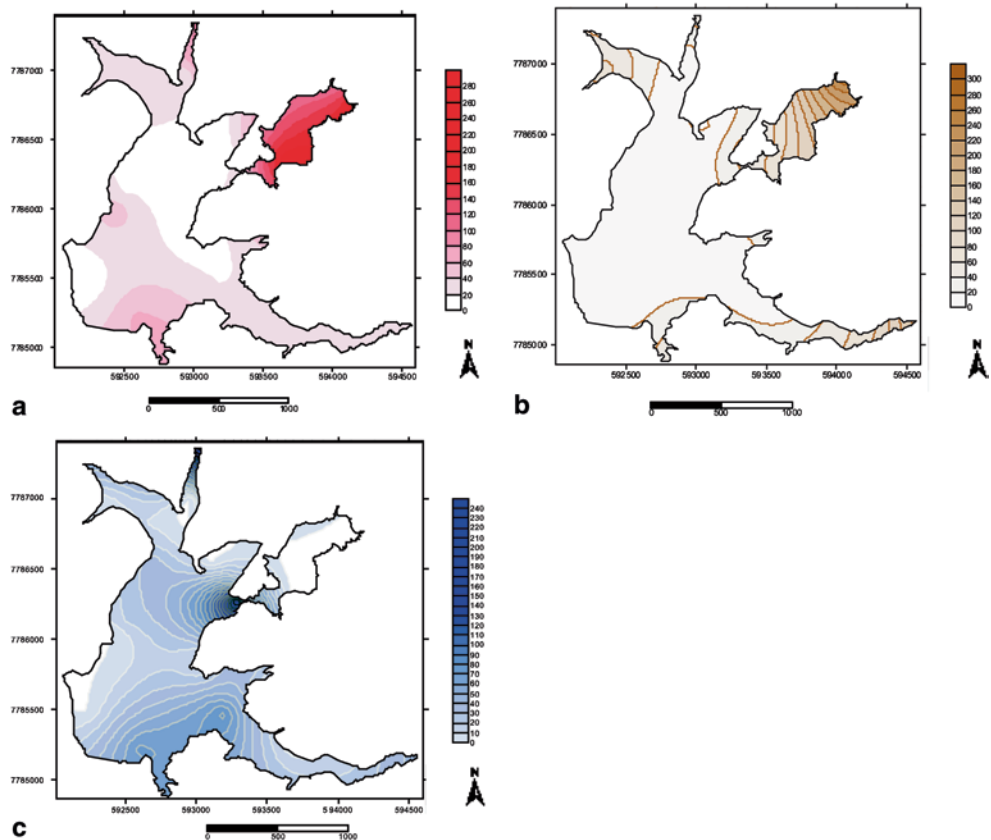
Some genera of Chironomidae were found only in oligotrophic sites (*Manoa*, *Pseudochironomus*, *Stenochironomus*, *Zavreliella*, *Lauterboniella*, *Paralauterboniella*), which is a good indication that these sites may serve as reference sites for an ecological quality assessment of reservoirs in tropical areas.

Several authors have shown that different chironomid species have different sensitivities to stress (Davies and Jackson 2006; Arimoro et al. 2007; Roque et al. 2010). For example, the genus *Fissimentum*, which was observed with a high number of individuals in oligotrophic sites, is considered an indicator of good water quality (Cranston and Nolte 1996). An evaluation of the spatial distribution of taxa showed that when both Oligochaeta and Chironomidae had the highest occurrence of organisms (ind m<sup>-2</sup>) in a reservoir arm, this was potentially characteristic of an input of untreated sewage waste from the surrounding intense urban occupation (Fig. 5.3).

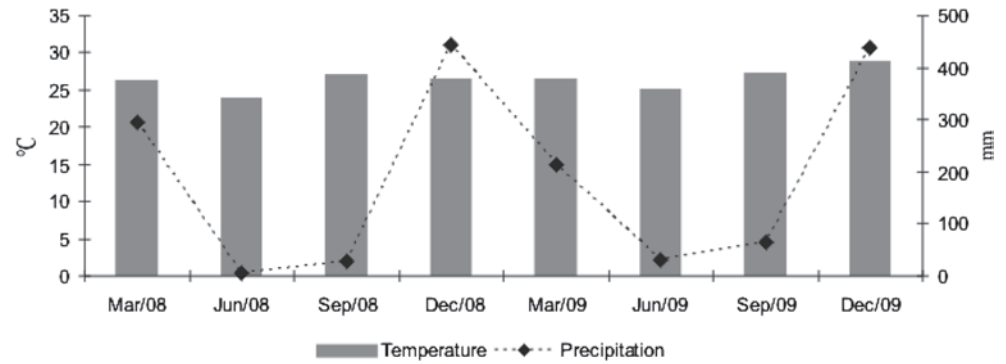
**Fig. 5.2** Percentage of organisms found in the main Ibirité reservoir: 1, 2, and 4 are resistant taxa, 3 are alien species, and 5 are typical organisms of lentic environments



**Fig. 5.3** Spatial distribution of organisms sampled in the main Ibirité reservoir: **a** Chironomidae, **b** Oligochaeta, and **c** *Melanoides tuberculatus*



**Fig. 5.4** Average monthly precipitation (mm, *bars*) and temperature ( $^{\circ}\text{C}$ , *dots*) observed during the sampling periods



**Fig. 5.5** Results of the 2nd-STAGE MDS for the three reservoirs based on biological data collected in December (Dec), March (Mar), June (Jun), and September (Sep) of 2008 (08) and 2009 (09) for the **a** Serra Azul, **b** Vargem das Flores, and **c** Ibirité reservoirs

### 5.7.5 Abiotic Data

The climatic data for 2008 and 2009 confirmed the existence of two distinct seasons: a wet season from December through March and a dry season from June through September. December 2008 had the highest rainfall average (442 mm), followed by January 2009 (282 mm). The driest months were June 2008 and June 2009, when no precipitation occurred. The maximum temperature during the study period was re-

corded in December 2009 ( $29.0^{\circ}\text{C}$ ) and the minimum was recorded in June 2008 ( $23.9^{\circ}\text{C}$ ; Fig. 5.4).

The 2nd-STAGE nMDS was consistent with the above findings, showing that there was no pattern of high correlation between the communities sampled in the same month of the year (e.g., December 2008 and December 2009) or the same season (dry and wet; Fig. 5.5). For rivers in both temperate and tropical regions, seasonal climate variability is accompanied by changes in the communities (Sporka



et al. 2006; Leunda et al. 2009; Puntí et al. 2009). Several previous studies (e.g., Feio et al. 2006; Aroviita et al. 2010) have shown that these changes are known to affect ecological assessments based on reference conditions that represent the systems only for a given season. However, the seasonal variability of temperature and precipitation observed during the 2 years of sampling in this study was not reflected in the benthic communities of the reservoirs. Seasonal variability in the communities was unpredictable and similar to the interannual variability. Other authors studying both subtropical systems (China) and temperate systems (Canada) have observed that rainfall and flood pulses did not influence the distribution of Chironomids in reservoirs because they are well adapted to fluctuations in the water level (Zhang et al. 2010; Furey et al. 2006).

The ANOSIM global  $R$  values for the three reservoirs showed wide variability within the sampling periods (ANOSIM Serra Azul: Global  $R=0.054$ ,  $p=0.001$ ; ANOSIM Ibirité:  $R=0.166$ ,  $p=0.001$ ; and ANOSIM Vargem das Flores:  $R=0.113$ ,  $p=0.001$ ). According to the TSI, 29 sites were classified as oligotrophic in the Serra Azul reservoir, and only one site was classified as mesotrophic. The Vargem das Flores reservoir had 26 sites classified as oligotrophic, with 1 site as mesotrophic and 3 sites as eutrophic. Over the 2 years of samples in the Ibirité reservoir, 28 sites were identified as eutrophic and 2 sites were mesotrophic (Fig. 5.6).

The environmental variables were found to have higher values in the eutrophic reservoir than in the oligotrophic reservoir. For example, the median electrical conductivity was 280.50 and 26.36  $\text{mS cm}^{-1}$  for the eutrophic and oligotrophic reservoirs, respectively. The exception was for the S, which had the highest average values in the oligotrophic reservoir (see Table 5.1). The decrease in diversity of the benthic communities in the Ibirité reservoir can likely be explained by an increase in the trophic status owing to the land use and occupation of the areas surrounding this reservoir. The spatial distribution of some of the environmental variables showed that one arm of the reservoir had high concentrations of nutrients (TN and high levels of TN/TP; Fig. 5.7).

### 5.7.6 Semiarid Lentic Ecosystems

Hydrological fluctuations can cause significant changes to lakes and reservoirs in arid and semiarid regions, where both seasonal and annual variability can cause changes in the shape and size of the ecosystems (Sánchez-Carrillo et al. 2007). The reservoirs in these regions can suffer significant intraannual variations of water volume, surface area, water residence time, and depth, which may affect their physical, chemical, and biological features.

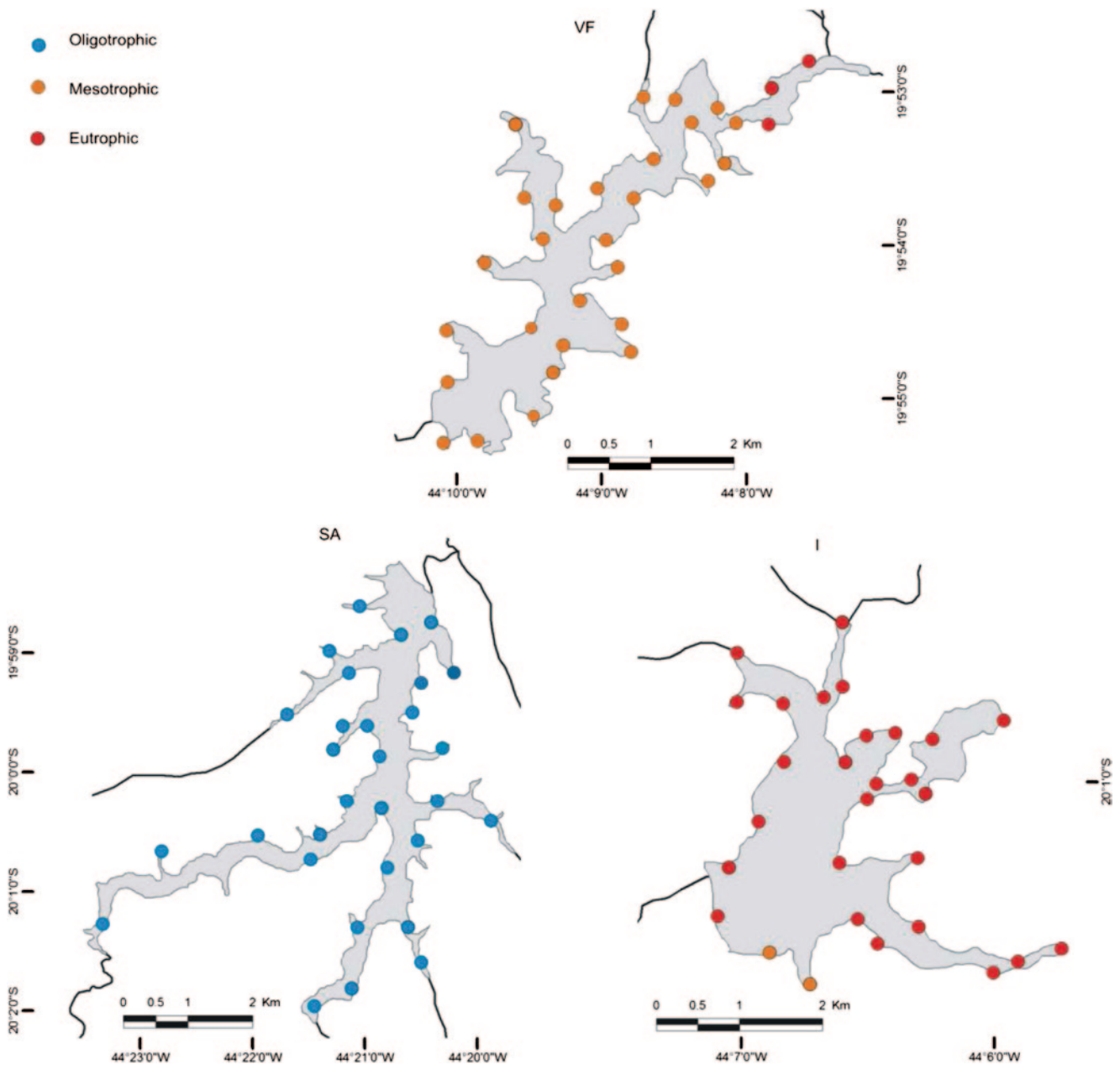
In lakes located in temperate regions, phosphorus is the main limiting resource of primary productivity and the concentration of phosphorus in the water is correlated with algae biomass (Dillon and Rigler 1974; OECD 1982). On the other hand, Thornton and Rast (1989, 1993) showed that reservoirs from semiarid regions respond differently to eutrophication than do lakes in temperate regions, from which the classical concepts of eutrophication were developed. Thus, the primary functions that influence the limnological dynamics may not be similar between these two regions, and, therefore, system responses to nutrient enrichment may be different (Thornton 1990; Tundisi et al. 1990).

Because the TSI was developed in temperate lakes, its application to tropical reservoirs has been widely questioned (Bouvy et al. 2000; Huzsar et al. 2000; Costa et al. 2006; Panosso et al. 2007). Thornton and Rast (1993) proposed that concentrations of TP above  $60 \mu\text{g l}^{-1}$  and Chla above  $12 \mu\text{g l}^{-1}$  are indicative of a eutrophic reservoir in semiarid regions. This designation is set at the point where the algal biomass would be limited more by light than by the concentration of phosphorus.

A typical phenomenon in eutrophic lakes is the occurrence of blooms of cyanobacteria, which are common planktonic organisms living in different aquatic ecosystems. Although they are primarily related to eutrophication (Watson et al. 1997), their distribution is not limited to high-productivity ecosystems (Reynolds 1997). Cyanobacteria blooms are globally distributed and are regulated by many environmental variables linked by geographical and ecological aspects (Ferrão-Filho et al. 2009).

Cyanobacteria blooms affect water quality by changing the pH, transparency, and biodiversity, and by producing odors and/or toxins, such as cyanotoxins (Blahová et al. 2008). Toxic blooms of cyanobacteria are common in man-made lakes in the semiarid regions of Brazil and they frequently cause human health threats (Molica et al. 2005; Costa et al. 2006; Vasconcelos et al. 2011).

Although cyanobacteria blooms in reservoirs in semiarid Brazil are mainly related to the eutrophic conditions of these systems, other factors may also contribute to the success of cyanobacteria. These factors include environmental constancy, annual rain deficit and lack of water renewal, high temperatures, high pH, low N/P ratio, low ammonium concentrations, and the absence of efficient predators (Bouvy et al. 1999, 2000; Huszar et al. 2000, 2006; Barbosa et al. 2010; Dantas et al. 2010).



**Fig. 5.6** Classification of sites according to the TSI index for the Vargem das Flores (VF), Ibirité (I), and Serra Azul (AS) reservoirs during 2008 and 2009

## 5.8 Case Study of the Trophic Evolution of a Reservoir Used to Supply the Citizens of the Paraíba State in the Semi-arid Region of Brazil

### 5.8.1 Site Description

The water resources of the Paraíba State are distributed throughout 11 watersheds (Fig. 5.8), which contain more than 123 reservoirs. These reservoirs hold more than 15 million m<sup>3</sup> of water intended for use as the public sup-

ply and they provide a maximum storage capacity of nearly 43,906,773,462 m<sup>3</sup>. The 20 reservoirs evaluated in this study are located between 6°48' to 8°12'S and 38°33' to 34°47'W (Fig. 5.8) and are present in two different hydrographic basins. The Paraíba and Piranhas River basins are the largest basins in the Paraíba State, and they represent the highest potential for water accumulation (28 and 67%, respectively).

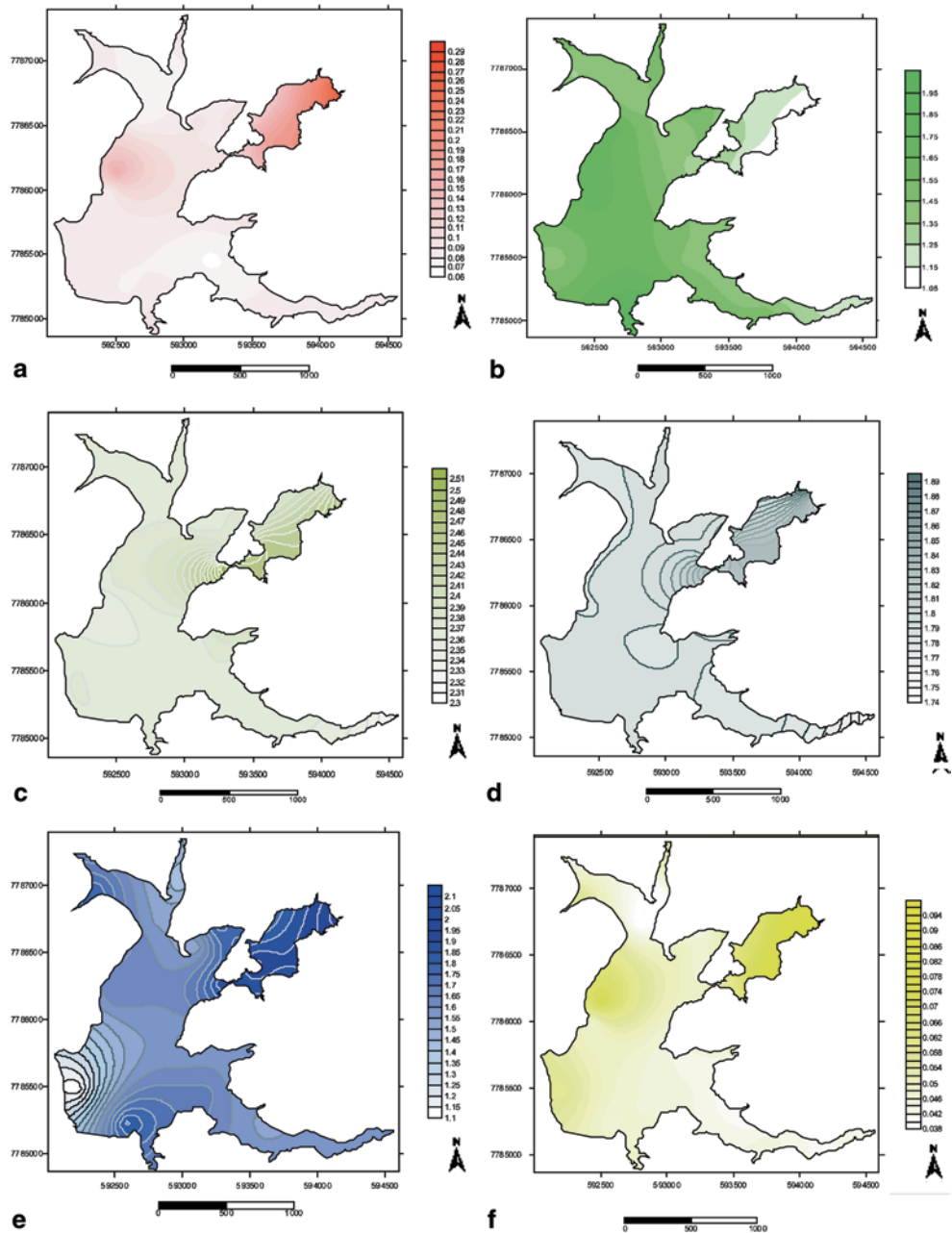
The climate classification of the reservoirs is BSw, according to Köppen's classification system, which describes a semi-arid hot climate with 7–9 dry months. The average temperatures are between 28 and 33 °C and the precipitation

**Table 5.1** Environmental variables sampled in the oligotrophic, mesotrophic, and eutrophic reservoirs

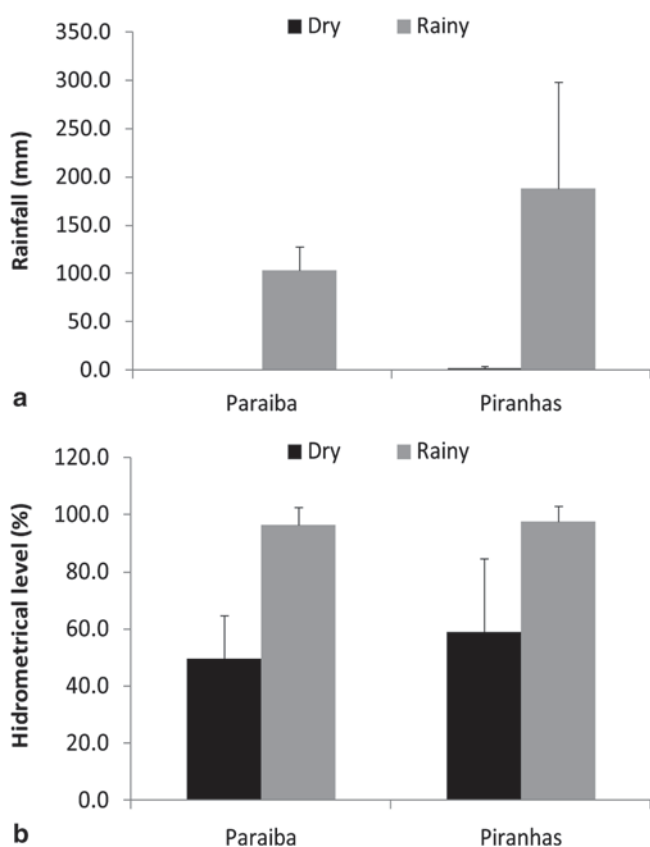
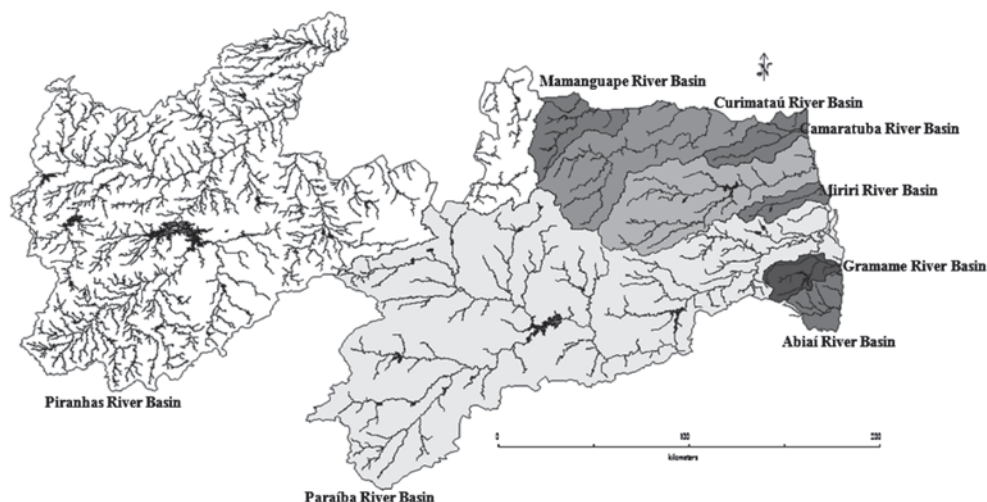
Environmental variables	Oligotrophic	Mesotrophic	Eutrophic
Secchi disc (m)	2.34±0.86	1.4±0.7	0.47±0.27
Water temperature (°C)	27.98±3.65	25.9±2.57	27.5±3.65
pH	7.69±0.44	8.13±0.50	8.33±0.98
Electrical conductivity (mS cm <sup>-1</sup> )	26.34±8.66	135.94±27.36	280.50±114.46
Total dissolved solids (mg l <sup>-1</sup> )	18.45±5.21	91.68±27.33	197.43±70.64
Turbidity (NTU)	5.05±10.29	15.48±28.06	55.78±66.71
Dissolved oxygen (mg l <sup>-1</sup> )	7.72±0.81	7.63±1.16	7.00±0.21
Chlorophyll a (µg l <sup>-1</sup> )	2.05±3.44	2.34±2.21	53.65±78.66
Total nitrogen (µg l <sup>-1</sup> )	0.05±0.04	0.22±0.25	0.29±0.21
Total phosphorus (mg l <sup>-1</sup> )	23.16±20.39	23.19±20.41	171.71±260.64
Orthophosphates (µg l <sup>-1</sup> )	7.21±5.12	7.90±5.82	22.62±86.78

The values represent the mean (±SD) of the 2-year sampling period

**Fig. 5.7** Spatial distribution of some of the environmental variables in the Ibirité reservoir: **a** total nitrogen (µg l<sup>-1</sup>); **b** chlorophyll a (µg l<sup>-1</sup>); **c** electrical conductivity (mS cm<sup>-1</sup>); **d** total dissolved solids (mg l<sup>-1</sup>); **e** turbidity (NTU); and **f** ratio of total nitrogen to total phosphorus



**Fig. 5.8** Map of Paraíba State (Brazil), including the different hydrographic basins, and the locations of the sampled reservoirs



**Fig. 5.9** Rainfall (a) and hydrometric levels (b) of the reservoirs during the dry and rainy seasons in the Paraíba and Piranhas River basins, Paraíba State, northeastern Brazil

is approximately 700 mm/year. In all the reservoirs, the dry periods occurred between August and March. These periods included low precipitation levels (40–280 mm), and the hydrometric levels of the reservoirs were generally low as well. During the rainy seasons, from May to July, the precipitation was higher (150–450 mm) and the reservoirs reached their maximum capacity. The wind velocity in the region was

typically negligible and ranged from 3 to 4 m s<sup>-1</sup>. The total annual evaporation varied between 2,500 and 3,000 mm with decreasing values trending from west to east.

Each of the 20 reservoirs was sampled eight times between August 2006 and August 2009, including sampling events in both the dry and rainy periods. During the dry periods, we observed low rainfall in the Piranhas (X=2.1 mm) and Paraíba (X=0.2 mm) River basins. The hydrometric levels of the reservoirs were generally low as well. In the Piranhas River, the hydrometric level of the reservoirs varied from 9.9 to 91.2% of their total capacity, while in the Paraíba River, reservoirs ranged from 36.9 to 73.8% (Fig. 5.9b). During the rainy seasons, the precipitation was higher in Piranhas River basin (X=187.4 mm) than in the Paraíba River basin (X=103.5 mm; Fig. 5.9a). During this period, 76% of the reservoirs reached their maximum capacity in the Piranhas River basin, and only 40% reached their capacity in the Paraíba River basin (Fig. 5.9).

### 5.8.2 Sample Collection

Environmental and phytoplankton samples were collected from one permanent site at the deepest part of each reservoir near the dam by holding a Van Dorn Bottle 10–20 cm beneath the water surface. Light transparency was assessed using an S. Temperature, pH, and electrical conductivity were measured using specific electrodes. Dissolved oxygen was measured by the Winkler method. Water samples were collected in PVC bottles that were previously cleaned with distilled water. The samples were transported to the laboratory on ice, where they were frozen and analyzed for nutrient concentrations (including inorganic dissolved nitrogen, dissolved reactive phosphorus, and TP) according to the procedures described by APHA (1992).

**Table 5.2** Limnological measurements in the reservoirs

Variables	Piranhas River					Paraíba River				
	Dry		Rainy			Dry		Rainy		
	Mean	CV (%)	Mean	CV (%)	<i>p</i>	Mean	CV (%)	Mean	CV (%)	<i>p</i>
Water temperature	29.9	6.72	29.57	8.93	0.74	28.2	6.49	28.2	3.96	0.93
Dissolved oxygen	7.95	13.4	7.008	11.5	0.01*	7.82	7.53	8.44	23.3	0.51
Alcalinity	20.4	24.4	17	28.7	0.09	29.2	24.5	18	18	0.01*
Inorganic dissolved nitrogen	7.49	212	75.07	121	0.01*	2.3	31.4	122	112	0.08
P-ortho	11.80	63.2	33.31	72.1	0.01*	70.2	71	92.6	64.8	0.53
Total-P	168.11	58.11	2.492	135	0.19	1,681.1	58.11	91.02	27.4	0.24

The mean and coefficient of variation (CV) are presented for the two sampling periods as well as the significance level of the differences between the dry and rainy season for Piranha and Paraíba Rivers (*p*)

\**p* < 0.05

### 5.8.3 Data Analysis

Relationships between environmental parameters and the density of cyanobacteria were studied using multiple regression analysis. The Akaike's information criterion (Johnson and Omland 2004) was used to extract the variables that significantly increased the amount of explained variation ( $\Delta AIC < 2$ ).

An analysis of variance (ANOVA) indicated significant differences between the rainy and dry seasons for the reservoirs in the Piranhas River basin for dissolved oxygen, inorganic dissolved nitrogen, and TP. For the reservoirs in the Paraíba River basin, only alkalinity was found to have a significant seasonal difference (Table 5.2).

Differences of limnological factors were observed between the reservoirs in the Piranhas and Paraíba River basins, except for TP and total alkalinity (Table 5.2). In the dry season, the Piranhas River reservoirs showed higher mean values for temperature, dissolved oxygen, and dissolved inorganic nitrogen. Higher mean concentrations of TP were registered in the Paraíba River reservoirs. During the rainy season, the Paraíba River reservoirs had higher concentrations of dissolved inorganic nitrogen and TP (Table 5.2).

Reservoirs in semiarid regions are considered to be eutrophic when the annual average concentration of TP is higher than  $60 \mu\text{g l}^{-1}$  (Thornton and Rast 1993). During the sampling period, an increase in the concentrations of TP in the reservoirs was observed, with averages ranging from  $39.3 \mu\text{g l}^{-1}$  (DP = 12.76) in 2006 to  $155.89 \mu\text{g l}^{-1}$  (DP = 81.16) in 2009. In 2006, 7.7% of the reservoirs were categorized as eutrophic and that percentage increased in each subsequent year, with 30.8% in 2007, 42.3% in 2008, and 57.7% in 2009 (Fig. 5.10).

We identified 188 taxa in the reservoirs, which were distributed among nine taxonomic groups: Chlorophyceae (61), Cyanobacteria (49), Bacillariophyceae (34), Euglenophyceae (14), Zygnemaphyceae (19), Dinophyceae (5), Chlamydomophyceae (4), Xanthophyceae (1), and Oedogoniophy-

ceae (1). No individual taxon was found to be present in all the sampled reservoirs. However, *Cyclotella meneghiniana*, *Cylindrospermopsis raciborskii*, *Aulacoseira granulata*, *Aphanocapsa elachista*, and *Pseudoanabaena limnetica* were the most frequently identified species. Out of the total number of identified taxa, 39 occurred only in the dry period and 30 were found only during the rainy period. *Aphanizomenon tropicalis* and *Coelastrum scabrum* were the most abundant of the 16 taxa that were found only in the Paraíba River, and *Dolichospermum spiroids*, *Merismopedia tenuissima*, and *Oocystis lacustris* were the most abundant of the 31 taxa found only in the Piranhas River.

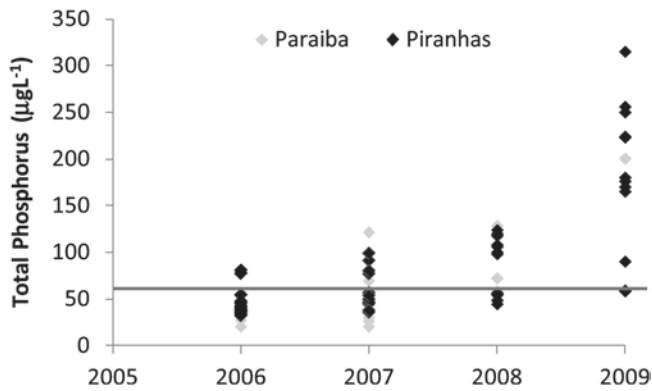
The most abundant species of cyanobacteria in the reservoirs were potential toxin producers. These species dominated the phytoplankton community in 16 of the reservoirs, especially during the dry periods, and represented 73.1% of the total individuals and 54.08% of the total algae during the rainy season. The occurrence of cyanobacteria blooms in reservoirs increased significantly between 2006 and 2009. In 2006, only 3% of the reservoirs were found to have cyanobacteria blooms, while in 2007 that number grew to 20%, in 2008 it was 45%, and in 2009 it reached 62%. Most of the observed cyanobacteria blooms were found in the Piranhas reservoirs (Fig. 5.11).

A multiple regression analysis with cyanobacteria density as the dependent variable showed a positive relationship with dissolved oxygen and a negative relationship with water transparency, pH, and inorganic dissolved nitrogen ( $R^2_{\text{adjusted}} = 0.45$ ;  $p < 0.05$ ). The equation to summarize these relationships is described by

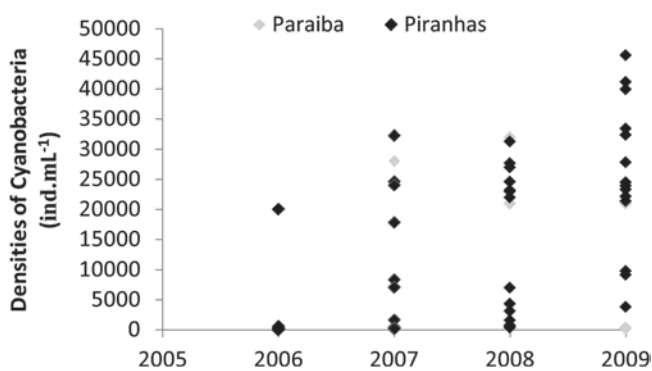
$$\ln \text{CYN} = 13 - 0.71 \ln \text{WT} - 3.5 \text{pH} + 0.29 \ln \text{DO} - 0.19 \ln \text{IDN},$$

where

CYN    Cyanobacteria density  
 WT    Water transparency  
 DO    Dissolved oxygen  
 IDN    Inorganic dissolved nitrogen.



**Fig. 5.10** Evolution of increasing phosphorus concentrations from 2006 to 2009 in the Paraíba River Basin

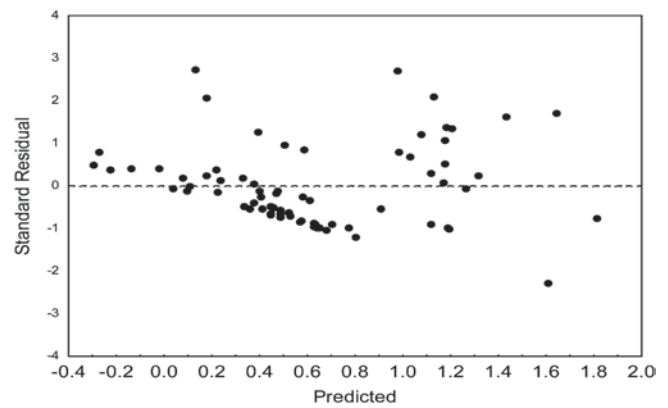


**Fig. 5.11** Occurrence of cyanobacteria blooms in the Paraíba River Basins and the evolution of cyanobacteria densities from 2006 to 2010

The proposed model to describe the cyanobacteria density in the reservoirs was found to have residuals with a normal distribution ( $W=0.96$ ;  $p=0.73$ ) by the analysis assumptions (Fig. 5.12).

The availability of phosphorus has been regarded as one of the most important factors for determining phytoplankton biomass and water quality in lakes and temperate wetlands (Dilon and Rigler 1974) of tropical and subtropical regions (Attayde and Bozelli 1999; Huzart et al. 2006). However, reservoirs have different characteristics than natural lakes. For example, they often have unique morphology and hydrodynamics, as well as high loading rates of nutrients from the sediments and the drainage basin (Wetzel 1990). Thus, the driving factors of limnological dynamics are not likely to be similar in lakes and reservoirs, and therefore, the system responses to nutrient enrichment are likely to be different (Tundisi et al. 1990).

Our models demonstrated that cyanobacteria blooms, which are a regular problem in Brazilian semiarid reservoirs (Vasconcelos et al. 2011), did not have a direct relationship with phosphorus concentrations. This result suggests that



**Fig. 5.12** Dispersion of the predicted values as a function of the residual pattern

phosphorus concentrations were not a good indicator of the trophic level of these reservoirs. The low transparency of the water was owing to high concentrations of suspended solids, which favors the growth of cyanobacteria while inhibiting other classes of phytoplankton.

The high suspended solids concentrations may have been influenced by an increase of aquaculture activities in the Brazilian semiarid reservoirs. The practice of farming fish in net cages is an important factor related to the elevated concentrations of nutrients and Chla, as well as a reduction of water transparency. The impact of fish farming on reservoirs is readily observable from the visible release of dissolved or suspended materials and metabolites from food remains (Starling et al. 2002; Lazzaro et al. 2003; Guo et al. 2009; Borges et al. 2010). These substances are associated with a high temperature, which favors an increase in the density of phytoplankton, and particularly cyanobacteria (Padišák 1997; Chorus and Bartram 1999; Reynolds et al. 1997), which may explain the high densities of cyanobacteria observed in the studied reservoirs.

Additional factors should be considered to help explain eutrophication in semiarid waters. Throughout the study period, the concentrations of TP and densities of cyanobacteria were clearly increasing. This can be explained by the increasing level of anthropogenic activities in the area and by the climatic features of the region. The long time necessary for water renewal is an important natural event that can maximize this problem. In addition to the low rainfall, other factors also influenced the eutrophication, including nutrient loads from tributaries during the rainy periods; furthermore, because renewal requires large inputs of relatively clean water, which did not occur during this period, the trophic levels of the reservoirs will likely be maintained or even worsen over time.

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