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Phytoplankton alpha diversity as an indicator of environmental changes in a neotropical floodplain

Luzia C. Rodrigues[a,c], Nadson R. Simões[b,*], Vania M. Bovo-Scomparin[a], Susicley Jati[a,c], Natalia F. Santana[a], Maria C. Roberto[a], Sueli Train[a]

[a] Núcleo de Pesquisas em Limnologia, Ictiologia e Aqüicultura, Universidade Estadual de Maringá—UEM, Av. Colombo, 5790, Bloco H-90, Sala 23, CEP 87020-900 Maringá, PR, Brazil

[b] Universidade Federal do Pará, Laboratório de Ecologia, Rua Coronel José Porfírio, 2515, São Sebastião, CEP—68372-040 Altamira, PA, Brazil

[c] Programa de Pós-graduação em Ecologia de Ambientes Aquáticos Continentais, Brazil

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ABSTRACT

Habitat fragmentation, overexploitation of natural resources, the introduction of alien species and environmental degradation in aquatic environments are the main causes of reductions in aquatic biota diversity. Phytoplankton represent good ecological indicators because they are highly diverse and rapidly respond to a wide array of environmental disturbances. We investigated the interannual variation in alpha diversity of the phytoplankton community in lakes of an alluvial floodplain. We predicted that the phytoplankton diversity decreases over time in lakes and rivers subjected to human activities, whereas those biotopes in areas under pristine environmental conditions do not show a reduction in alpha diversity. Phytoplankton samples were taken quarterly over a period of eleven years (2000–2010), from ten localities associated with three large rivers, which showed different uses of the watershed. The time series of alpha diversity was analysed, to assess the temporal trend, in addition to their relationships with environmental factors. Phytoplankton alpha diversity in the Upper Paraná River floodplain ranged between 4 and 87 species and showed a mean of 30 (±16.5). Sites associated with the Paraná River showed a decline in diversity, which was associated with transparency, nitrogen and phosphorus forms. These results reflect a combination of seston retention by damming and an increase in the N:P ratio, which appears to negatively affect phytoplankton diversity. If temporal trends in environmental variation and the phytoplankton community remain, the future consequences for phytoplanktonic diversity in the Paraná subsystem will be severe, which might cause changes in the trophic structure and dynamics, and therefore in the functioning of environments, since this community is one of the main sources of energy for other trophic levels.

1. Introduction

Current estimates have suggested that high levels of biodiversity loss are occurring (Smith et al., 1993; Pimm et al., 1995; Luck and Daily, 2003; Sax and Gaines, 2003) because the processes that maintain or increase diversity, such as evolutionary and demographic processes, have been superseded by the rapid increase in environmental impacts, such as habitat fragmentation, the overexploitation of natural resources, the introduction of alien species and environmental degradation (Tockner et al., 2002; Dudgeon et al., 2006). This reduction in diversity leads to a loss of genetic resources, functional diversity, trophic changes, ecosystem properties and services, which affect human well-being in the social sense as well as in the economic and cultural senses (Luck and Daily, 2003; Hooper et al., 2005; Millennium Ecosystem Assessment, 2005; Abson and Termansen, 2010).

The trophic position and the functional role of lost or threatened species create implications concerning the magnitude and impact of species loss in a specific ecosystem. Thus, phytoplankton species are unique because of their functional role in the production of organic carbon, functioning as a basal source of energy in many aquatic ecosystems (Lindeman, 1942; Reynolds and Descy, 1996). Moreover, phytoplankton show rapid responses to a wide array of environmental disturbances (Cottingham and Carpenter, 1998; Lepistö et al., 2004; Paerl et al., 2009) and have a high species

* Corresponding author. Tel.: +55 93 3515 1079.
E-mail addresses: luziac.rodrigues@gmail.com (L.C. Rodrigues), nadsonressye@yahoo.com.br (N.R. Simões), vaniascomp@gmail.com (V.M. Bovo-Scomparin), susi@nupelia.uem.br (S. Jati), natalia_nf@hotmail.com (N.F. Santana), mcroberto@nupelia.uem.br (M.C. Roberto), train@nupelia.uem.br (S. Train).

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diversity under natural conditions, as suggested by the paradox of plankton (Hutchinson, 1961) and textbooks (Hutchinson, 1976; Reynolds, 1984, 2006), which make this community an excellent indicator of environmental disturbance.

In floodplains, the hydrosedimentological regime is the main driving force that acts on phytoplankton (Neiff, 1990; Train Sueli Rodrigues, 1998; Paidere et al., 2007; Bovo-Scomparin and Train, 2008; Borges and Train, 2009); it controls the availability of resources (light and nutrients) and demographic processes. These ecological systems maintain a high diversity (Shiel et al., 1998; Borges and Train, 2009; Lansac-Tôha et al., 2009), due to habitat heterogeneity and hydrological temporal variability (Tockner et al., 1999; Simões et al., 2013), but they have been threatened by many environmental changes (Agostinho et al., 2004; Simões et al., 2012; Bonecker et al., 2013).

It is thus necessary to understand the features that affect species diversity in order to implement effective conservation strategies in these ecological systems, to mitigate biodiversity loss, assess ecological responses to natural and human disturbances, and detect changes in the structure and function of ecosystems (Cingolani et al., 2010; Lindenmayer and Likens, 2010; Magurran et al., 2010).

In this study, we investigated the interannual variation in alpha diversity of the phytoplankton community in aquatic biotopes with distinct hydrodynamics, in a Neotropical floodplain over eleven years, with the aim of examining whether changes in environmental variables due to human impact negatively affect phytoplankton diversity. To do this, we tested the hypothesis that different trends of alpha diversity exist between environments subjected to a greater intensity of human activities and those environments with pristine conditions. We predicted that phytoplankton diversity decreases over time in lakes and rivers subjected to human activities, whereas those biotopes in areas under pristine environmental conditions do not show a reduction in alpha diversity.

2. Material and methods

2.1. Characterisation of the study area

The Paraná River basin is the second largest in South America in terms of length and drainage area. At its upper stretch, where the alluvial valley of the Upper Paraná River floodplain is situated (22°30′–22°00′S, 53°00′–53°30′W) (Supplementary material 1), the floodplain is formed by rivers, secondary channels, backwaters, and temporary and isolated lakes (Agostinho et al., 2004), which frame three distinct subsystems (Paraná, Ivinhema and Baía) associated with the three large rivers of the region. The Paraná subsystem is directly associated with the water level of the Paraná River, which is strongly related to the operation of a reservoir located 40 km upstream (the Usina Hidrelétrica Engenheiro Sérgio Motta reservoir). The Baia River, which follows a course parallel to the Paraná River, is also influenced by the operation of this upstream reservoir, but the Baia River preserves a pristine system of wetlands. The Ivinhema area is located in a dam-free region in the Ivinhema River State Park. Therefore, this last sub-system is not affected by upstream dams. Among the sites, those associated with the Paraná River show a higher degree of degradation, whereas the Ivinhema and Baia River show pristine conditions.

The conservation of the Upper Paraná River floodplain is indispensable for Brazilian biodiversity, because this region shelters an important fraction of the original biota of the basin. Three protected areas are present in this region, indicating their relevance for biodiversity conservation; however, their ecological integrity is threatened by a series of upstream reservoirs and an inappropriate use of the watershed (Agostinho et al., 2004).

2.2. Sampling strategy

This study was carried out during eleven years (2000–2010) as part of a long-term ecological research (the Brazilian Long Term Ecological Program/CNPq), with samples taken quarterly (except in 2001 and 2003, when only two samplings were conducted), totalling 40 samples for each site and 400 samples in this study.

In each subsystem, we selected three or four sampling sites with distinct hydrodynamic characteristics (lotic and lentic) to maximise the environmental variability: one lotic environment in each subsystem (Paraná, Baia and Ivinhema Rivers), and at least two lentic environments in each subsystem. Among lentic environments, we sampled lakes perennially connected to each river (connected lakes) and lakes that remain unconnected to each river (isolated lakes) for most of the year. Particularly in the Paraná subsystem, we sampled two connected lakes (connected lakes 1 and connected lakes 2) (Supplementary material 1).

Phytoplankton samples were taken under the water surface (depth 20 cm) from the studied environments. These samples were preserved with acidified Lugol's solution and the analysis was performed according to Utermöhl (1958) and Lund et al. (1958).

Physics and chemistry variables were measured as follows: transparency (m; Secchi disc), dissolved oxygen (mg L^{-1}; YSI portable oximeter), temperature (°C; thermometer coupled to the oximeter), pH (Digimed portable potentiometer), electric conductivity ($\mu S\,cm^{-1}$; Digimed portable potentiometer), turbidity (NTU; LaMotte2008© portable turbidimeter), inorganic suspended solids and organic suspended solids ($\mu g\,L^{-1}$), total alkalinity ($\mu Eq\,L^{-1}$), total nitrogen ($\mu g\,L^{-1}$), nitrate ($\mu g\,L^{-1}$), ammonium ions ($\mu g\,L^{-1}$), total phosphorus ($\mu g\,L^{-1}$) and soluble reactive phosphorus ($\mu g\,L^{-1}$) concentrations. Details of the methods employed for obtaining limnological variables can be found in a specific limnological study of this floodplain (Roberto et al., 2009).

2.3. Data analysis

Principal component analyses (PCA) were run for each subsystem with all environmental variables (water temperature, dissolved oxygen, conductivity, pH, total alkalinity, total nitrogen, nitrate, ammonium, total phosphorus, soluble reactive phosphorus, inorganic suspended solids and organic suspended solids), to characterise the environmental variability among sites sampled within each subsystem. Data were standardised via a correlation matrix.

PCA scores were correlated with a temporal sequence (2000–2010), to indicate whether environmental gradients (axes) increase or decrease with time. We verified the temporal autocorrelation to validate the correlation test. We did not find autocorrelation in the first order, but in two lakes, autocorrelation in the second order was recorded. This was expected due to the seasonality of the floodplain. Additionally, the N:P rate, which represents limitations for the physiological state of phytoplankton (Reynolds, 2006), was also used to characterise the temporal dynamics of potential predictors of phytoplankton.

The phytoplankton alpha diversity is defined as the species richness of a site (Whittaker, 1972). Alpha diversity time series were analysed to test the temporal increases or decreases in diversity, in addition to their relationships with environmental predictors. This procedure was performed in two steps: first, the temporal increase or decrease in diversity was tested; second, models were fitted to represent the temporal diversity variation as a function of environmental predictors.

Initially, trends were fitted using mixed models with a fixed explanatory variable (T—a discrete sequence ranging from 1 to 40) and a random effect for the different years (λ). The full model was described by the equation $y = \alpha + \tau T + \lambda Y + \varepsilon$, where y is diversity, α is the intercept, τ—is the slope that shows the relationship between