Ecology of the epilithic diatom community in a low-order stream system of the Guaíba hydrographical region: subsidies to the environmental monitoring of southern Brazilian aquatic systems.

HERMANY[1], G., SCHWARZBOLD[1], A., LOBO[2], E.A. & OLIVEIRA[3], M.A.

[1] Ecology Department, Biosciences Institute. Federal University of Rio Grande do Sul – UFRGS - Av. Bento Gonçalves, 9500, Campus do Vale, Porto Alegre, RS, Brazil. guilherme.hermany@ufrgs.br; albano@ecologia.ufrgs.br

[2] Laboratory of Limnology. University of Santa Cruz do Sul – UNISC - Avenida Independência 2293, Santa Cruz do Sul, RS, Brazil. lobo@unisc.br

[3] Department of Biology, CCNE, Federal University of Santa Maria – UFSM – Av. Roraima, S/N, Campus Camobi, Santa Maria, RS, Brasil. CEP 97105-900. angelicaoliveira@yahoo.com

ABSTRACT: Ecology of the epilithic diatom community in a low-order stream system of the Guaíba hydrographical region: subsidies to the environmental monitoring of southern Brazilian aquatic systems. From the relationship between the structure of epilithic diatom communities in the Schmidt Stream - a small water course in the Guaíba hydrographical region, RS, Brazil – and changes in water quality, as well as physical alterations of the environment such as flow, riparian shading, width and depth of the channel, this study aimed to gather subsidies to be applied in aquatic ecosystem monitoring and conservation programs. Between January and February 2004, four field trips were conducted to six sampling sites, adding up to 24 samples. Analysis of the epilithic diatom community was carried out on samples scrubbed off five replicated stones, representing a total area of 125 cm². Samples were cleaned with $k_2Cr_2O_7$, H_2SO_4 and HCl prior to the assembly of permanent glass slides. Species distribution and abundance patterns were explored through the use of the Indicator Species Analysis and environmental factors influencing this configuration were singled out by Canonical Correspondence Analysis. The results confirmed the preference of the genera Cymbella and Encyonema for environments where slow current and high luminosity prevail, and the adaptability of Cocconeis to shading. The species Nitzschia acicularis and Surirella tenera were found to be good indicators of meso-eutrophic conditions. Encyonema perpusillum stood out as a representative of oligo/b-mesossaprobic habitats, while Sellaphora pupula was abundant in ion-rich environments. Comparing the species responses to oxygen saturation, BOD, COD, conductivity, total phosphorus, total dissolved solids and turbidity, two indicator groups were established.

Key-words: periphyton, natural substrata, lotic environments, biomonitoring.

RESUMO: Ecologia da comunidade de diatomáceas epilíticas de um sistema de rio de baixa ordem da Região Hidrográfica do Guaíba: subsídios ao monitoramento ambiental de ecossistemas aquáticos sul-brasileiros. A partir da relação entre a estrutura da comunidade de diatomáceas epilíticas de uma microbacia sul-brasileira, mudanças na qualidade da água e alterações físicas do meio, como fluxo, sombreamento ripário, largura e profundidade, objetivou-se colher subsídios para aplicação em programas de manejo e conservação de ecossistemas aquáticos. No verão de 2004 foram realizadas quatro excursões científicas a seis estações, totalizando 24 amostragens. Para o estudo das diatomáceas, uma área de 25 cm² foi raspada de cinco rochas submersas, obtendo-se 125 cm² por sítio. O material foi oxidado com $k_2Cr_2O_7$, H_2SO_4 e HCl para confecção de lâminas permanentes. Padrões de distribuição e abundância das espécies foram explorados através da Análise de Espécies Indicadoras e os fatores ambientais responsáveis por esta configuração foram evidenciados pela Análise de Correspondência Canônica. Os resultados confirmaram a preferência de Cymbella e Encyonema por ambientes de baixo fluxo e elevada luminosidade, bem como a adaptabilidade de Cocconeis ao sombreamento. Nitzschia acicularis e Surirella tenera revelaram-se indicadoras de ambientes meso-eutróficos. Encyonema perpusillum destacou-se no habitat oligo/b-mesossapróbico, e Sellaphora pupula no sítio de maior concentração de eletrólitos. Pelo cruzamento das respostas das espécies à saturação de oxigênio, DBO_5, DQO, condutividade, fósforo total, sólidos totais dissolvidos e turbidez, foi possível estabelecer ainda dois grupos sinalizadores da qualidade da água.

Palavras-chave: perifíton, substrato natural, ambientes lóticos, biomonitoramento.

Introduction

A thorough knowledge of the structure and function of periphytic communities is essential for a better understanding of freshwater systems; however, this sort of information in Brazil is still recent, and remains scarce (Oliveira, 1996). Periphytic communities have yet to be recognised as key-components in the metabolism of aquatic systems, mainly in small and shallow water bodies, abundant in Brazilian territory. Such lack of information might even compromise the application of conservation measures and alternatives of environmental management (Bicudo et al., 1995). Potential bioindicators are, therefore, under-explored, and studies with this approach are restricted, mainly, to Southern Brazil, emphasising epilithic diatoms (e.g. Lobo & Callegaro, 2000; Lobo et al., 1996; 2002; 2004a; 2004b; 2004c; Salomoni, 2004). Recently, in the South-East, Mourthé-Junior (2000) studied epilithic diatoms related to water quality in the Velhas River, near Belo Horizonte, capital of Minas Gerais State. Also, Souza (2002) researched the same subject in the Monjolinho River region of São Carlos, São Paulo State.

Considered as efficient information systems in environmental monitoring, explored through the analysis of structural characteristics of the communities, several auto-ecological indices – using relative abundance and ecological preferences of diatom species – have been widely developed and put into use in Europe (Stevenson & Pan, 1999). These indices are either based upon detailed characterisation of communities comprised of several taxa (Prygiel, 1991) or simple description based on few genera and species, allowing its use by non-specialists (Rumeau & Coste, 1988). Most of these indices detect organic pollution (e.g. Lange-Bertalot, 1979a) although recently, Kelly & Whitton (1995) developed a specific index for the evaluation of the trophic state of water systems (recalibrated later by Kelly, 1998).

Along the same lines, in Latin America, Gómez & Licuirsi (2001) published a regional water quality evaluation index, for rivers and streams of the Argentinean Pampa, denominated Pampean Diatom Index (IDP). This index is based upon the sensitivity of the epipelic diatom biocenosis, integrating effects of organic enrichment and eutrophication.

In Brazil, Lobo et al. (2002) developed the first saprobic system in the country, based upon the use of epilithic diatoms for water quality assessment in lotic systems of Southern Brazil, having studied 183 samples from 31 sampling sites in 18 streams and rivers of the Guaíba Hydrographical Region. This study was complemented by Lobo et al. (2004a), incorporating the eutrophication problem.

However, changes to community structure along environmental gradients have been observed in several systems and in different scales (e.g. Whittaker, 1967). In addition, several natural factors have been considered to cause variations in specific composition of river diatom communities (Katoh, 1991). Current velocity, dissolved oxygen and light intensity usually show one single environmental gradient, varying from headwaters to the lower regions, known as the longitudinal “continuum” (Vannote et al., 1980). A great number of biotic and abiotic characteristics have already been investigated along these gradients (Naiman & Sedell, 1979; Minshall et al., 1983). Consequently, mathematical indices will have little significance unless the above mentioned natural variations are taken into account. Significant changes in community composition do not necessarily confirm the hypothesis that pollution events are taking place. The correct conclusions can only be reached when species ecology and their interactions are taken into account (Patrick, 1993).

Additionally, multivariate techniques may reveal important factors and environmental gradients, not known prior to the analysis. In contrast, biotic indices are used to evaluate one previously determined factor (Lobo et al., 1995; Lobo et al., 2002). However, research on diatom communities using the multivariate approach is scarce in Brazil (e.g. Souza, 2002; Lobo et al., 2004a). Even rarer is the effective number of studies concerning community variation on less impacted sites, where natural structural variations are more evident (e.g. Miranda, 2003). Information on these aspects is of fundamental importance in order to validate the auto-ecological systems.

As well as application on environmental monitoring, information about ecological preferences of indicator species from quantitative data of present day studies, aiming to locate optimal development points and tolerance ranges along environmental gradients in lakes (e.g. Hall & Smol, 1992) and rivers (e.g. O´Connel et al., 1997; Winter & Duthie, 2000), become efficient reference sources to the study of hydrochemical changes in freshwater environments in paleolimnological research. This knowledge of ecosystem natural variability, as well as conditions previous to impact, may contribute greatly to management strategies, based on realistic goals for the recovery of areas impacted by human action (Reavie et al., 1998; Hall & Smol, 1999; Stevenson & Smol, 2002).

An important contribution to our understanding of diatom ecology has been made by Shirata (1985), Moreira Filho et al. (1990), Torgan & Biancamano (1991) and Moro & Fürstenberger (1997), who published diatom species catalogues, listing environmental requirements (such as salinity, saprobity, pH, current, habitat, trophic state and temperature). In Rio Grande do Sul State, however, quantitative studies, generating statistically tested data from epilithic diatoms, on aspects such as light, current velocity and ionic concentration are still scarce (e.g. Miranda, 2003).

Therefore, the present research aimed to investigate the longitudinal variation on epilithic diatom community structure in a low order watershed in the hydrographical region of Guaíba, Rio Grande do Sul State, Brazil, in relation to alterations in water quality as well as physical changes, such as flow, riparian shading, channel width and depth.

Material and methods

Study area

The Schmidt Stream Hydrographical Basin has an area of 50.53 km² and is located in the central region of Rio Grande do Sul State, to the East of Santa Cruz do Sul Municipal District, in a transition area between the Central Lowlands and the beginnings of the State’s highlands. Its main course extends for 14.2 km, with its source located at 52°24’25'' West and 29°39’10'' South. Its confluence with the River Taquari-Mirim is located at 52°18’20'' West and 29° 40’15'' South. The range of altitude along the stream course goes from 260 m at the main source to 59 m at its mouth (Fig. 1). The stream runs through sandstone and its bed is mainly made of cobbles. The climate is characterised as Cfa – humid sub-tropical - according to the Köppen classification (Moreno, 1961).

On 5th, 13th, and 17th January, and 3rd February 2004, field trips were carried out to six sampling stations - S1 to S6 - along the Schmidt Stream (Fig. 1), adding up to

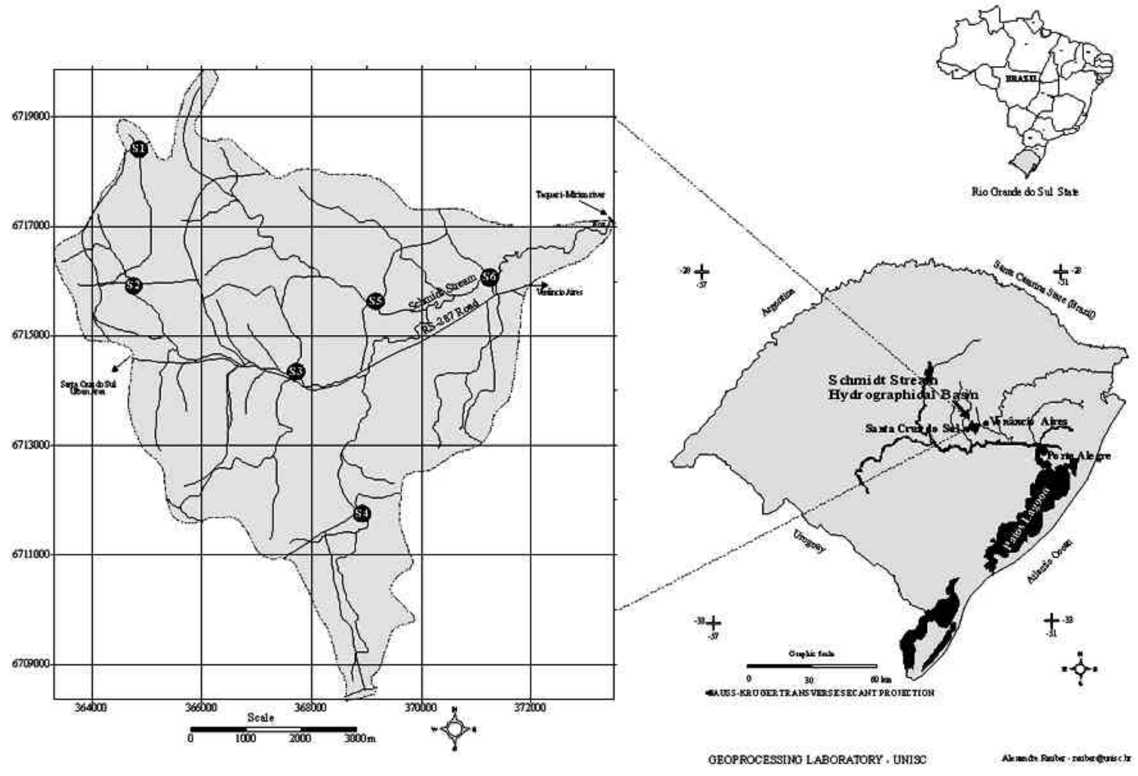

Figure 1: Map of study area with the location of Schmidt Stream hydrographical basin in Rio Grande do Sul State, Brazil, contextualized in the Guaíba Hydrographical Region, and showing the sampling sites.

24 samplings. At station S1, where the stream is of first order, the average width was 1.8 m, and average depth was 6.3 cm. At stations S2, S4 and S5 the stream is of second order, and average width equalled 3.3 m, 2.6 m, and 3.6 m, respectively. Average depth was 9.8 cm at S2, 11.7 cm at S4, and 13.8 cm at S5. Finally, at stations S3 and S6, the stream is a third order canal; average width at the former was 6.4 m, with 29.7 cm of average depth, whereas the latter had an average width of 3.2 m, and 8.5 cm of average depth.

The main activities in the surrounding area include agriculture - tobacco and rice - (S1, S5 and S6) as well as cattle farming (S1 and S4). There are also illegal dwellings located near the stream banks, along the RS-287 road near S2 and S3.

Physical and chemical analyses

The following variables were measured: electric conductivity (COND), % of O_2 saturation (% O_2), pH, turbidity (TURB), biochemical oxygen demand (BOD), chemical oxygen demand (COD), total phosphorus (TP), nitrate (NO_3^-), total dissolved solids (TDS), suspended solids (SS) and silicon (SiO_4^{-2}). Sampling and analyses techniques were carried out according to American Public Health Association (1999).

Levels of saprobity were defined after the chemical classification of Hamm (1969) and Dworski (1982), based on average values of oxygen saturation deficit (OSD), BOD and COD. The most critical value defined the final classification of the sampling site.

Other variables measured were depth of rock substrate, current velocity at the same depth where stones were collected (using a current meter for shallow waters), stone diameter, and channel width at sampling site and time. Shading due to riparian forest was subjectively calculated, according to Kawencka (1985, 1986) using a lightmeter on an open sky day, within one hour and ten minutes from the first to the last sampling site.

Sampling

At each sampling unit, an area of 25 cm² was scrubbed off the upper surface of five submerged stones using a toothbrush, following Kobayasi & Mayama (1982), Lobo (1995), Kelly et al. (1998), Miranda (2003) and Souza (2002). A 125 cm² composed sample was thus obtained from each sampling site. Selected stones were first sprayed with deionised water in order to remove sand and sediment layers, as well as diatoms associated with these substrates (Round, 1993). Samples were then treated with potassium dichromate, sulphuric acid, and hydrochloric acid, prior to the assembly of permanent glass slides using Naphrax as mounting medium. Three drops of ethanol were placed on the coverslip during the drying process, in order to avoid clustered distribution of cells (Round, 1993).

In order to estimate species density as the number of valves per square centimetre, the resulting numbers of cells were multiplied by the conversion factor given by Souza (2002) and Lobo (1995):

$Cf = [(N_0/N_1) * (VO_0/VO_1) * (V_0/V_1) * (1/A_0)]$, where:

N_0 = total number of transects on the slide;
N_1 = number of transects counted on the slide;
VO_0 = original sample volume, obtained from the scrubbed stones;
VO_1 = volume of the sub-sample, used in the cleaning process;
V_0 = volume of the final sample, diluted and cleaned;
V_1 = volume of sample used to mount the glass slide;
A_0 = original stone area where the sample was obtained.

This strategy was adopted instead of relative abundance, which is commonly used, especially in Brazil and Europe, (e.g. Lobo et al., 2004a; 2004b; 2004c; Rimet et al., 2004), following Cox (1998) due to the great heterogeneity of habitats at the different sampling sites. All individuals found on the slide were counted and identified under light microscopy (1,000x) until a representative population was recorded, determined by the method of graph stabilisation (Bicudo, 1990). A minimum of 90% counting effectiveness (Pappas & Stormer, 1996) and 600 valves were always registered.

Taxa identification was based on specialised bibliography and journals. Permanent glass slides are kept at the Limnology Laboratory of the University of Santa Cruz do Sul (slides EL-918 to EL-940).

Summer was the season of choice for this research to be conducted, essentially aiming to generate new information to improve biomonitoring techniques. This was based on Kelly et al. (1998), who stated that the use of diatoms as bioindicators should be put to test when water level is lowest, and expected pollutant concentration is, therefore, highest. Also according to the same authors, algal communities react more rapidly to alteration in water quality when temperature is higher. The same research strategy has recently been used by O´Connell et al. (1997), Ghosh & Gaur (1998) and Soininen et al. (2004).

Statistical analyses

Indicator Species Analysis (Dufrêne & Legendre, 1997) was used to explore species distribution and abundance standards, applied on the species density matrix, and using sampling sites (Fig. 1) as categorical variable.

Subsequently, Canonical Correspondence Analysis (CCA) (Ter Braak, 1986) was applied, to identify the environmental factors most influential to species distribution. Score ordination scale was optimised by species, selecting for the scatter plot those derived from the environmental matrix (linear correlation scores) - an option recommended by Palmer (1993). Data variability was explained by means of intraset correlations (McCune & Mefford, 1999), described by Ter Braak (1986) as immune to the multicolinearity problem. Monte Carlo Test (999 permutations; p£0.05) was used to verify the significance of the results (whether the values were or not attributed by chance). Log of indicator species densities (log |x+1|) and of abiotic variables, except riparian shading comprised, respectively, the biotic and the abiotic data matrices. Both analyses were carried out using PC-Ord for Windows version 4.0 software (McCune and Mefford, 1999). The abiotic variables which showed the highest correlations with the ordination axes were selected to characterise the ecological preferences of the diatom species.

As long as ordination is an analysis of exploratory character, statistical significance of the differences found between environmental variables were investigated by means of randomization test using block design, (Pillar & Orlóci, 1996). Thus, sampling dates were used as blocks, isolating temporal influence.

Correlations between species and variables were also studied through randomization tests. The correlation coefficient between the variables was calculated and tested from the null hypothesis (Ho) that there was no association between pairs in 10,000 iterations (Pillar, 2001). Correlations between environmental variables, using the same procedure, were also tested when necessary for better discussion of results.

Randomization tests were carried out with the Multiv 2.1.1 software (Pillar, 2001).

Results

Water quality – physical and chemical aspects

Table I shows the variation ranges of physical and chemical variables of water at the different sampling units of the experimental design.

During the sampling period, the depth of Schmidt Stream, ranged from 5 cm to 34 cm, and channel width varied from 1.5 m to 6.8 m. Station 3 (S3 - middle course) had the deepest points as well as the greatest channel width, as opposed to S1, (upper course) which had the shallowest water column and narrowest channel. Average stone diameter was similar among all sampling sites. Greatest current velocities were found at stations S5 and S6, - lower course. At S1, water flow was below the flow-meter's detection level (Tab. I).

Water pH was close to neutral in all sites, varying from 6.4 to 7.7. Conductivity and total dissolved solids had the greatest values at sites S1, S2 and S5. Turbidity was highest at S1 and S4. Sampling site S3 showed the lowest average value of suspended solids (2.4 mg L^{-1}), and the values of this variable were highest at S1 (9.6 mg L^{-1}) (Tab. I).

Regarding nutrients, the highest average nitrate concentrations were found at S2 (888 mg L^{-1}) and S4 (845 mg L^{-1}), while the lowest mean values were registered at S5 (590 mg L^{-1}). Phosphorus and silicon were also high at S2 (Tab. I).

Average biochemical oxygen demand was very low at all sites (‹2.0 mg L^{-1}); chemical oxygen demand, on the other hand, was high at S1 (11.0 mg L^{-1}). Nevertheless undetectable levels of this variable were found at sites S3, S4, S5 and S6 (Tab. I). Saturated dissolved oxygen was highest at S3 and lowest at S1, S2 and S4. Due to the use of the most critical value for classification regarding saprobity, levels at all sampling sites were defined according to this latest variable.

Thus, three levels of organical pollution were identified: oligo/b-mesossaprobic (very low pollution) at site S3;b-mesossaprobic (moderate pollution) at sites S5 and S6, and b/a-mesossaprobic (critical pollution) at S1, S2 and S4.

Results of randomisation tests applied to abiotic data are given in Table II.

Table I: Mean, standard deviation (±) and range (minimum and maximum) of environmental variables at the sampling sites. (**BOD**: Biochemical oxygen demand; **COD**: chemical oxygen demand; **COND**: Conductivity; **COORD**: geographical coordinates; **D**: density of valves; **DEPTH**: river depth; **DMT**: diameter of stones; **LIG**: light intensity; **NO_3^-**: nitrate; **SiO_2^{-2}**: silicon; **SS**: suspended solids; **TDS**: total dissolved solids; **TP**: total phosphorus; **TURB**: turbidity; **VEL**: current velocity; **WIDTH**: width of river channel; **%O_2**: saturated dissolved oxygen).

		S1	**S2**	**S3**	**S4**	**S5**	**S6**
COORD. (UTM)		22J 0364875	22J 0364734	22J 0367805	22J0368872	22J 0369402	22J 0371301
		6718002	6716080	6714323	6711905	6715839	6716076
LIG. (lux)		10000	220	› 50000	2400	4200	3000
COND (mS cm^{-1})	Mean	182(±27)	160(±13)	85(±8)	79(±10)	110(±5)	80(±5)
	Min.-Max	153-219	144-173	74-93	65-90	104-116	73-83
% O_2	Mean	60.0(±4.0)	61.4(±10.1)	88.1(±6.9)	63.2(±9.8)	74.7(±8.0)	77.6(±7.8)
	Min.-Max	55.9-65.0	50.1-74.4	80.2-95.3	52.6-75.0	66.2-81.7	70.0-84.5
pH	Mean	7.1(±0.3)	7.1(±0.4)	7.2(±0.6)	7.2(±0.5)	7.2(±0.6)	7.2(±0.5)
	Min.-Max.	6.6-7.4	6.6-7.4	6.5-7.7	6.7-7.7	6.4-7.6	6.6-7.7
WIDTH (m)	Mean	1.8(±0.2)	3.3(±0.9)	6.4(±0.4)	2.6(±0.4)	3.6(±0.5)	3.2(±0.5)
	Min.-Max.	1.5-2.0	1.9-3.9	6.0-6.8	2.2-3.0	3.0-4.1	2.4-3.5
DMT (cm)	Mean	17.3(±5.4)	14.4(±1.6)	14.9(±5.5)	14.6(±5.5)	20.1(±5.4)	13.8(±1.4)
	Min.-Max.	12.6-25.0	12.5-16.1	6.7-18.4	9.9-22.5	15.4-27.0	12.2-15.5
DEPTH (cm)	Mean	6.3(±1.2)	9.8(±1.2)	29.7(±4.5)	11.7(±5.3)	13.8(±2.3)	8.5(±1.3)
	Min.-Max.	5.0-7.7	8.3-11.1	24.7-34.1	7.5-19.4	10.5-15.3	6.6-9.3
VEL (cm s^{-1})	Mean	‹0.1	10.3(±4.0)	6.7(±2.1)	11.2(±2.1)	32.3(±10.1)	50.3(±11.1)
	Min.-Max.	-	6.6-15.4	4.0-8.8	8.6-13.0	23.0-46.4	40.4-65.6
BOD (mg L^{-1})	Mean	1.8(±2.9)	1.3(±1.2)	0.3(±0.4)	0.6(±0.9)	‹0.2	0.6(±1.0)
	Min.-Max.	‹0.2-6.1	‹0.2-2.8	‹0.2-0.9	‹0.2-1.8	-	‹0.2-2.1
COD (mg L^{-1})	Mean	11.0(±1.4)	7.7(±1.1)	5.0(±1.1)	5.1(±0.4)	‹5.0	7.6(±4.5)
	Min.-Max.	9.0-12.1	6.3-8.9	‹5.0-6.0	‹5.0-5.6	-	‹5.0-14.2
TP (mg L^{-1})	Mean	26(±13)	76(±15)	30(±5)	30(±8)	26(±6)	26(±9)
	Min.-Max.	20-45	59-94	22-33	20-39	20-32	20-39
NO_3^- (mg L^{-1})	Mean	685(±59)	888(±85)	685(±128)	845(±210)	590(±328)	748(±126)
	Min.-Max.	600-730	800-1000	500-780	600-1100	100-800	600-900
TDS (mg L^{-1})	Mean	112.4(±16.4)	95.8(±7.3)	55.9(±9.7)	61.9(±7.1)	72.0(±8.9)	53.3(±8.8)
	Min.-Max.	96.0-135.0	89.5-104.5	47.5-69.5	51.5-66.5	59.5-80.0	42.5-63.5
TURB (uT)	Mean	16.1(±8.7)	6.2(±1.1)	5.3(±1.7)	20.6(±10.6)	4.4(±1.3)	11.9(±2.9)
	Min.-Max.	9.7-28.9	5.3-7.7	4.0-7.7	12.7-35.8	3.4-6.1	8.2-15.1
SS (mg L^{-1})	Mean	9.6(±8.8)	9.3(±4.9)	2.4(±1.3)	8.1(±4.1)	3.8(±1.4)	3.9(±1.5)
	Min.-Max.	3.0-22.5	2.0-13.0	1.0-4.0	3.5-12.5	2.0-5.5	2.0-5.5
SiO_2^{-2} (mg L^{-1})	Mean	19.5(±2.1)	30.8(±16.4)	22.3(±1.1)	18.9(±1.2)	23.8(±1.1)	18.9(±0.4)
	Min.-Ma.	16.8-21.5	21.5-55.4	20.7-23.1	17.5-19.9	22.3-24.9	18.3-19.1
D (val/cm^2)	Mean	35,711(±14,622)	10,610(±3,178)	177,290(±46,893)	6,688(±3,761)	31,142(±24,370)	32,314(±31,157)
	Min.-Max.	19,765-54,995	7,176-14,872	126,671-239,693	2,825-10,138	8,856-57,415	2,468-61,537

Table II: Results of randomization hypothesis testing, for comparison between environmental variables, at the different sampling sites along the Schmidt Stream hydrographical basin, RS, Brazil. (**BOD**: Biochemical oxygen demand; **COD**: chemical oxygen demand; **COND**: Conductivity; **D**: density of valves; **DEPTH**: river depth; **DMT**: diameter of stones; **NO_3^-**: nitrate; **SiO_2^{-2}**: silicon; **SS**: suspended solids; **TDS**: total dissolved solids; **TP**: total phosphorus; **TURB**: turbidity; **VEL**: current velocity; **WIDTH**: width of river channel; **$\%O_2$**: saturated dissolved oxygen). p = probability that the sum of squares (Q) obtained by randomization (Q_{RND}) be as extreme as the sum of squares of the observed data (Q_{OBS}) in 10,000 iterations. * = significant results with a£0.05. - = raw data below detection level.

	$p(Q_{RND}?Q_{OBS})$														
Contrasts	**Cond(µS cm^{-1})**	**%O2**	**pH**	**Wi(m)**	**(cm)**	**(cm)**	**($cm\ s^{-1}$)**	**($mg\ L^{-1}$)**	**($mg\ L^{-1}$)**	**($\mu g\ L^{-1}$)**	**($\mu g\ L^{-1}$)**	**($mg\ L^{-1}$)**	**(uT)**	**($mg\ L^{-1}$)**	**($mg\ L^{-1}$)**	**(val/cm^2)**
S1 vs. S2	0.6633	0.9051	0.8447	0.0974	0.4219	0.3180	–	0.9549	0.3137	0.0007*	0.3140	0.4996	0.0731	0.9311	0.0152*	0.2446
S1 vs. S3	0.0014*	0.0002*	0.3135	0.0001*	0.3526	0.0001*	–	0.0905	0.0176*	0.6117	0.9436	0.0003*	0.0289*	0.0081*	0.3997	0.0902
S1 vs. S4	0.0002*	0.7192	0.2871	0.2935	0.3627	0.1804	–	0.2048	0.0359*	0.6237	0.4013	0.0048*	0.5957	0.9664	0.8304	0.0665
S1 vs. S5	0.0637	0.0651	0.2024	0.0335*	0.4356	0.0571	–	0.0057*	0.0001*	0.9579	0.2083	0.0485*	0.0061*	0.1217	0.2768	0.6978
S1 vs. S6	0.0003*	0.0272*	0.3888	0.1073	0.3181	0.5076	–	0.2108	0.1913	0.9362	0.6653	0.0003*	0.6204	0.1401	0.8382	0.4730
S2 vs. S3	0.0143*	0.0007*	0.4247	0.0165*	0.8969	0.004*	0.5286	0.0649	0.2192	0.0024*	0.3330	0.0107*	0.7610	0.0072*	0.1645	0.0021*
S2 vs. S4	0.0029*	0.8050	0.3893	0.5230	0.9004	0.7743	0.8576	0.1700	0.2905	0.0029*	0.7159	0.0479*	0.0122*	0.8970	0.0061*	0.5636
S2 vs. S5	0.1631	0.0863	0.2972	0.7188	0.1040	0.4175	0.0690	0.0041*	0.0041*	0.0002*	0.005*	0.2122	0.4947	0.0957	0.2803	0.4206
S2 vs. S6	0.006*	0.0387*	0.5309	0.9212	0.8515	0.7292	0.0049*	0.1870	0.7646	0.0003*	0.4066	0.0055*	0.2018	0.1132	0.0051*	0.6427
S3 vs. S4	0.7668	0.0009*	0.9295	0.0010*	0.9978	0.0084*	0.4197	0.7164	0.8687	0.9847	0.4089	0.6450	0.0044*	0.0107*	0.2918	0.0001*
S3 vs. S5	0.3387	0.1446	0.7863	0.0354*	0.0722	0.0393*	0.0077*	0.4250	0.1701	0.6220	0.2443	0.2633	0.7232	0.3746	0.6204	0.0328*
S3 vs. S6	0.8346	0.2748	0.8920	0.0097*	0.9477	0.0006*	0.0001*	0.6891	0.3491	0.6417	0.6334	0.8370	0.1086	0.3410	0.2969	0.0101*
S4 vs. S5	0.2100	0.1305	0.8642	0.2857	0.0737	0.5845	0.1015	0.2284	0.1229	0.6337	0.029*	0.5210	0.0008*	0.1293	0.1890	0.1503
S4 vs. S6	0.9416	0.0672	0.8283	0.5729	0.9501	0.5160	0.0094*	0.9729	0.4556	0.6548	0.5815	0.5099	0.3011	0.1387	0.9897	0.2813
S5 vs. S6	0.2415	0.7432	0.6933	0.6392	0.0636	0.2325	0.4678	0.2094	0.0144*	0.9735	0.1340	0.1929	0.0404*	0.9530	0.1934	0.7299

Diatom community composition

Diatom community analysis found a total of 147 species, distributed in 41 genera and 23 families: Achnanthaceae (Achnanthes, Lemnicola, Planothidium), Achnanthidiaceae (Achnanthidium), Naviculaceae (Adlafia, Eolimna, Geissleria, Mayamaea, Navicula, Naviculadicta, Nupela), Amphipleuraceae (Amphipleura, Frustulia), Catenulaceae (Amphora), Aulacoseiraceae (Aulacoseira), Stauroneidaceae (Carpatograma, Craticula, Fistullifera), Cocconeidaceae (Cocconeis), Stephanodiscaceae (Cyclotella), Cymbellaceae (Cymbella, Encyonema, Placoneis), Diadesmidaceae (Diadesmis, Luticola), Diploneidaceae (Diploneis), Eunotiaceae (Eunotia), Sellaphoraceae (Fallacia, Sellaphora), Gomphonemataceae (Gomphonema), Pleurosigmataceae (Gyrosigma), Bacillariaceae (Hantzschia, Nitzschia, Tryblionella) Melosiraceae (Melosira), Neidiaceae (Neidium), Pinnulariaceae (Pinnularia), Surirellaceae (Surirella, Stenopterobia), Fragilariaceae (Ulnaria) and Thalassiosiraceae (Thalassiosira).

Lowest average density was registered at stations S2 and S4 (10,610 and 6,688 valves/cm^2, respectively), while the highest values were found at S3 (average of 177,290 valves/cm^2, with a maximum of 239,693 valves/cm^2), S1 (average 35,711 valves/cm^2), S6 (average 32,314 valves/cm^2), and S5 (average 31,142 valves/cm^2) (Tab. I).

Indicator species analysis singled out 31 taxa with abundances and frequencies significantly associated to one of the tested environments (p£0.05). Six taxa were indicators of S1, eight of S2, thirteen of S3, one of S4, two of S5 and again one of S6 (Tab. III; Fig. 2).

Joint analysis and community environmental preferences

Canonical correspondence analysis explained 51.9% of the data variability on the first three axes. According to Ter Braak and Prentice (1988), such low explicability is to be expected in ordination analyses of ecological data, due to the complexity of factors determining community structure. However, species-environment correlations for axes 1 (r=0.974), 2 (r=0.965) and 3 (r=0.958), indicated strong linear relationships between epilithic algae and environmental variables. Results of Monte Carlo test gave statistical significance (p£0.05) to all three ordination axes (Tab. IV).

On the first axis (explaining 23.4% of the variability) intra-set correlations revealed as the strongest variables: total phosphorus (r=-0.748), conductivity (r=-0.634) and total dissolved solids (r=-0.625), on the negative end of the axis. Additionally, on the positive end of axis one, saturated dissolved oxygen was a strong variable (r=0.623). On the second axis (explaining 15% of the variability) intra-set correlations gave the greatest weights to total dissolved solids (r=0.519) and conductivity (r=0.495), both on the positive end. Finally, on the third axis (explaining 13.5% of the variability) there was a strong positive contribution from turbidity (r=0.750) and chemical oxygen demand (r=0.626) as well as current velocity (r=0.703) and channel width (r=-0.649) on the negative end (Tab. IV; Fig. 3).

Consequently, while figure 3 shows the importance of trophic and ionic enrichment on species distribution and abundance, figure 4 - intersection of axes 3 and 1 - exhibits the contribution of physical factors and variables derived from them, mainly turbidity and current velocity.

Species environmental preferences where determined according to the location of sampling units along the ordination axes (Figs. 3 and 4). S2, for instance, is situated near the negative extreme of axis 1, showing greater affinity of its indicator species for environments with higher levels of phosphorus and ionic concentration, as well as higher tolerance for dissolved solids. In contrast, S3 located closer to the positive extreme, is characterised by communities typical of well oxygenated habitats.

Figure 4 displays the location of S1 and S4 on the positive end of axis 3, characterising the taxa present at these sites as more tolerant to habitats of higher turbidity, and slower current. The ordination of sampling units located closer to the negative end of this axis - towards which the vectors of current velocity, channel width and oxygen saturation extended - was not as sharp, due to the reduced number of indicator species found at the sites where the former variables had the highest scores. Still, dissolved oxygen was an important factor over the populations found at S3, and so was the direct relationship between current velocity and oxygenation of sites S5 and S6, considering the significant correlation found (r=0.4; p=0.04).

Correlations between species and abiotic variables (Tab. V) were used to shed

Table III: Results of Indicator Species Analysis using sampling site as categorical variable and the Monte Carlo test with 10,000 interactions. Only statistically significant (a£0.05) results are shown. Ecological preferences of epilithic diatom species, indicating different zones of the Schmidt Stream Hydrographical Basin, RS, Brazil. **C:** ionic concentration (1- low (74-93 mS cm^{-1}); 2- intermediate (104-116 mS cm^{-1}); 3- high (144-219 mS cm^{-1}). **S**: Saprobity (1- Oligo/b-mesossaprobic; 2- b-mesossaprobic; 3- b/a-mesossaprobic). **T:** Trophy (1- oligotrophic (‹ 45 mg L^{-1} PT); 2- meso/eutrophic (59-94 mg L^{-1} PT); **IV:** indicative value). p=probability that IV obtained by randomization be as extreme as IV of observed data in 10,000 iterations.Ø=requirement not defined for the character.

Sampling Unit	Species	IV	p	C	S	T
S1	Cymbella tumida (Brébison) Van Heurck	98.4	0.0003	Ø	Ø	Ø
S1	Gyrosigma acuminatum (Kützing) Rabenhorst	87.1	0.0011	Ø	Ø	Ø
S1	Sellaphora pupula (Kützing) Mereschkowsky	80.2	0.0003	3	3	Ø
S1	Gomphonema brasiliense Grunow	58.5	0.0391	Ø	Ø	Ø
S1	Neidium affine (Ehrenberg) Pfitzer	57.9	0.0164	Ø	Ø	Ø
S1	Navicula viridula (Kützing) Ehrenberg	56.5	0.0039	Ø	Ø	Ø
S2	Stenopterobia sp.	89.3	0.0004	3	Ø	2
S2	Navicula angusta Grunow	75.0	0.0109	3	3	2
S2	Planothidium rupestoides (Hohn) Round & Bakhtiyarova	75.0	0.0138	Ø	Ø	2
S2	Achnanthes sp. 3	68.7	0.0046	3	Ø	2
S2	Nitzschia acicularis (Kutzing) W. Smith	63.4	0.0063	Ø	Ø	2
S2	Pinnularia sp. Krasske	62.1	0.0059	3	Ø	2
S2	Tryblionella victoriae Grunow	61.4	0.0340	Ø	3	2
S2	Surirella tenera Gregory	53.8	0.0225	Ø	Ø	2
S3	Encyonema perpusillum (Cleve) D.G. Mann	98.4	0.0009	1	1	Ø
S3	Encyonema silesiacum (Bleich in Rabenhorst) D.G. Mann	93.1	0.0002	Ø	Ø	Ø
S3	Adlafia drouetiana (Patrick) Metzeltin & Lange-Bertalot	80.6	0.0009	Ø	Ø	Ø
S3	Luticola goeppertiana (Beisch) D. G. Mann	80.5	0.0007	Ø	1	Ø
S3	Geissleria aikenensis (Patrick) Torgan & Oliveira	69.0	0.0013	Ø	1	Ø
S3	Achnanthidium minutissimum (Kützing) Czarnecki	67.8	0.0020	Ø	Ø	Ø
S3	Nupela sp.	67.7	0.0011	Ø	Ø	Ø
S3	Achnanthes sp. 2	64.7	0.0052	Ø	1	Ø
S3	Nitzschia amphibia Grunow	64.0	0.0008	Ø	Ø	Ø
S3	Eolimna subminuscula (Manguin) Lange-Bertalot	63.4	0.0077	Ø	Ø	Ø
S3	Achnanthes exigua Grunow	58.9	0.0013	Ø	Ø	Ø
S3	Encyonema mesianum (Cholnoky) D. G. Mann	49.5	0.0478	Ø	Ø	Ø
S3	Sellaphora seminulum (Grunow) D.G. Mann	47.7	0.0191	Ø	Ø	Ø
S4	Cocconeis fluviatilis Wallace	74.1	0.0093	Ø	3	Ø
S5	Sellaphora sp.	53.4	0.0155	Ø	Ø	Ø
S5	Ulnaria ulna (Nitzsch) Compère	46.9	0.0318	Ø	Ø	Ø
S6	Navicula symmetrica Patrick	64.1	0.0465	Ø	2	Ø

a light onto which factors or groups of factors were most influential on the distribution and abundance of each taxa, in cases when the environment to which the taxon was linked by the indicator species analysis was determined by more than one single abiotic variable. This is the case, for instance, of sampling site S2 which, as well as its high phosphorus values, showed high ion concentration. Thus, it was possible to conclude that the response of the species Nitzschia acicularis, for example, is associated to total phosphorus rather than conductivity, since significant correlation of this species was only found with the first variable (Tab. V).

Following this interpretation, the ecological classifications were obtained (Tab. III).

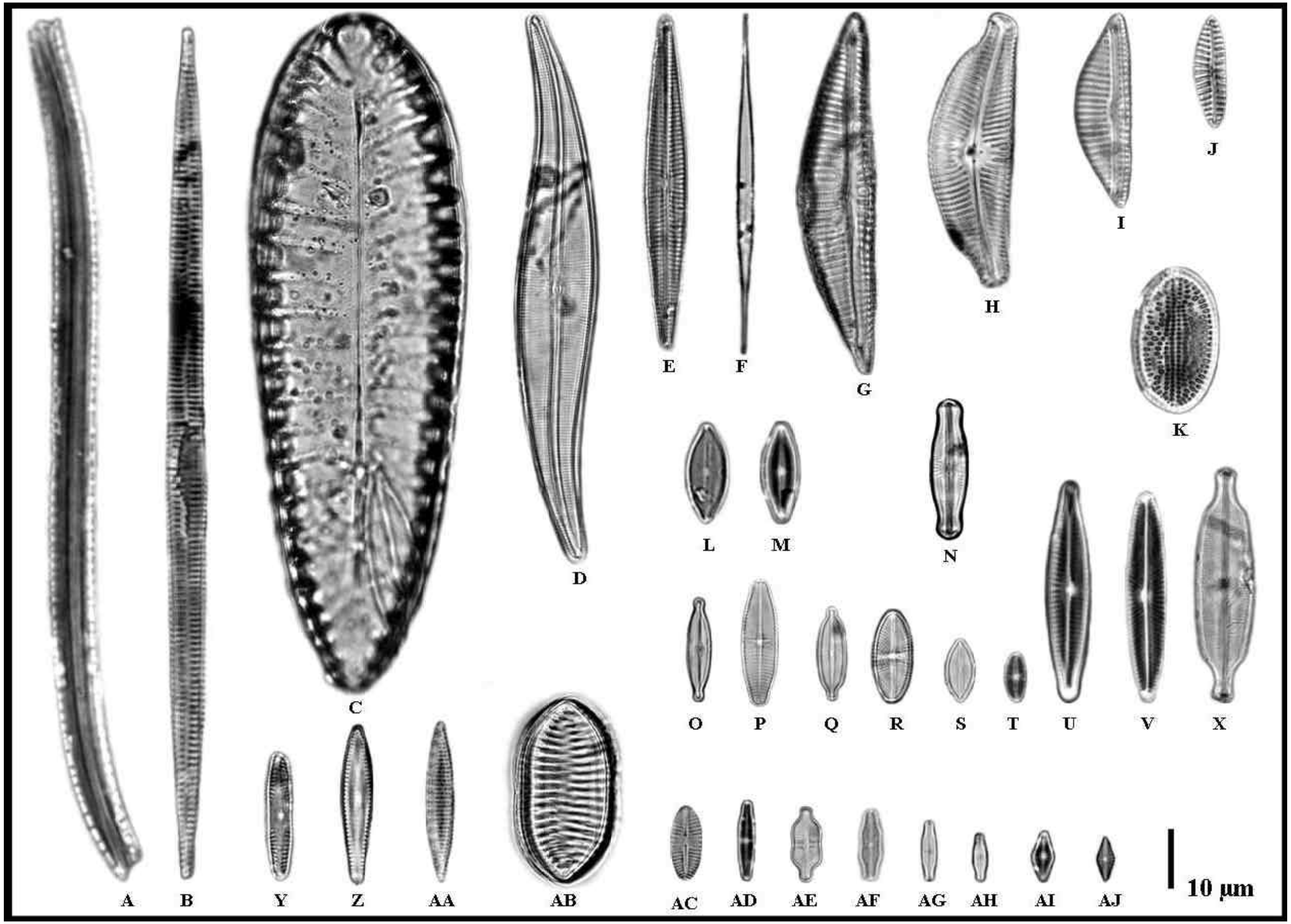

Figure 2: Indicator diatoms species in Schmidt Stream, RS, Brazil. **A:** Stenopterobia sp.; **B:** Ulnaria ulna; **C:** Surirella tenera; **D:** Gyrosigma acuminatum; **E:** Navicula angusta; **F:** Nitzschia acicularis; **G:** Encyonema mesianum; **H:** Cymbella tumida; **I:** Encyonema silesiacum; **J:** Encyonema perpusillum; **K:** Cocconeis fluviatilis, **L,M:** Nupela sp.; **N:** Sellaphora pupula; **O:** Sellaphora sp.; **P:** Geissleria aikenensis; **Q:** Adlafia drouetiana; **R:** Luticola goeppertiana; **S**: Eolimna subminuscula; **T:** Sellaphora seminulum; **U:** Navicula viridula; **V:** Navicula symmetrica; **X:** Neidium affine; **Y:** Pinnularia sp.; **Z:** Gomphonema brasiliense; **AA:** Nitzschia amphibia; **AB:** Tryblionella victoriae; **AC:** Planothidium rupestoides; **AD:** Achnanthidium minutissimum; **AE:** Achnanthes exigua; **AF-AH:** Achnanthes sp. 2; **AI, AJ:** Achnanthes sp. 3.

Table IV: Results of Canonical Correspondence Analysis (CCA) using the indicator species matrix and the physical and chemical variables of the different sampling sites along Schmidt Stream, RS, Brazil.

	Axis 1	**Axis 2**	**Axis 3**
Eigenvalues (m)	0.174	0.112	0.101
% of variance explained	23.4	15.0	13.5
Cumulative % explained	23.4	38.4	51.9
Pearson correlation (Species-environment)	0.974	0.965	0.958
Monte Carlo Test (p)	0.0130	0.0260	0.0010
Eigenvalues Species-environment correlations	0.0200	0.0260	0.0050
Variables		**"intra-set" correlations**	
Conductivity	-0.634	0.495	0.323
Saturated dissolved oxygen	0.623	0.205	-0.544
pH	0.126	0.241	-0.155
Turbidity	0.006	-0.289	0.750
Width of river channel	0.311	-0.209	-0.649
Diameter of stones	0.056	0.176	-0.045
River depth	0.433	-0.219	-0.540
Current velocity	0.132	-0.349	-0.703
Biochemical oxygen demand	-0.384	-0.130	0.315
Chemical oxygen demand	-0.420	0.185	0.626
Total phosphorus	-0.748	-0.129	-0.381
Nitrate	-0.373	-0.270	0.072
Total dissolved solids	-0.625	0.519	0.430
Suspended solids	-0.431	-0.265	0.406
Silicon	-0.482	-0.169	-0.347

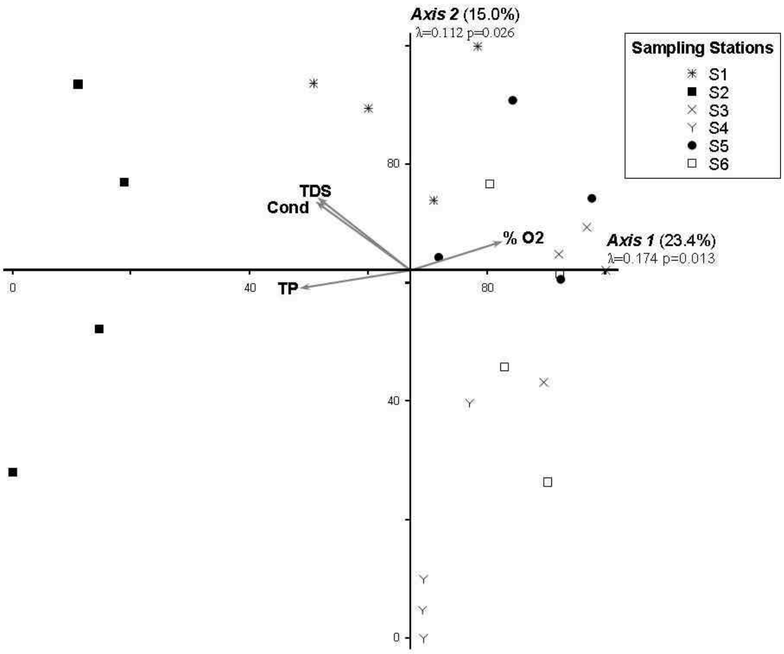

Figure 3: Scatter plot based on Canonical Correspondence Analysis (CCA). Ordination of the sampling units along axes 1 and 2 (**COND:** conductivity; **TP:** total phosphorus; **TDS:** total dissolved solids; **$\%O_2$:** percentage of O_2 saturation).

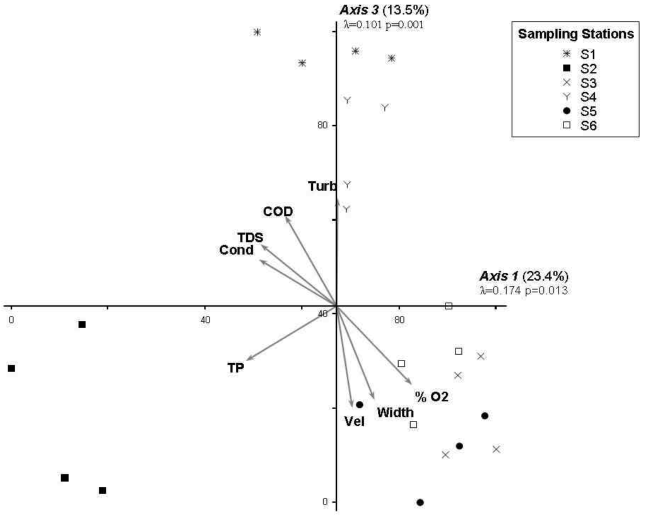

Figure 4: Scatter plot based on Canonical Correspondence Analysis (CCA). Ordination of the sampling units along axes 3 and 1. (**COND:** conductivity; **COD:** chemical oxygen demand; **WIDTH:** width of stream at the sampling site; **TP:** total phosphorus; **TDS:** total dissolved solids; **TURB:** turbidity; **VEL:** current velocity; **%O$_2$:** percentage of O$_2$ saturation).

Table V: Correlations of indicator species with environmental variables singled out by CCA (**COD**: chemical oxygen dernand; **COND**: Conductivity; **TDS**: total dissolved solids; **TP**: total phosphorus; **TURB**: turbidity; **VEL**: current velocity; **WIDTH**: width of river channel; **%O2**: saturated dissolved oxygen). * = significant results with a£;0.05. ** = significant results with a£0.01.

	Cond (μS cm^{-1})	**% O2**	**Turb (uT)**	**Width (m)**	**Vel (cm s^{-1})**	**COD (mg L^{-1})**	**TP (μg L^{-1})**	**TDS (mg L^{-1})**
Achnanthes exigua	-0.16	0.13	-0.29	0.43*	-0.05	-0.29	0.13	-0.09
Achnanthes sp. 2	-0.25	0.54**	-0.43*	0.57**	0.10	-0.41*	0.00	-0.14
Achnanthes sp. 3	0.56**	-0.23	-0.34	-0.16	-0.05	0.09	0.60**	0.55**
Achnanthidium minutissimum	0.08	-0.05	0.03	0.39	-0.49*	-0.07	0.00	0.10
Adlafia drouetiana	0.01	0.28	-0.24	0.45*	-0.37	-0.25	-0.21	-0.03
Cocconeis fluviatilis	0.11	-0.66**	0.53**	-0.33	-0.24	0.18	0.22	0.22
Cymbella tumida	0.67**	-0.45*	0.37	-0.53**	-0.82**	0.62**	-0.04	0.65**
Encyonema mesianum	-0.05	0.20	0.25	0.09	-0.35	0.33	-0.16	0.07
E. perpusillum	-0.43*	0.54**	-0.34	0.68**	0.12	-0.43*	-0.03	-0.57**
E. silesiacum	-0.14	0.16	-0.11	0.40	-0.26	-0.01	-0.14	-0.31
Eolimna subminuscula	0.47*	0.17	-0.60**	0.35	-0.45*	-0.06	0.10	0.36
Geissleria aikenensis	-0.34	0.67**	-0.47*	0.56**	0.07	-0.50*	-0.10	-0.18
Gomphonema brasiliense	0.22	0.26	-0.05	-0.01	-0.40	0.08	-0.25	0.28
Gyrosigma acuminatum	0.70**	-0.44*	0.12	-0.48*	-0.59**	0.52**	0.05	0.75**
Luticola goeppertiana	-0.13	0.63**	-0.42*	0.48*	-0.07	-0.25	-0.18	-0.12
Navicula angusta	0.44*	-0.47*	-0.21	-0.01	-0.03	0.24	0.67**	0.32
N. symmetrica	-0.30	0.62**	-0.36	0.31	0.36	-0.27	-0.27	-0.20
N.viridula	0.26	-0.14	0.17	-0.47*	-0.30	0.35	-0.28	0.40
Neidium affine	0.32	-0.28	0.33	-0.43*	-0.57**	0.45*	-0.08	0.44*
Nitzschia acicularis	0.19	-0.08	-0.15	0.20	0.13	0.07	0.64**	0.22
N. amphibia	0.39	0.14	-0.50*	0.43*	-0.36	0.10	0.05	0.31
Nupela sp.	0.06	0.23	-0.32	0.40	-0.12	-0.24	0.33	0.08
Pinnularia sp.	0.56**	-0.30	-0.13	-0.22	-0.10	0.34	0.52**	0.53**
Planothidium rupestoides	0.37	-0.13	-0.20	-0.01	0.00	0.18	0.75**	0.35
Sellaphora pupula	0.52*	-0.58**	0.41*	-0.52**	-0.41	0.27	-0.09	0.59**
S. seminulum	0.01	0.39	-0.38	0.37	-0.28	-0.22	-0.17	0.05
Sellaphora sp.	-0.10	0.35	-0.49**	0.21	0.41*	-0.63*	-0.07	-0.05
Stenopterobia sp.	0.48*	-0.31	-0.36	0.03	0.05	0.07	0.72**	0.34
Surirella tenera	0.13	-0.40	0.07	0.03	0.06	0.11	0.43*	0.28
Tryblionella victoriae	0.23	-0.58**	0.15	-0.10	-0.02	0.17	0.61**	0.22
Ulnaria ulna	0.12	0.18	-0.27	-0.08	0.14	-0.27	-0.08	0.22

Discussion

Physical and chemical variables

On the hypothetical profile of any lotic system, the decrease of variables such as oxygen saturation and current velocity, as well as the increase of depth, width, oxygen consumption, temperature, nutrients and suspended solids are usually expected (Vannote et al., 1980; Schäfer, 1984; Silva et al., 1998).

However, the situation found in Schmidt Stream contradicts theoretical hypotheses - more specifically, regarding oxygen saturation, current velocity and suspended solids. While oxygen saturation was higher on third order channels, current velocity was highest at sites near the confluence with Taquari River, where depth and channel width, on some sampling dates, were even lower than in second order tributaries (Tab. I). Such uncharacteristic features of the lower portion of the stream may be a consequence of land use and occupation around its course, where irrigated rice cultures and a series of small reservoirs built by farmers have altered the natural hydrodynamic of the channel.

Schmidt stream is, actually, a hydrographical basin of incomplete zonation, where the theory presented by Vanote et al. (1980) has limited application. Moreover, the stream ends on the middle course of Taquari-Mirim River, which, in turn, turns out its waters into the middle reach of the Taquari River.

According to Vannote et al. (1980), headwater streams allow the greatest interaction with the surrounding landscape along a lotic system; hence, they are predominantly collectors, processors and carriers of material originated in the land system. Consequently, the significant increase in total dissolved solids at sites S1 and S2 (Tab. I and II) illustrates the important role played by superficial drainage on electrolyte input to the water. This process is more prominent at sites located closer to the main source of Schmidt Stream, as the primary land use activity is growth of grazing fields for dairy farming.

Whitton (1975) separates the increase of phosphorus concentration in water by natural causes (e.g. weathering of rocks) from that caused by man, since the later is usually linked to nitrogen increase. At site S2, where the highest phosphorus levels were found, one of the highest average nitrogen levels of the whole sampling period was concomitantly registered. Between sites S1 and S2, the stream receives domestic effluents from a small village, as well as effluent from farm houses. It is, therefore, possible to infer that the increase of phosphate between these sites is due to human occupation.

Several tributaries originating in regions of preserved forests reach the main course before the third sampling site, contributing to the dilution of solids carried from upstream (Fig. 1, Tab. I).

Diatom community

Hill (1996) postulated that algal biomass in streams is often related to the level of canopy development of the riparian forest. In some cases, biomass determined by chlorophyll-a on open habitats may reach four to six times the values found where tree shading by riparian vegetation is greater. Besides, Lamb & Lowe (1987) demonstrated that diatom cell density may be up to three times higher at low current velocity sites (15 cm s^{-1}), than at sites of more rapid waters (40 cm s^{-1}). This proclivity, however, was not confirmed by the present study, since no significant differences of diatom cell density were found between S1 and S6, sites of contrasting flow regimes (Tab. II).

Biomass at sampling site S3, calculated from average valve density, was 26 times greater than biomass at S4, and 16 times greater than values found at S2 (Tab. I and II), the latter two being highly shaded environments (Tab. I). Thus, fluctuations of density appear to be primarily reflecting the community response to light energy availability.

Hill (1996) also pointed out the indirect effects of light on community architecture which, ultimately, influences community composition. With increments in vertical structure caused by enhancement of primary production, there may be selection pressure, favouring species which are able to extend above or move among the periphytic matrix. Such circumstances were verified at sampling sites S1 and S3, which had the greatest light incidence among the selected sampling stations (Tab. I), and where the most expressive indicative values were given to species of the genera Cymbella and Encyonema (Tab. III).

According to Admiral & Peletier (1979), facultative heterotrophic species, such as representatives of the genera Nitzschia and Navicula, make use of this feature under low light regimes; this may explain the predominance of Nitzschia acicularis and Tryblionella victoriae (Sin. Nitzschia levidensis var. victoriae) at station S2, the most shaded site (Tab. I).

On the other hand, the decrease of species significantly associated to stations S5 and S6, shown by the indicator species analysis (Tab. III), reflects the negative influence of high current velocity over the community. At such environments, a bi-dimensional structure prevails, determined by prostrate or apically attached species, the ones that, according to Stevenson (1996), are best adapted to sites where intense flow predominates. In contrast, taxa as Cymbella tumida (Sin. Cocconema tumidum Brébisson in Kützing; Cymbella stomatophora Grunow) and Encyonema perpusillum (Sin. Cymbella perpusilla Cleve), with weak fixation structures, are easily dislodged and are, therefore, associated with sites of lower current velocities.

Hence, the genera Encyonema and Cymbella met, at stations S1 and S3, favourable conditions of low current velocity, aiding their development (Tab. I). Therefore, due to their anatomic structure, which guarantees greater competitive ability on the use of available light energy (foot and mucilage tube), they surpassed the development of species composing lower strata, and established themselves as dominant colonizers.

Margaléf (1983) pointed out that suspended materials accumulating in river zones of low velocity are colonized by algae of slow sliding motion, such as Nitzschia, Navicula, Pinnularia, Surirella, Stenopterobia and Gyrosigma. Supporting this postulate, the genera listed above were found in high numbers at sampling stations S1 and S2, environments where the lowest flow and highest concentrations of suspended solids were registered (Tab. I and III).

Factors determining the predominance of Cocconeis fluviatilis at S4 may also be emphasised. Even though Cocconeis is a genus usually referred to as characteristic of high flow habitats (e.g. Opshal et al., 2003), in Schmidt Stream, flow did not seem to be a major variable determining the high indicator value of Cocconeis fluviatilis. This comes on account of the similarity of flow values among S2, S3 and S4, and the absence of significant differences between these sampling sites (Tab. II), as well as the lack of correlation of this species with current velocity (Tab. V).

Studying the effects of shading by a bridge on the benthic algal community of the River Olczyski in Polland, Kawecka (1985, 1986), found positive development numbers for the genus Cocconeis (C. placentula var. euglypta, in this case) under reduced light conditions, confirming previous research data that showed extensive populations of this species only in mountain streams with intact riparian forest (Kawecka, 1980). Furthermore, Robinson & Rushforth (1987) found good adaptability of Cocconeis to shaded environments which were also affected by substrate disturbance.

Considering the above, along with the data shown in Table I - which define S4 as a habitat of high riparian shading and turbidity - the light gradient may be singled out as one of the major variables determining distribution and abundance of this species in the Schmidt Stream hydrographical basin. Other elements such as grazing by macroinvertebrates may also influence these patterns. This postulation is based on previous studies of periphytic communities. For instance, Moore (1975) working on an English river, found that Cocconeis placentula was barely affected by grazing, compared to other diatoms, due to its adnate habit and strong adherence to substrata. According to Steinman (1992), prostrate forms such as Cocconeis are well adapted to hard grazing pressure while bigger forms, with vertical growth, are more susceptible to its effects. Further studies regarding the influence of these variables over the periphytic diatom community are needed in order to clarify this question.

Environmental preferences of indicator species, emphasizing biomonitoring

Progressive decrease of oxygen saturation and increase of biochemical as well as chemical oxygen demand are usual indicators of organic pollution in water systems (Branco & Necchi Jr., 1997; Silva et al., 1998). Conductivity, on the other hand, is said to be an efficient indicator of nutrient enrichment, since the ions determining its intensity are not involved in biological processes, and are less susceptible to relative oscillations than nutrients themselves (Biggs, 1995; Soininen et al., 2004). Sudden changes of phosphate concentrations, along with increase of nitrogen ions indicate artificial eutrophication processes (Whitton, 1975; Esteves, 1998). Finally, increase in turbidity and total dissolved solids indicate, in water quality evaluation studies, environmental degeneration (COMITESINOS, 1990, 1993; DMAE, 2003).

Based on these statements and crossing species responses with the variables mentioned (Tab. V), two water quality signalling groups for the Schmidt Stream were outlined: Achnanthes sp. 3, Cocconeis fluviatilis, Navicula angusta, Nitzschia acicularis, Tryblionella victoriae, Pinnularia sp., Planothidium rupestoides, Sellaphora pupula (Sin. Navicula pupula Kützing), Stenopterobia sp. and Surirella tenera, related to zones of human impact and Achnanthes sp. 2, Encyonema perpusilum, Geissleria aikenensis (Sin. Geissleria schimidiae Lange-Bertalot, Navicula aikenensis Patrick), Luticola goeppertiana (Sin. Navicula goeppertiana (Bleisch) H. L. Smith), Navicula symmetrica and Nitzschia amphibia, characteristic of environments with low mineral and organic content as well as high oxygen levels.

Surirella tenera has already been referred to as a species characteristic of contextually meso-eutrophic waters by Van Dam et al. (1994). Similarly, Nitzschia acicularis characterised these environments, according to Van Dam et al. (1994), Lelan & Porter (2000) and Winter & Duthie (2000).

Wetzel et al. (2002), working in the upper, middle, and lower courses of the River Pardo hydrographical basin, RS, Brazil, registered mean total phosphorous values of 13, 54 and 134 mg L^{-1} TP respectively, in three months of sampling. After the application of the Indicator Species Analysis (Dufrêne & Legendre, 1997), Sellaphora pupula (Sin. Navicula pupula Kützing) was associated with the lower courses. In addition, Lobo et al. (2004a; 2004b) regarded S. pupula as characteristic of sampling sites where total phosphorus averaged 82 mg L^{-1}. The preference of S. pupula for more eutrophic environments was also recorded by Lobo et al (2004c), through the study of the urban streams Condor and Capivara, in the City of Porto Alegre, RS, Brazil. Total phosphorus average in the former equalled 119 mg L^{-1}, whereas the later averaged 454 mg L^{-1} for the same variable. S. pupula was therefore associated only with Capivara Stream.

In the Schmidt stream, nevertheless, S. pupula did not show such affinity for eutrophic waters, being more abundant and frequent at S1, a less eutrophic site, where conductivity is high (tab. I and III). Surveys carried out in the Guaíba hydrographical basin (Wetzel et al., 2002; Lobo et al., 2004c) supported the referred preference of Sellaphora pupula for sites of higher electrolyte concentration, since the most eutrophic sites in those studies were also the ones with the highest conductivity values.

It is therefore, possible to infer that the response of this population is determined not by phosphorus availability in the environment, but instead by the set of non-specific ions which increase conductivity levels. This hypothesis is sustained by the lack of correlation of this species with total phosphorus as well as by its positive association with conductivity (Tab. V).

Van Dam et al. (1994), based upon Lange-Bertalot (1978; 1979b), established five water quality classes, joining up oligossaprobic with oligo/b-mesossaprobic and b/a-mesossaprobic with a-mesossaprobic. Data from the Schmidt stream have confirmed Encyonema perpusilum as an oligo/b-mesossaprobic species.

The environmental response of Achnanthidium minutissimum (Sin. Achnanthes minutissima), Cymbella tumida, Eolimna subminuscula (Sin. Navicula subminuscula Manguin), Gyrosigma acuminatum, Neidium affine and Sellaphora sp. to chemical stressors was masked by their correlation with current velocity (Tab. V). Regarding Encyonema mesianum, Encyonema silesiacum, Gomphonema brasiliense, Nupela sp., Sellaphora seminulum (Sin. Navicula seminulum Grunow) and Ulnaria ulna (Sin. Synedra ulna (Nitzsch) Ehrenberg; Fragilaria ulna (Nitzsch) Lange-Bertalot var. ulna), no correlations with the main variables given by the CCA ordination were found (Tab. V). These species have, therefore, been considered of low descriptive potential within this particular hydrographical basin.

It must be pointed out that the classification given hereby is a tentative one, evolving in concert with other bioindication techniques, as the understanding of the interactions between water quality and community integrity increases (Hill et al., 2000). Thus, the permanent revision of species environmental requirements is a fundamental aid to validation and calibration of specific tolerances (Lobo et al., 2004c). Moreover, Van Dam et al. (1994), quoting Lange-Bertalot (1978, 1979b), pointed out that the appearance of species highly tolerant to organic pollution cannot indicate, by itself, highly saprobic environments, since taxa distribution is not limited by decreases in saprobity, but by the increasing degradation of the system. On the other hand, the presence of taxa not tolerant to high pollution levels may be considered evidence of its absence. May be cited as examples of this situation, Sellaphora seminulum, widely referred to as tolerant to organic pollution and eutrophication (Lange-Bertalot, 1979a; Kobayasi & Mayama, 1982; Kobayasi & Mayama, 1989; Van Dam et al., 1994; Lobo et al., 2002; Lobo et al., 2004a; 2004b; 2004c; Rimet et al., 2004; Salomoni, 2004), and Luticola goeppertiana, described as resistant to organic pollution by Lange-Bertalot (1979a), Van Dam et al. (1994), Lobo et al. (2002) and Salomoni (2004). In the Schmidt Stream, however, these species were found in oligotrophic and oligosaprobic waters, mainly at sampling site S3.

Acknowledgements

The authors are grateful to Carlos Eduardo Wetzel and Pablo Diego Gressler, for help with field work; to Dr André Jasper (Biology Department at UNIVATES) and to Dr Paulo Alves de Souza (Geosciences Institute, UFRGS), for help with photographic material; to Dr Lezilda Torgan (Natural Sciences Museum of the Zoo-Botanical Foundation RS) for help with photographic material and taxonomical review; to Daniela Bes for help with field work and taxa identification; and, finally to CAPES for an MSc grant to G Hermany.

References

Admiral, W. & Peletier, H. 1979. Influence of organic compounds and light limitation on the growth rate of estuarine benthic diatoms. Br. Phycol. J., 14:197-206.

American Public Health Association. 1999. Standard methods for the examination of water and wastewater. 19th ed. Washington. 1 CD-ROM.

Bicudo, D.C. 1990. Considerações sobre metodologias de contagem de algas do perifiton. Acta Limnol. Bras., 3:459-475.

Bicudo, D.C., Necchi Jr., O. & Chamixaes, C.B.C.B. 1995. Periphyton studies in Brazil: present status and perspectives. In: Tundisi, J.G., Bicudo, C.E.M. & Tundisi, T.M. (eds.) Limnology in Brazil. Graftex, Rio de Janeiro. p.37-58.

Biggs, B.J. 1995. The contribution of flood disturbance, catchment geology and land use to the habitat template of periphyton in stream ecosystems. Freshwater Biol., 33:419-438.

Branco, L.H.Z. & Necchi Jr., O. 1997. Variação longitudinal de parâmetros físicos e químicos em três rios pertencentes a diferentes bacias de drenagem na região noroeste do estado de São Paulo. Acta Limnol. Bras., 9:165-177.

COMITESINOS. 1990. Utilização de um índice de qualidade da água para o Rio dos Sinos/RS. COMITESINOS, Porto Alegre. 33p.

COMITESINOS. 1993. Aplicação de um índice de qualidade da água para o Rio dos Sinos. DMAE, Porto Alegre. 59p.

Cox, E.J. 1998. A rationale and some suggestions for developing rapid biomonitoring techniques using identification of live diatoms. In: Proceedings of XV Diatom Symposium. Perth, Austrália, p.43-50.

DMAE. 2003. Avaliação da qualidade da água do Lago Guaíba. Subsídios para a gestão da Bacia Hidrográfica. DMAE, Porto Alegre. 34p. (Ecos pesquisa, 7).

Dufrêne, M. & Legendre, P. 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Monogr., 67:345-366.

Esteves, F.A. 1998. Fundamentos de Limnologia. Interciência, Rio de Janeiro. 602p.

Ghosh, M. & Gaur, J.P. 1998. Current velocity and the stablishment of stream algal periphyton communities. Aquat. Bot., 60:1-10.

Gómez, N. & Licursi, M. 2001. The Pampean Diatom Index (IDP) for assessment of rivers and streams in Argentina. Aquat. Ecol., 35:173-181.

Hall, R.I. & Smol, J.P. 1992. A weighted-averaging regression and calibration model for inferring total phosphorus concentration from diatoms in British Columbia (Canada) lakes. Freshwater Biol., 27:417-434.

Hall, R.I. & Smol, J.P. 1999. Diatoms as indicators of lake eutrophication. In: Stormer, E.F. & Smol, J.P. (eds.) The diatoms: applications for the environmental and earth sciences. Cambridge University Press, Cambridge. p.128-168.

Hamm, A. 1969. Die Ermittlung der gewässergüteklassen bei fliebgewässen nach dem gewässergütesystem und gewässergütenomogramm. Müncher Beiträge Abwasser Fisch. Flussbiol., 15:46-48.

Hill, W. 1996. Effects of light. In: Stevenson, J., Bothwell, M.L. & Lowe, R.L. (eds.) Algal ecology: freshwater benthic ecosystems. Academic Press, San Diego. p.121-148.

Hill, B.H., Herlihy, A.T., Kaufmann, P.R., Stevenson, R.J., McCoemick, F.H. & Johnson, C.B. 2000. Use of periphyton assemblage data as an index of biotic integrity. J. North Am. Benthol. Soc., 19:50-67.

Katoh, K. 1991. Variation in species composition of diatom assemblages in an Unpolluted River. Jpn. J. Limnol., 52:145-151.

Kawecka, B. 1980. Sessile algae in European mountain streams. 1. The ecological characteristics of communities. Acta Hydrobiol., 22:361-420.

Kawecka, B. 1985. Ecological characteristics of sessile algal communities in the Olczyski stream (Tatra Mts, Poland) with special consideration of light and temperature. Acta Hydrobiol., 27:299-310.

Kawecka, B. 1986. The effect of light deficiency on communities of sessile algae in the Olczyski stream (Tatra Mts, Poland). Acta Hydrobiol., 28:379-386.

Kelly, M.G. 1998. Use of the trophic diatom index to monitor eutrophication in rivers. Water Res., 32:236-242.

Kelly, M.G. & Whitton, B.A. 1995. The trophic diatom index: a new index for monitoring eutrophication in rivers. J. Appl. Phycol., 7:433-444.

Kelly, M.G., Cazaubon, A., Coring, E., Dell´Uomo, A., Ector, L., Goldsmith, B., Guash, H., Hürlimann, J., Jarlman, A., Kawecka, B., Kwandrans, J., Laugaste, R., Lindstrom, E.A., Leitao, M., Marvan, P., Padisák, J., Pipp, E., Prygiel, J., Rott, E., Sabater, S., Van Dam, H. & Vizinet, J. 1998. Recommendations for the routine sampling of diatoms for water quality assessments in Europe. J. Appl. Phycol., 10:215-224.

Kobayasi, H. & Mayama, S. 1982. Most pollution tolerant diatoms of severely polluted rivers in the vicinity of Tokyo. Jpn. J. Phycol., 30:188-196.

Kobayasi, H. & Mayama, S. 1989. Evaluation of river water quality by diatoms. Korean J. Phycol., 4:121-133.

Lamb, M.A. & Lowe, R. 1987. Effects of current velocity on the physical structuring of diatom (Bacillariophyceae) communities. Ohio J. Sci., 87:72-78.

Lange-Bertalot, H. 1978. Diatomeen-Differentialarten anstelle von Leitformen: ein geeigneteres Kriterium der Gewässerbelastung. Arch. Hydrobiol. Suppl., 51: 393-427.

Lange-Bertalot, H. 1979a. Pollution tolerance of diatoms as a criterion for water quality estimation. Nova Hedwigia, 64: 285-304.

Lange-Bertalot, H. 1979b. Toleranzgrenzen und populationsdynamik benthischer Diatomeen bei unterschiedlich starker starker Abwasser-belastung. Arch. Hydrobiol. Suppl., 56:184-219.

Leland, H.V. & Porter, S.D. 2000. Distribution of benthic algae in the upper Illinois River basin in relation to geology and land use. Freshwater Biol., 44:279-301.

Lobo, E.A. 1995. Comparative study of the approaches for evaluation of river water quality using epilithic diatom assemblages. Tokyo, Tokyo University of Fisheries, 187p (PhD Thesis).

Lobo, E.A. & Callegaro, V.L. 2000. Avaliação da qualidade de águas doces continentais com base em algas diatomáceas epilíticas: enfoque metodológico. In: Tucci, C.E.M. & Marques, D.M. (eds.) Avaliação e controle da drenagem urbana. Ed. da UFRGS, Porto Alegre. p.277- 300.

Lobo, E.A., Katoh, K. & Aruga, Y. 1995. Response of epilithic diatom assemblages to water pollution in rivers in the Tokyo Metropolitan area. Freshwater Biol., 34:191-204.

Lobo, E.A., Callegaro, V.L.M., Oliveira, M.A., Salomoni, S.E., Schuler, S. & Asai, K. 1996. Pollution tolerant diatoms from lotic systems in the Jacui Basin, Rio Grande do Sul, Brasil. Iheringia Ser. Bot., 47:45-72.

Lobo, E.A., Callegaro, V.L & Bender, P. 2002. Utilização de algas diatomáceas epilíticas como indicadoras da qualidade da água em rios e arroios da Região Hidrográfica do Guaíba, RS, Brasil. EDUNISC, Santa Cruz do Sul. 127p.

Lobo, E., Callegaro, V.L.M., Hermany, G., Bes, D., Wetzel, C.E. & Oliveira, M.A. 2004a. Use of epilithic diatoms as bioindicators, with special emphasis to the eutrophication problem of lotic systems in Southern Brazil. Acta Limnol. Bras., 16(1):25-40.

Lobo, E.A., Callegaro, V.L.M., Hermany, G., Gómez, N. & Ector, L. 2004b. Review of the use of microalgae in South America for monitoring rivers, with special referente to diatoms. Vie & Milieu, 54:105-114.

Lobo, E.A., Callegaro, V.L., Wetzel, C.E., Hermany, G. & Bes, D. 2004c. Water quality evaluation of Condor and Capivara Streams, Municipal District of Porto Alegre, RS, Brazil, using epilithic diatom communities as bioindicators. Oceanol. Hydrobiol. Stud., 33:77-93.

Margaléf, R. 1983. Limnología. Omega, Barcelona. 1120p.

McCune, B. & Mefford, M.J. 1999. PC-ORD. Multivariate analysis of ecological data, version 4. MjM Software Design, Gleneden Beach. 237p.

Minshall, G.W., Petersen, R.C., Cummins, K.W., Bott, T.L., Sedell, J.R., Cushing, C.E. & Vannote, R.L. 1983. Interbiome comparison of stream ecosystem dynamics. Ecol. Monogr., 53:1-25.

Miranda, A.L.B. 2003. Estrutura da comunidade e tipos funcionais de algas epilíticas na Bacia Hidrográfica do Rio Maquiné, RS, Brasil. Porto Alegre, Universidade Federal do Rio Grande do Sul, 100p (PhD Thesis).

Moore, J.W. 1975. The role of algae in the diet of Asellus aquaticus L. and Gammarus pullex L. J. Anim. Ecol., 44:719-730.

Moreira Filho, H., Valente-Moreira, I.M., Souza-Mosimann, R.M. & Cunha, J.A. 1990. Avaliação florística e ecológica das diatomáceas (Chrysophyta, Bacillariophyceae) marinhas e estuarinas nos Estados do Paraná, Santa Catarina e Rio Grande do Sul. Estud. Biol., 25:5-48.

Moreno, J.A. 1961. O clima do Rio Grande do Sul. Secretaria da Agricultura, Porto Alegre. 42p.

Moro, R.S. & Fürstenberger, C.B. 1997. Catálogo dos principais parâmetros ecológicos de diatomáceas não marinhas. Editora da UEPG, Ponta Grossa. 282p.

Mourthé-Junior, C.A. 2000. Modificações estruturais na comunidade de diatomáceas em um gradiente de poluição hídrica. Trecho superior da bacia do rio das Velhas (região metropolitana de Belo Horizonte – MG). Belo Horizonte, Universidade Federal de Minas Gerais, 79p (Master Thesis).

Naiman, R.J. & Sedell, J.M. 1979. Benthic organic matter as a function of stream order in Oregon. Arch. Hydrobiol., 87:404-422.

O´Connell, J.M., Reavie, E.D. & Smol, J.P. 1997. Assessment of water quality using epiphytic diatom assemblages on Cladophora from the St. Lawrence River (Canada). Diatom Res., 12:55-70.

Oliveira, M.A. 1996. Ecologia do perifíton de substrato artificial em cursos d´água do trecho médio do Arroio Sampaio, Mato Leitão, RS, Brasil. Porto Alegre, Universidade Federal do Rio Grande do Sul, 128p (Master Thesis).

Opsahl, R.W., Wellnitz, T. & Poff, N.L. 2003. Current velocity and invertebrate grazing regulate stream algae: results of an in situ electrical exclusion. Hydrobiologia, 499:135-145.

Palmer, M.W. 1993. Putting things in even better order: the advantages of canonical correspondence analysis. Ecology, 74:2215-2230.

Pappas, J.L. & Stormer, E.F. 1996. Quantitative method for determining a representative algal sample count. J. Phycol., 32:693-696.

Patrick, R. 1993. Factors to consider in interpreting diatom change. Nova Hedwigia, 106:361-377.

Pillar, V.D. 2001. MULTIV: multivariate exploratory analysis, randomization testing and bootstrap resampling. User´s guide v.2, 42p. Disponível em: http://ecoqua.ecologia.ufrgs.br.

Pillar, V.D. & Orloci, L. 1996. On randomization testing in vegetation science: multifactor comparisons of relevé groups. J. Veg. Sci., 7:585-592.

Prygiel, J. 1991. Use of benthic diatoms in surveillance of the Artois-Picardie Basin hydrobiological quality. In: Whitton, B.A., Rott, E. & Friedrich, G. (eds.) Use of algae for monitoring rivers. Institut für Botanik, Universität Innsbruck, Austria. p.89-96.

Reavie, E.D., Smol, J.P., Carignan, R. & Lorrain, S. 1998. Diatom paleolimnology of two fluvial lakes in the St. Lawrence River: a reconstruction of environmental changes during the last century. J. Phycol., 34:446-456.

Rimet, F., Ector, L., Cauchie, H.M. & Hoffmann, L. 2004. Regional distribution of diatom assemblages in the headwater streams of Luxembourg. Hydrobiologia, 520:105-117.

Robinson, C.T. & Rushforth, S.R. 1987. Effects of physical disturbance and canopy cover on attached diatom community structure in an Idaho stream. Hydrobiologia, 154:49-59.

Round, F.E. 1993. A review and methods for the use of epilithic diatoms for detecting and monitoring changes in river water quality. HMSO, London. 63p.

Rumeau, A. & Coste, M. 1988. Initition à la systématique des diatomées d´eau douce pour l´utilisation pratique d´un índice diatomique générique. Bull. Fr. Peche Piscic., 309:12-69.

Salomoni, S. 2004. Diatomáceas epilíticas indicadoras da qualidade de água na bacia do Rio Gravataí, Rio Grande do Sul, Brasil. São Carlos, Universidade Federal de São Carlos, 230p (PhD Thesis).

Schäfer, A. 1984. Fundamentos de ecologia e biogeografia das águas continentais. Ed. da UFRGS, Porto Alegre. 532p.

Shirata, M.T. 1985. Catálogo de diatomáceas (Chrysophyta, Baccilariophyceae) de água doce do Estado do Paraná, Brasil. Estud. Biol., 13:1-64.

Silva, A.M.M., Henry, R., Carvalho, L.R. & Santini, J.A.J. 1998. A capacidade de autopurificação de um curso de água: um estudo de caso no Rio Pardo (Botucatu, SP). Acta Limnol. Bras., 10:83-99.

Silva-Benavides, A.M. 1996. The use of water chemistry and benthic diatom communities for qualification of a polluted tropical river in Costa Rica. Rev. Biol. Trop., 44:395-416.

Slàdecek, V. 1973. System of water quality from the biological point of view. Arch. Hydrobiol. Ergeb. Limnol., 7:1-218.

Soininen, J., Paavola, R. & Muotka, T. 2004. Benthic diatom communities in boreal streams: community structure in relation to environmental and spatial gradients. Ecography, 27:330-342.

Souza, M.G.M. 2002. Variação da comunidade de diatomáceas epilíticas ao longo de um rio impactado no município de São Carlos-SP e sua relação com variáveis físicas e químicas. São Carlos, Universidade Federal de São Carlos, 168p (PhD Thesis).

Steinman, A.D. 1992. Does an increase in irradiance influence periphyton in a heavily-grazed woodland stream? Oecologia, 91:163-170.

Stevenson, J. 1996. An introduction to algal ecology in freshwater benthic habitats. In: Stevenson, J., Bothwell, M.L. & Lowe, R.L. (eds.) Algal ecology: freshwater benthic ecosystems. Academic Press, San Diego. p.121-148.

Stevenson, R.J. & Pan, Y. 1999. Assessing environmental conditions in rivers and streams with diatoms. In: Stormer, E.F. & Smol, J.P. (eds.) The diatoms: applications for the environmental and earth sciences. Cambridge University Press, Cambridge. 469p.

Stevenson, R.J. & Smol, J.P. 2002. Use of algae in environmental assessments. In: Wehr, J.D. & Sheath, R.G. (eds.) Freshwater algae of North America: ecology and classification. Academic Press, San Diego. p.775-804.

Ter Braak, C.J.F. 1986. Canonical correspondence analysis: a new eigenvector technique for multivariate direct gradient analysis. Ecology, 67:1167-1179.

Ter Braak, C.J.F. & Prentice, I.C. 1988. The theory of gradient analysis. Adv. Ecol. Res., 18:271-313.

Torgan, L.C. & Biancamano, M.I. 1991. Catálogo das diatomáceas (Bacilariophyceae) referidas para o Estado do Rio Grande do Sul, Brasil, no período de 1973 a 1990. Cad. Pesqui. Ser. Bot., 3:1-201.

Van Dam, H., Mertens A. & Skindelam, J. 1994. A coded checklist and ecological indicator values of freshwaters diatoms from Netherlands. Neth. J. Aquat. Ecol., 28:117-133.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R. & Cushing, C.E. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci., 37:130-137.

Wetzel, C.E., Lobo, E.A., Oliveira, M.A., Bes, D. & Hermany, G. 2002. Diatomáceas epilíticas relacionadas a fatores ambientais em diferentes trechos dos Rios Pardo e Pardinho, Bacia Hidrográfica do Rio Pardo, RS, Brasil: resultados preliminares. Cad. Pesqui. Ser. Biol., 14:17-38.

Whittaker, R.H. 1967. Gradient analysis of vegetation. Biol. Rev., 42:207-264.

Whitton, B.A. 1975. River ecology. Blackwell Scientific Publications, London. 615p.

Winter, J.G. & Duthie, H.C. 2000. Epilithic diatoms as indicators of stream total N and total P concentration. J. North Am. Benthol. Soc., 19:32-49.

Received: 08 August 2005
Accepted: 11 April 2006