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Effects of dredging on benthic diatom assemblages in a lowland stream

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ABSTRACT

The objective of this study was to assess the effects of dredging on the structure and composition of diatom assemblages from a lowland stream and to investigate whether the response of diatom assemblages to the dredging is also influenced by different water quality. Three sampling sites were established in Rodríguez Stream (Argentina); physico-chemical variables and benthic diatom assemblages were sampled weekly in spring 2001. Species composition, cell density, diversity and evenness were estimated. Diatom tolerance to organic pollution and eutrophication were also analyzed. Differences in physico-chemical variables and changes in benthic diatom assemblages were compared between the pre- and post-dredging periods using a *t*-test. Data were analyzed using Principal Components Analysis (PCA), non-metric multidimensional scaling (MDS) ordination and cluster analysis. The effects of dredging in the stream involve two types of disturbances: (i) in the stream bed, by the removal and destabilization of the substrate and (ii) in the water column, by generating chemical changes and an alteration of the light environment of the stream. Suspended solids, soluble reactive phosphorus and dissolved inorganic nitrogen were significantly higher in post-dredging periods. Physical and chemical modifications in the habitat of benthic diatoms produced changes in the assemblage; diversity and species numbers showed an immediate increase after dredging, decreasing at the end of the study period. Changes in the tolerance of the diatom assemblage to organic pollution and eutrophication were also observed as a consequence of dredging; in the post-dredging period sensitive species were replaced by either tolerant or most tolerant species. These changes were particularly noticeable in site 1 (characterized by its lower amount of nutrients and organic matter previous to dredging), which showed an increase in the amount of nutrients and oxygen demand as a consequence of sediment removal. However, these changes were not so conspicuous in sites 2 and 3, which already presented a marked water quality deterioration before the execution of the dredging works.

1. Introduction

Structural interventions in the natural hydrological cycle through canalization or damming of rivers, diversion of water within or among drainage basins, and the over-pumping of aquifers are usually undertaken with a beneficial objective in mind. Experience has shown, however, that the resulting long-term environmental degradation often outweighs these benefits (Meybeck and Helmer, 1996).

The most frequent effects of dredging on aquatic ecosystems comprise changes in the concentration of suspended solids, turbidity, light penetration, and the increase of nutrients and toxic substances in the water column as a consequence of sediment removal (Armengol, 1998; Lewis et al., 2001; Newell et al., 1998). Moreover, dredging may result in losses of river habitat, increase in flow velocity, losses of bank-side vegetation, erosion of bed and bank material, decrease of soil stability, increase of sediment loads to the river, and changes in water depth and in-stream temperature (Brooker, 1985). These changes have different kinds of impacts on the aquatic life inhabiting lakes, reservoirs or rivers (Armengol, 1998).

The dredging of rivers and streams of the Pampean plain (which have intrinsically low slopes) is frequently carried out with the aim of increasing their discharge capacity towards the Río de la Plata or the Atlantic Ocean. Frequently, the inappropriate use of the land, inadequate urban planning, and contamination accelerate the siltation process that leads to the need of dredging and mud extraction activities.

In recent years many researches have been carried out focusing on the effects of dredging and dredged material disposal on biological resources. While most of these studies concentrated on marine and estuarial systems (Boyd et al., 2005; Díaz, 1994; Erftemeijer and Lewis, 2006; Kenny and Rees, 1994, 1996; Lewis et al., 2001; Nayar et al., 2004; Newell et al., 1998, 2004; Piersma et al., 2001; Robinson et al., 2005; Sánchez-Moyano et al., 2004; Smith et al., 2006; Szymelfenig et al., 2006; Witt et al., 2004), there is

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a paucity of information pertaining to the impact of dredging in freshwater ecosystems such as rivers and reservoirs (Armengol, 1998; Prat et al., 1999).Of those studies most have concentrated on macrobenthos community (Boyd et al., 2005; Cooper et al., 2007; Díaz, 1994; Greig and Pereira, 1993; Kenny and Rees, 1996; Lewis et al., 2001; Newell et al., 1998, 2004; Piersma et al., 2001; Prat et al., 1999; Robinson et al., 2005; Sánchez-Moyano et al., 2004; Smith et al., 2006; Szymelfenig et al., 2006; Winger and Lasier, 1995; Wirth et al., 1996; Witt et al., 2004). Implications of the impacts of dredging on the biota have been also reported for birds (Howarth et al., 1982), fishes (Rice and White, 1987) and aquatic plants (Brookes, 1987; Combs et al., 1983; Erftemeijer and Lewis, 2006; Lee et al., 1982). At present few works have been published referring to dredging impact on phytoplankton and periphytic algae (Lewis et al., 2001; Nayar et al., 2004; Prat et al., 1999).

Among the most severe consequences of dredging on biological organisms, some studies on fish have demonstrated reductions in total density and biomass, increases of fry mortality, and changes in population structure (Brooker, 1985). Several studies have shown significant reductions in the density of benthic invertebrates, losses of species typical of slower-flowing or vegetated reaches, loss of diversity and fall in species number (Brooker, 1985; Cooper et al., 2007; Díaz, 1994; Kenny and Rees, 1996; Lewis et al., 2001; Newell et al., 1998, 2004; Robinson et al., 2005; Smith et al., 2006; Szymelfenig et al., 2006; Witt et al., 2004). Studies on algal-periphyton have reported increases in total density, decreases in diversity indices, and changes in community composition in the post-dredging period (Lewis et al., 2001).

Apart from these studies, there is very little information on the effects of dredging on benthic diatoms. According to Stevenson and Bahls (1999) diatoms are useful ecological indicators because they are abundant in most lotic ecosystems and their species are differentially adapted to a wide range of ecological conditions. The great number of species provides multiple, sensitive indicators of environmental change and the specific conditions of their habitat. Lowe and Laliberte (1996) pointed out that diatoms are probably the most widespread and abundant group among all the divisions of benthic algae, and that epipelic microhabitats are usually dominated by highly motile diatoms capable of moving over and between fine particles of sediment. Diatom importance in river and stream ecosystems is based on their fundamental role in food webs, oxygenation of surface waters, and linkage in biogeochemical cycles. As one of the most species-rich components of river and stream communities, diatoms are important elements of biodiversity and genetic resources in rivers and streams (Stevenson and Pan, 1999). Among biological indicators diatoms have one of the shortest generation times (Rott, 1991). They reproduce and respond rapidly to environmental change and provide early warning indicators of both pollution increases and habitat restoration success (Stevenson and Pan, 1999). According to Patrick and Reimer (1966) some species may divide several times a day (e.g. 2–8 divisions per day).

The objective of this study was to assess the effects of dredging on the structure and composition of diatom assemblages from a lowland stream (Rodríguez Stream) and to investigate whether the response of diatom assemblages to the dredging is also influenced by different water quality.

2. Study area and dredging works

This study covered a 12-km stretch of Rodríguez Stream (total length = 22 km), a lotic system (stream order = 2) located in Buenos Aires Province, Argentina. The bottom substrate is mostly composed of slime-clay, except in the lower part where gravel is the dominant substrate. Three sampling sites were established (Fig. 1). Site 1 was placed in a horticultural zone, site 2 was located

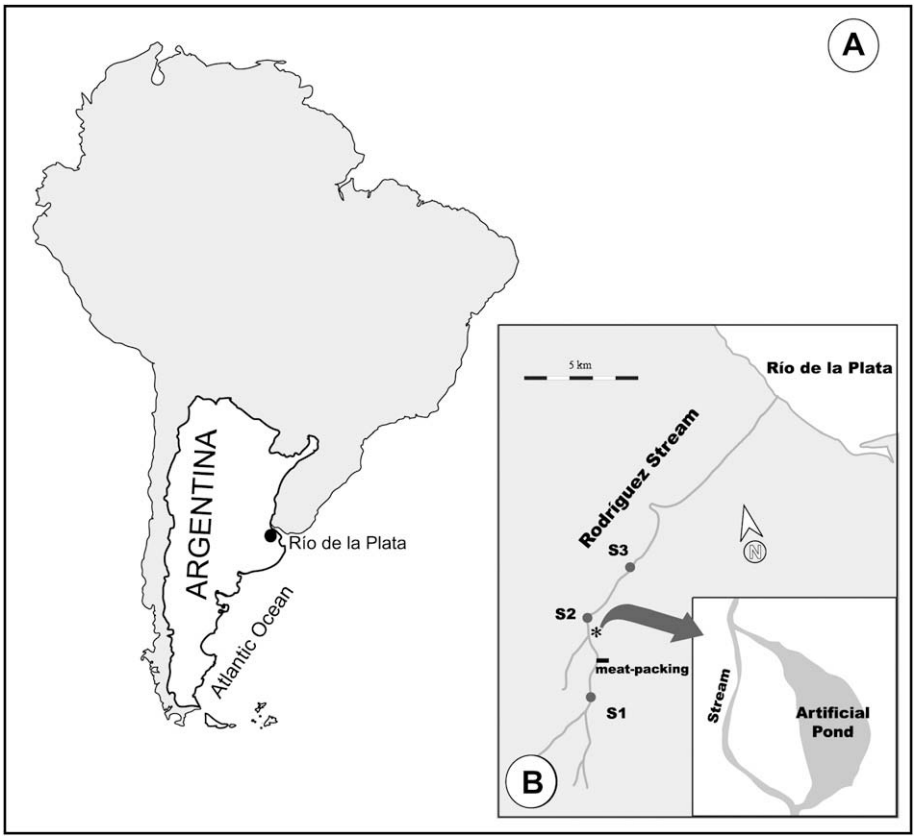

Fig. 1. (A) Study area and (B) location of the sampling sites and the artificial pond in the Rodríguez Stream, Buenos Aires Province, Argentina.

downstream of an artificial pond and receives the effluents from a meat-packing company, whereas site 3 was located in an urban zone receiving domestic effluents.

Dredging works were performed by using an excavator that removed bottom sediments and disposed them at the stream margins. The dredging works started in site 1 on 13 November 2001. Two weeks later they continued in site 2, and in spite of proceeding downstream in the subsequent weeks they did not reach site 3 during the study period. The latter site received, however, an important quantity of suspended solids during the last 3 weeks of the study.

3. Material and methods

3.1. Field and laboratory work

The sampling period lasted 8 weeks. Three weeks corresponded to the pre-dredging period and the last five ones to post-dredging one (which was considered to start from the beginning of the works on site 1). Conductivity, pH, flow, and temperature were evaluated weekly *in situ*. Water samples were collected in order to analyze: NO_3^--N, NO_2^--N, NH_4^+-N, PO_4^{3-}-P, suspended solids, BOD_5 and COD (APHA, 1998).

In each sampling site 10 sub-samples were collected by pipetting of the surface layer of the sediment (5–10 mm) and fixed in 4% formaldehyde to analyze the benthic diatoms (Stevenson, 1984; Lowe and Laliberte, 1996). Cell density was estimated using a Sedgwick–Rafter chamber and expressed per square centimeter. For taxonomic analysis, the sample was oxidized with hydrogen peroxide, and the reagents extracted by successive washing steps by centrifugation. The samples were subsequently mounted in Naphrax®. Three hundred valves from each sample were counted using an Olympus BX 50 microscope with interference phase contrast.

The species were classified according to their tolerance to organic pollution and eutrophication following Lange-Bertalot (1979), Gómez and Licursi (2001), and Licursi and Gómez (2003). Species number, Shannon and Wiener diversity index (H', calculated using ln), and evenness (Ludwig and Reynolds, 1988) were calculated.

3.2. Statistical analysis

Differences in physico-chemical and structural variables of the diatom assemblage were compared between the pre- and post-dredging periods using a *t*-test. To determine statistically significant differences in diatom species tolerance between pre- and post-dredging periods an unpaired *t*-test was employed. When the data were not normally distributed and/or had unequal variances they were analyzed by Mann–Whitney Rank Sum test.

To explore the temporal evolution of the assemblage during the study period a Principal Components Analysis (PCA) was performed on the structural variables of the diatom assemblage that showed significant differences. This analysis was performed using CANOCO v 4.53.

Aiming to identify patterns in the assemblage structure, species composition (pre- and post-dredging) was compared by using non-metric multidimensional scaling (MDS) ordination and cluster analysis, using the Bray–Curtis similarity measure; species abundance data were $\log_e(x+1)$ transformed (McCune and Grace, 2002; Warwick and Clarke, 1993). A combination of MDS and cluster analysis results in a mutual consistency of representations (Cheng, 2004).

4. Results

4.1. Physical and chemical characteristics of the pre- and post-dredging periods

Physico-chemical data measured during the study period are summarized in Table 1. Conductivity was higher during the post-dredging period. BOD_5 and COD diminished during the

Table 1
Physical and chemical characteristics of the sampling sites at the Rodríguez Stream during the pre- and post-dredging periods

Site	Date	Temperature (°C)	Conductivity ($\mu S\,cm^{-1}$)	BOD_5 ($mg\,l^{-1}$)	COD ($mg\,l^{-1}$)	N-NH_4^+ ($mg\,l^{-1}$)	N-NO_2^- ($mg\,l^{-1}$)	N-NO_3^- ($mg\,l^{-1}$)
Pre-dredging period								
S1	24/10	24.0	395	7	53	0.18	0.12	2.22
S2		24.0	883	50	59	0.13	0.36	1.64
S3		24.8	907	54	63	0.15	0.57	4.78
S1	31/10	17.6	160	5	50	0.27	0.07	0.01
S2		22.4	349	51	57	0.22	0.10	0.12
S3		23.1	402	30	54	0.30	0.25	2.55
S1	7/11	18.5	226	5	42	0.12	0.10	0.75
S2		22.1	807	95	163	0.10	0.60	0.23
S3		23.2	728	81	138	0.10	0.28	3.13
Post-dredging period								
Dredging in site 1								
S1	13/11	20.0	187	1	49	0.22	0.01	0.11
S2		20.0	362	47	59	0.24	0.14	0.91
S3		20.1	559	26	48	0.24	0.30	2.78
S1	20/11	25.0	626	4	29	0.07	0.17	2.19
S2		26.2	1190	18	38	12.80	0.71	0.50
S3		26.1	907	13	30	10.45	1.10	2.51
Dredging in site 2								
S1	27/11	21.4	302	2	23	0.20	0.08	1.60
S2		27.2	381	40	60	8.12	0.42	0.94
S3		25.2	603	27	56	3.60	0.50	2.31
S1	4/12	20.1	750	25	66	0.29	0.27	3.20
S2		24.4	1205	21	50	11.87	0.70	0.62
S3		23.3	957	34	48	8.70	1.31	3.07
S1	11/12	22.8	668	27	45	0.36	0.24	3.40
S2		28.6	1436	33	49	0.45	0.37	2.10
S3		27.3	1060	38	78	2.69	0.62	2.70

post-dredging period, particularly in sites 2 and 3, while in site 1 these variables showed a tendency to increase towards the end of the study. On the other hand, the nutrients represented by soluble reactive phosphorus (SRP) and dissolved inorganic nitrogen (DIN: NO_3^--N; NO_2^--N; NH_4^+-N) showed a marked increase in sites 2 and 3 immediately after dredging started in site 1, whereas in this site an increase in nutrient amount occurred 3 weeks later (Fig. 2). The dredging carried out in site 1 produced an increase in suspended solids (Fig. 2). Such increase was immediate in site 1 and, as works proceeded downstream, it was also detected in sites 2 and 3. Since an artificial pond (located between sites 1 and 2) favored sediment retention, the increase in suspended solids downstream of site 1 only occurred after the execution of the works started in site 2 (Fig. 2). The *t*-test performed on the physico-chemical variables showed that suspended solids, SRP and DIN were significantly different ($p < 0.1$) between pre- and post-dredging periods.

Some stream hydraulic conditions changed as a consequence of the dredging (Table 2). In site 1, width, flow, and discharge increased while in site 2 width and depth increased and flow and discharge decreased in the post-dredging period.

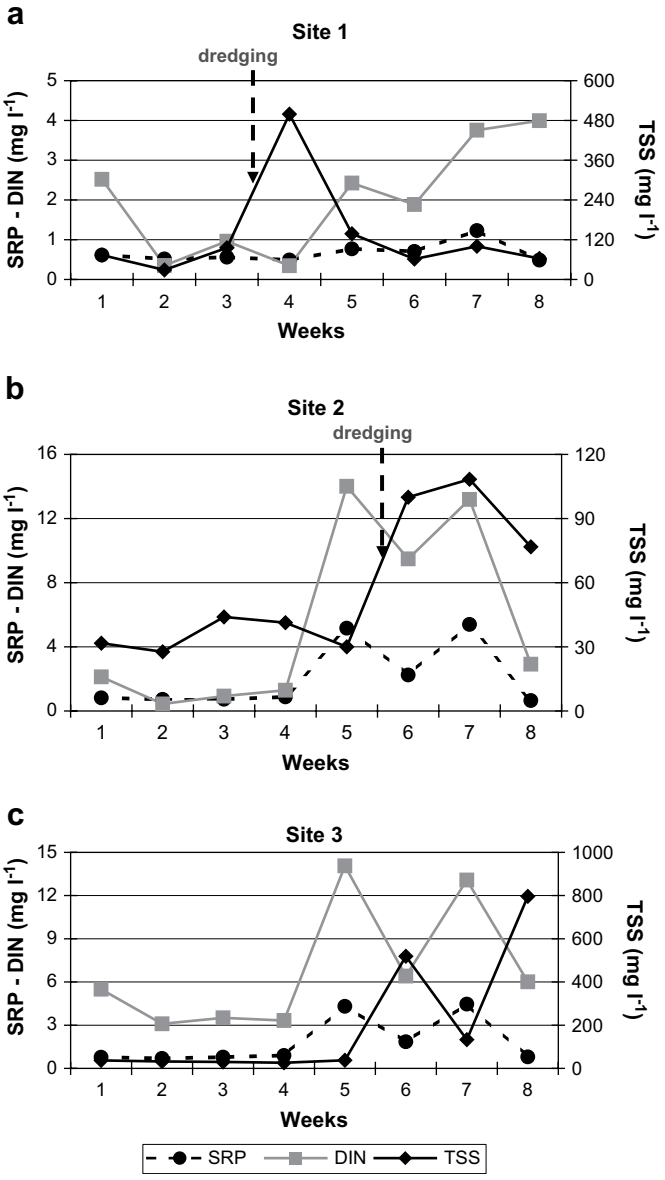

Fig. 2. Total suspended solids (TSS), soluble reactive phosphorus (SRP), and dissolved inorganic nitrogen (DIN: nitrates + nitrites + ammonium) throughout the study period in sampling sites 1 (a), 2 (b), and 3 (c). The arrows indicate the start of the dredging works in sites 1 and 2 (November 2001).

Table 2
Stream hydraulic conditions of the Rodríguez Stream during the pre- and post-dredging periods

		Width (m)	Depth (m)	Flow ($m\,s^{-1}$)	Discharge ($m^3\,s^{-1}$)
S1	Pre-dredging	2.62 (±0.13)	0.25 (±0.09)	0.28 (±0.27)	0.22 (±0.26)
	Post-dredging	4.30 (±0.25)	0.20 (±0.06)	0.57 (±0.52)	0.40 (±0.10)
S2	Pre-dredging	2.95 (±0.37)	0.18 (±0.03)	0.68 (±0.34)	0.38 (±0.25)
	Post-dredging	5.25 (±0.35)	0.28 (±0.13)	0.32 (±0.46)	0.23 (±0.19)
S3	Pre-dredging	6.83 (±0.29)	0.16 (±0.01)	0.88 (±0.33)	0.94 (±0.39)

Since site 3 was not dredged during the study period, the data shown correspond to the pre-dredging period.

4.2. Benthic diatom assemblages

4.2.1. Species composition, diversity, and density

A total of 66 species were identified (Table 3). Diversity and species numbers showed an immediate increase after dredging in sites 1 and 2, decreasing at the end of the study period. In site 3, where no dredging was carried out, the diversity and species numbers diminished after dredging in site 1 and showed an increasing trend towards the end of the study period (Fig. 3).

Density decreased immediately after the dredging, this reduction was also recorded in site 3 as a consequence of the dredging in site 2. Towards the end of the study this parameter showed values that were similar to or higher than those recorded at the beginning, except in site 3 where dredging progressing downstream continued affecting the diatom assemblages (Fig. 3).

The *t*-test between pre- and post-dredging revealed the existence of significant differences ($p < 0.001$) in diversity, species numbers, and density. Therefore, these relevant biological descriptors were used to carry out a PCA aimed at exploring the temporal evolution of the assemblage. Axis 1 explained 67.8% of the total variation, while axis 2 explained 29.9% of it. Table 4 shows the correlations between the aforementioned variables and each axis.

In sites 1 and 2, where the dredge extracted sediments, the PCA distances between sampling points were greater during the post-dredging period (Fig. 4a and b). On the other hand, in site 3 (Fig. 4c), where the bottom remained unaltered but changes occurred in the water column, the inter-sample distances were smaller between weeks 4 and 8 (post-dredging period) than those observed in sites 1 and 2. Although the effects of dredging affected the diatom assemblages of all sampling sites, the structural changes were more pronounced in the sites where both the stream bed and the water column were disturbed (sites 1 and 2). Changes were less evident in site 3, where only the water column was affected.

The MDS ordination for benthic diatom assemblages from sampling sites of the Rodríguez Stream is presented in Fig. 5. In site 1, the pre-dredging samples form a tight cluster, indicating a high similarity between samples, while post-dredging samples are much more diffusely distributed (higher derived variance, S^2). This separation of the post-dredging samples indicates that they are biologically dissimilar; the stress value obtained was 0.02. For two-dimensional ordinations, stress <0.05 gives an excellent representation with no prospect for misinterpretation (Clarke and Warwick, 1994). This ordination was also observed in site 2; in this site the fourth week samples grouped with the pre-dredging samples indicating a delay in the response of the diatom assemblage to upstream dredging. This is probably due to the delay in the increase of suspended solids as a consequence of the artificial pond located between sites 1 and 2. Finally in site 3, where no dredging was carried out, there was no clear separation between pre- and post-dredging samples suggesting that changes in diatom

Table 3
List of diatom species identified in the samples taken from the Rodríguez Stream

Species	Acronym
Achnanthidium minutissimum (Kütz.) Czarnecki	ADMI
Amphora libyca Ehrenberg	ALIB
Amphora montana Krasske	AMMO
Amphora veneta Kützing	AVEN
Anomoeoneis sphaerophora (Ehrenberg) Pfitzer	ASPH
Caloneis bacillum (Grunow) Cleve	CBAC
Cocconeis placentula Ehrenberg	CPLA
Craticula accomoda (Hustedt) Mann	CRAC
Craticula halophila (Grunow ex Van Heurck) Mann	CHAL
Denticula elegans Kützing	DELE
Diadesmis confervacea Kützing	DCOF
Diadesmis contenta (Grunow ex V. Heurck) Mann	DCOT
Diploneis pseudovalis Hustedt	DPSO
Encyonema silesiacum (Bleisch in Rabh.) Mann	ESLE
Eolimna subminuscula (Manguin) Moser Lange-Bertalot & Metzeltin	ESBM
Fallacia monoculata (Hustedt) Mann	FMOC
Fallacia pygmaea (Kützing) Stickle & Mann ssp. *pygmaea* Lange-Bertalot	FPYG
Frustulia vulgaris (Thwaites) De Toni	FVUL
Gomphonema clavatum Ehrenberg	GCLA
Gomphonema minutum (Ag.) Agardh f. *minutum*	GMIN
Gomphonema parvulum Kützing	GPAR
Gyrosigma acuminatum (Kützing) Rabenhorst	GYAC
Gyrosigma nodiferum (Grunow) Reimer	GNOD
Hantzschia amphioxys (Ehrenberg) Grunow	HAMP
Hippodonta capitata (Ehrenberg) Lange-Bertalot, Metzeltin & Witkowski	HCAP
Lemnicola hungarica (Grunow) Round & Basson	LHUN
Luticola mutica (Kützing) Mann	LMUT
Melosira varians Agardh	MVAR
Navicula angusta Grunow	NAAN
Navicula cincta (Ehrenberg) Ralfs	NCIN
Navicula cryptocephala Kützing	NCRY
Navicula difficillima Hustedt	NDIF
Navicula erifuga Lange-Bertalot	NERI
Navicula kotschyi Grunow	NKOT
Navicula pseudobryophila Hustedt	NPBY
Navicula schroeteri Meister	NSHR
Navicula tenelloides Hustedt	NTEN
Navicula trivialis Lange-Bertalot	NTRV
Navicula veneta Kützing	NVEN
Nitzschia amphibia Grunow	NAMP
Nitzschia brevissima Grunow	NBRE
Nitzschia communis Rabenhorst	NCOM
Nitzschia debilis (Arnott) Grunow	NDEB
Nitzschia frustulum (Kützing) Grunow	NIFR
Nitzschia linearis (Agardh) W. Smith	NLIN
Nitzschia microcephala Grunow	NMIC
Nitzschia palea (Kützing) W. Smith	NPAL
Nitzschia sigma (Kützing) W. Smith	NSIG
Nitzschia umbonata (Ehrenberg) Lange-Bertalot	NUMB
Pinnularia borealis Ehrenberg var. *rectangularis* Carlson	PBOR
Pinnularia brauniana (Grunow) Mills	PBRN
Pinnularia gibba Ehrenberg	PGIB
Pinnularia microstauron (Ehrenberg) Cleve	PMIC
Pinnularia subcapitata Gregory	PSCA
Planothidium lanceolatum (Brebisson ex Kützing) Lange-Bertalot	PTLA
Rhoicosphenia abbreviata (C. Agardh) Lange-Bertalot	RABB
Sellaphora pupula Kützing	SPUP
Sellaphora seminulum (Grunow) Mann	SSEM
Surirella angusta Kützing	SANG
Surirella brebissonii Krammer & Lange-Bertalot	SBRE
Surirella ovalis Brébisson	SOVI
Surirella tenera Gregory	SUTE
Tryblionella apiculata Gregory	TAPI
Tryblionella calida (Grunow in Cl. & Grun.) Mann	TCAL
Tryblionella hungarica (Grunow) D.G. Mann	THUN
Ulnaria ulna (Nitzsch.) Compère	UULN

assemblage are not as strong when the bottom remains unaltered and changes occur only in the water column. The results from the hierarchical clustering are reasonably consistent with those obtained from MDS.

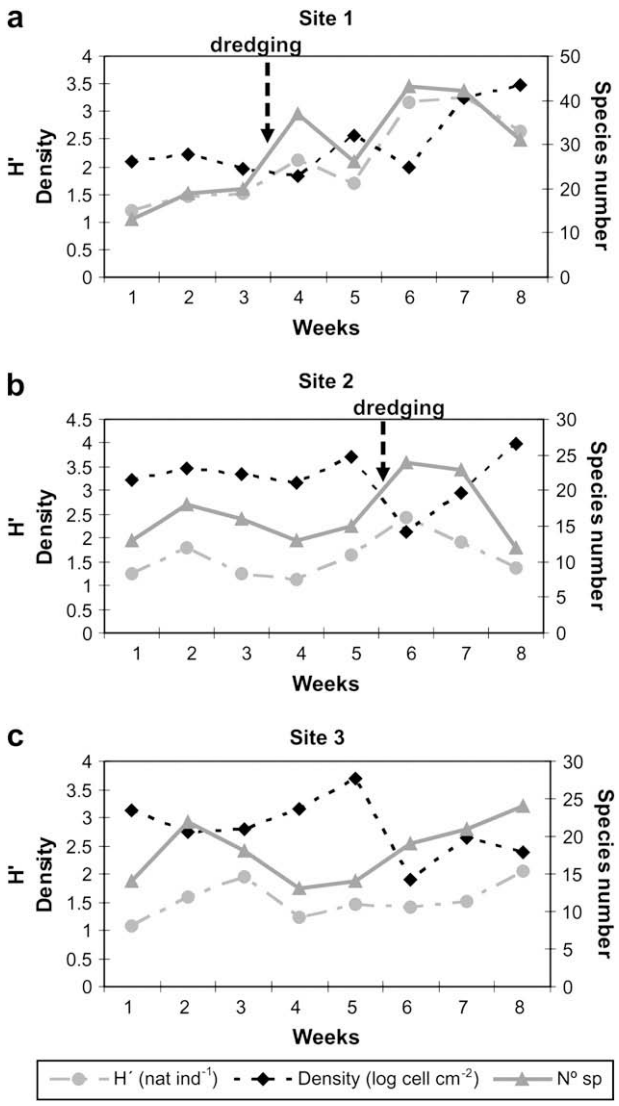

Fig. 3. Shannon and Wiener diversity index (calculated using ln), density (expressed as logarithm), and number of species recorded throughout the study period in sampling sites 1 (a), 2 (b), and 3 (c). The arrows indicate the start of the dredging works in sites 1 and 2 (November 2001).

Dredging produced changes in species composition of diatom assemblages. During the pre-dredging period, site 1 was characterized by the dominance of *Rhoicosphenia abbreviata*, immediately after dredging *Nitzschia amphibia* became dominant. At the end of the study, species' composition was completely different being *Navicula erifuga* and *Nitzschia palea* the dominant species, meanwhile various species were represented with lower relative abundances, such as *Gomphonema clavatum*, *N. amphibia*, *R. abbreviata*, *Navicula kotschyi*, *Hippodonta capitata*, *Nitzschia linearis*, and *Melosira varians*, among others (Fig. 6).

Diatom composition in site 2 was not affected by the dredging upstream (Fig. 6) due to the influence of the small artificial pond. During the first 5 weeks *Gomphonema parvulum* was the dominant species, reaching more than 60%. However, changes were evident

Table 4
PCA eigenvectors

	Factor 1	Factor 2
Density	−0.48	**−0.88**
Species number	**0.98**	−0.11
Diversity (H')	**0.92**	−0.34

Bold values indicate $P < 0.05$.

immediately after dredging in site 2, when this species decreased with the increase of different taxa, such as *Sellaphora pupula*, *N. palea*, *Diadesmis confervacea*, *Gomphonema minutum*, *Nitzschia brevissima*, *Diadesmis contenta*, *Pinnularia microstauron*, *Eolimna subminuscula*, *Luticola mutica*, *N. amphibia*, among others. At the end of study period, *S. pupula* was the dominant species and *G. parvulum* and *N. palea* the subdominant taxa.

In site 3, changes in species composition were less evident than in other sites. *G. parvulum* was always the dominant species; taxa such as *N. palea*, *S. pupula*, *Nitzschia umbonata*, *Achnanthidium minutissimum*, *N. amphibia*, *E. subminuscula*, and *Pinnularia gibba* were recorded with lower abundance (Fig. 6).

4.2.2. Species tolerance

The analysis of species tolerance to organic pollution and eutrophication showed that, prior to dredging, site 1 was characterized by a great percentage (60%) of species sensitive to pollution (Fig. 7). After dredging the proportion of tolerant species ($p < 0.05$) increased significantly in detriment of sensitive species ($p < 0.01$), which did not reach 20% in abundance during the last weeks of the study.

Contrarily, in sites 2 and 3 the changes in the species tolerance were not statistically significant owing to the high nutrient concentrations and oxygen demands already existing in the pre-dredging period that caused the dominance of tolerant and most tolerant species and a low proportion of sensitive ones.

5. Conclusions and discussion

During dredging the increases in turbidity and suspended solids produce important changes in the light climate of the water body, thus reducing light penetration. As a consequence, an increase in nutrients was expected that might affect the biomass of the primary producers (Armengol, 1998). Fine, non-living particles in suspension interfere with light penetration and constitute an important environmental factor in the growth and distribution of both attached and planktonic algae (Reynolds, 1996).

In the present study we could establish that the effects of dredging in the stream involve two types of disturbances: (i) in the stream bed, by the removal and destabilization of the substrate and (ii) in the water column, by generating chemical changes and an alteration of the light environment of the stream. The effects of this

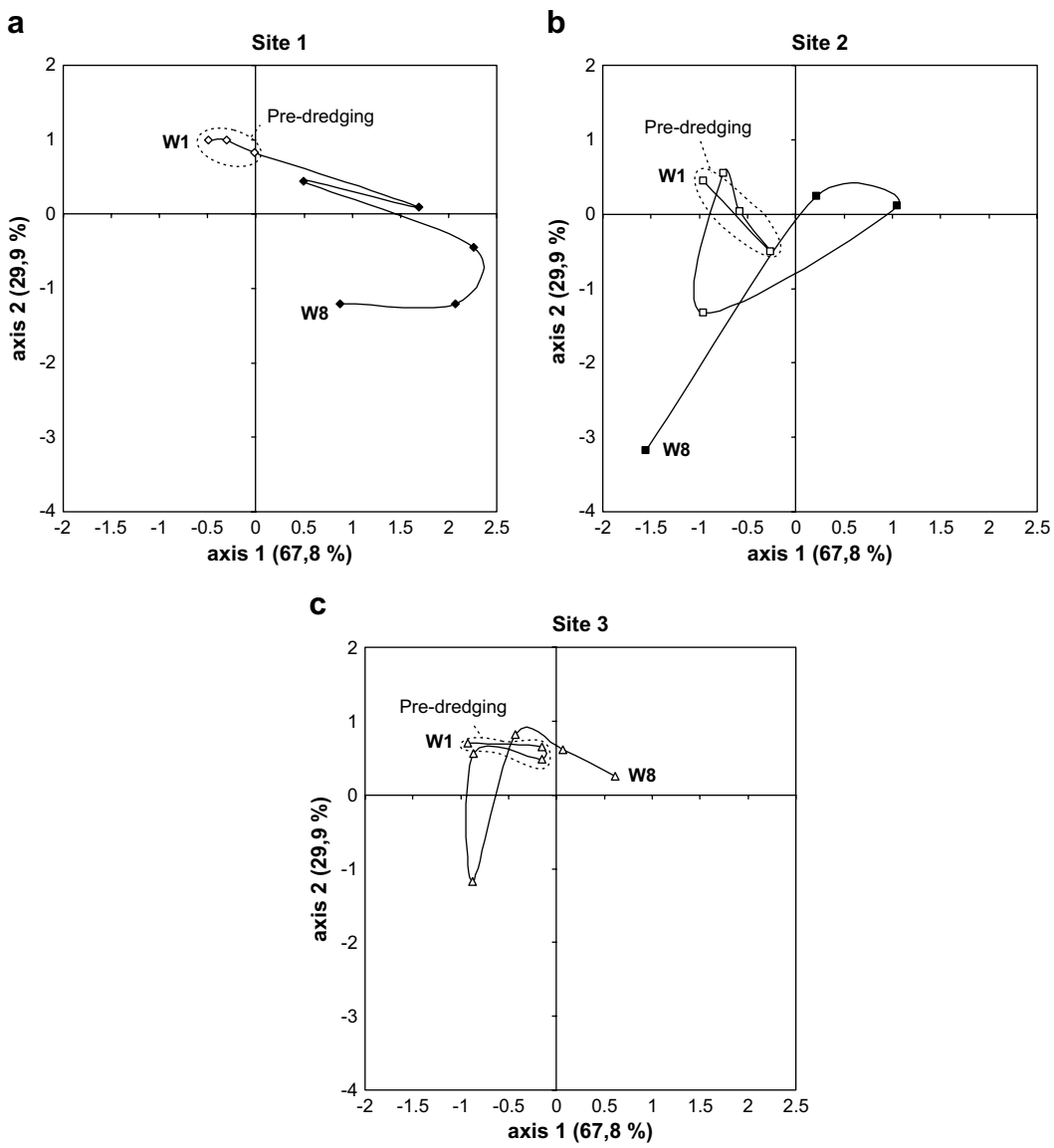

Fig. 4. Biplot of the PCA scores performed on the structural variables of the diatom assemblages in sampling sites 1 (a), 2 (b), and 3 (c). The dotted area groups the samples taken before the starting of the dredging works in site 1 (first 3 weeks). Solid dots indicate the samples taken after dredging at each sampling site.

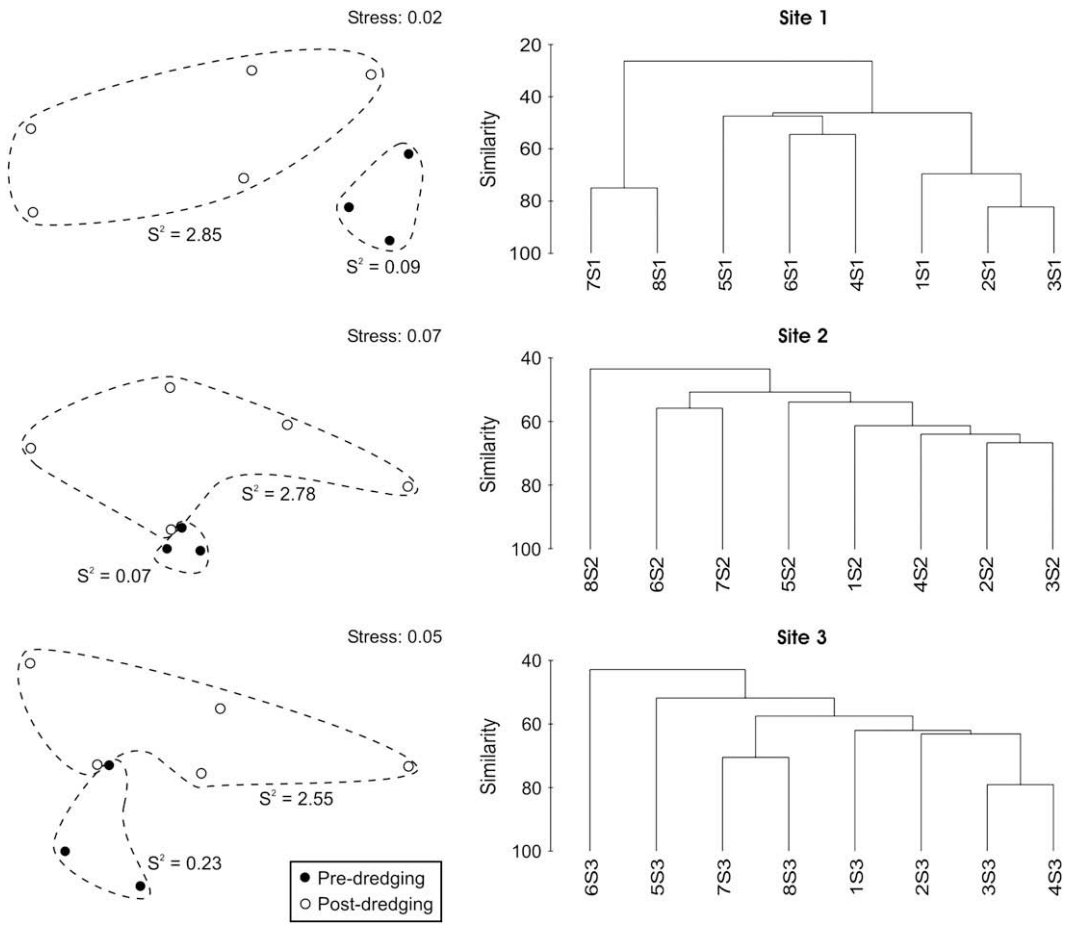

Fig. 5. Differences in diatom assemblage composition of the three sites: pre- and post-dredging, based on $\log_e(x+1)$ transformed diatom abundances and Bray–Curtis similarities. MDS plot and clustering. Estimates of the variation of pre- and post-dredging samples are shown.

type of disturbance included, during a first stage, an increase in the amount of suspended solids, and subsequently an increase in the amount of nutrients.

Physical and chemical modifications in the habitat of benthic diatoms from the Rodríguez Stream produced changes in the assemblage that included modifications in its structural parameters as well as changes in the tolerance of the assemblage to organic pollution and eutrophication. Benthic diatoms' density decreased immediately after the dredging and it started to increase in the subsequent weeks, as a consequence of the increase in the nutrients load in the water column. According to Burkholder (1996) in turbid habitats, impacted by sediment loading/resuspension, periphyton actually can thrive in part because of enhanced access to nutrients from close association with the incoming sediments that become an accumulated substratum. Our results are coincident with those reported by Lewis et al. (2001), who found significant increments in algal density after the dredging works in a bay in Florida. These authors also describe a non-significant decrease in the diversity indices between pre- and post-dredging periods. However, we found that diversity and species numbers increased in the sites where dredging works were carried out, while variations were only minor in site 3.

The responses observed during the post-dredging period, mainly an increase in diversity, are coincident with those occurring during the first successional stages that are characterized by the existence of empty niches (Begon et al., 1999).

The effects of dredging on benthic diatoms assemblage of the Rodríguez Stream were also evidenced by changes in the specific composition related to the tolerance of the species to organic pollution and eutrophication. In agreement with observations reported by many authors in relation to changes of the assemblage in response to disturbances of the aquatic environment (Cattaneo et al., 1998; Guasch et al., 1999; Gustavson and Wängberg, 1995; Kelly and Whiton, 1995; Lange-Bertalot, 1979; Pan et al., 2000; Rott, 1991; Sabater, 2000; Symoens et al., 1988; Ivorra, 2000), our study shows that the disturbances produced by dredging determine a decline of the sensitive species that are replaced by either tolerant or most tolerant species. These changes were particularly noticeable and statistically significant in site 1 (characterized by its lower amount of nutrients and organic matter previous to dredging), which showed an increase in the amount of nutrients and oxygen demands as a consequence of sediment removal. However, these changes were not so conspicuous in sites 2 and 3 that already presented a marked water quality deterioration before the execution of the dredging works that caused the dominance of tolerant and most tolerant species prior to dredging.

Descy and Coste (1990) mention the fact that diatoms are scarcely sensitive to physical modifications of the aquatic ecosystems (canalizations, bottom perturbations, etc.) and that the determinism of the structure of diatom communities is mainly linked to the water chemical characteristics. Chemical changes and substrate instability usually go together as a result of a dredging work, and it is very difficult to differentiate their effects on organisms. In our study, however, and since site 3 was not dredged during the study period, changes in the diatom assemblage in this site were the response only to changes in the water column, while

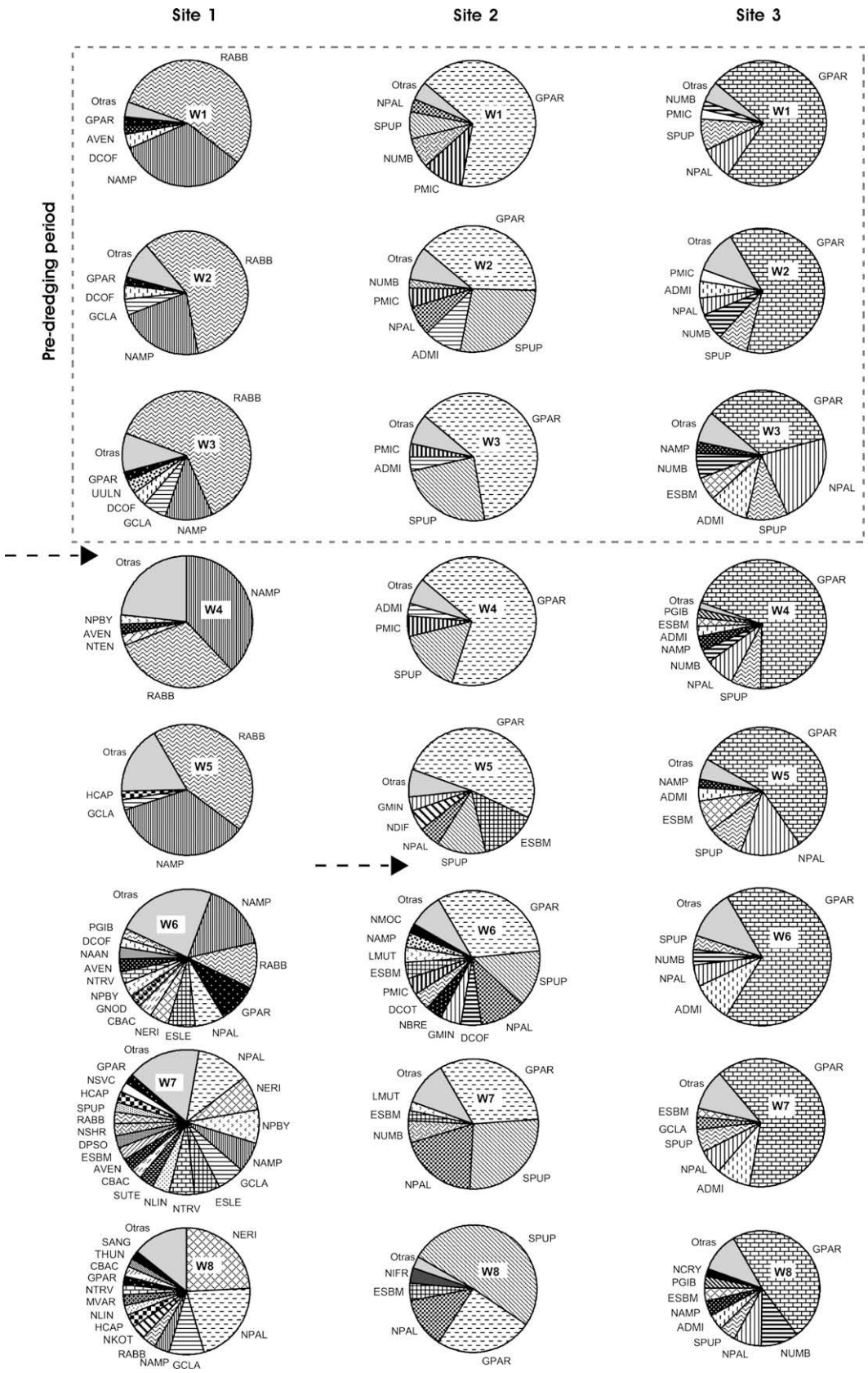

Fig. 6. Diatom composition during the pre- and post-dredging periods in the sampling sites. The arrows indicate the time of dredging in sites 1 and 2. For abbreviation of species names see Table 3.

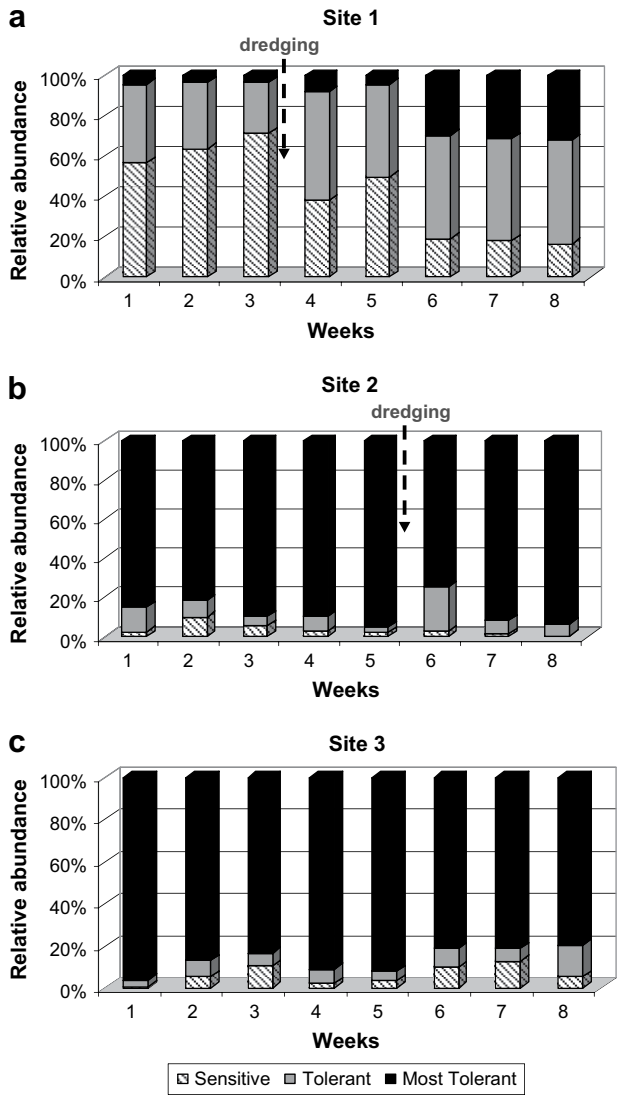

Fig. 7. Species tolerance to eutrophication and organic pollution in sampling sites 1 (a), 2 (b) and 3 (c) throughout the study period. The start of the dredging works in sites 1 and 2 (November 2001) is indicated.

those observed in sites 1 and 2 were the combined response to changes in both the water column and the sediments.

The results herein presented emphasize the need to conduct further studies on the impacts of dredging on the biota, especially in plain lotic systems, where this is a usual practice with still barely known effects. Moreover, this research demonstrated the value of using benthic diatom assemblages to assess dredging impact on aquatic systems.

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