The effect of flow on the macrozoobenthos structure in a re-opened oxbow lake: a case study of the Słupia River, northern Poland

KRYSTIAN TADEUSZ OBOLEWSKI, KATARZYNA GLIŃSKA–LEWCZUK & SZYMON KOBUS

Department of Land Reclamation and Environmental Management, University of Warmia and Mazury, Plac Łódzki 2, 10-756 Olsztyn, Poland

obolewsk@apsl.edu.pl

Abstract This study focuses on the assessment of relationships between flow and macrozoobenthos structure that was performed in a re-opened oxbow lake, Osokowy Staw, located near the Słupia River, northern Poland. Macrozoobenthos samples were taken between 1999 and 2005 at six sampling sites near the shore and four in the middle of the water body. In July 2000 the Osokowy Staw was re-opened and connected with the river through PVC pipes which enabled free water inflow and outflow. After re-opening, macrozoobenthos density increased from 99 to 659 individuals m^{-2} and the wet biomass from 0.03 to 73.1 g wet weight m^{-2}, although these increases were not statistically insignificant. In the closed Osokowy Staw the dominant species was *Asellus aquaticus*. After re-opening it was replaced by bivalves and Chironomidae larvae (during the first year) and then, in 2005, *A. aquaticus* became the most abundant again. The number of taxa increased from four in the closed water body to 17 during the first year after re-opening and 14 in the next year. The Shannon biodiversity index also increased from H′ = 0.35 (in 1999) to H′ = 1.5 (in 2005). Revitalization processes of the re-opened oxbow lake were connected with the qualitative and quantitative recolonization by riverine macrozoobenthos.

Key words oxbow lake; macrozoobenthos; reconnection; Słupia River, Poland

INTRODUCTION

Oxbow lakes are bodies of water that have been poorly described in terms of biological conditions. Only a few reports have been published on the macrozoobenthos of oxbow lakes located in catchments of the southern Baltic Sea: in the Warta River (Hajduk & Hajduk, 1984; Jezierska-Madziar *et al.*, 2000; Gromadzińska-Graczyk, 2005), the Wkra River (Lewin, 2001), in central (Piechocki, 2006) and northern Poland (Obolewski, 2006; Obolewski & Glińska-Lewczuk, 2006; Obolewski *et al.*, 2009).

When it comes to studies of Polish Pomeranian rivers, only a few reports on macrozoobenthos properties of oxbow lakes in the Słupia basin have been published as a preliminary for wider studies (Obolewski, 2006; Obolewski & Glińska-Lewczuk, 2006).

Distribution of macrozoobenthos is determined by the whole set of biotic and abiotic factors (Kajak, 1988). Important factors identified in previous studies include: water temperature (Wells & Demas, 1979), predator pressure (Dermott, 1988), phytoplankton biomass (Rasmussen, 1988), detritus mass (Drake, 1984), particle size of bottom deposits, substrate configuration and type (Opaliński, 1971; Higler, 1981).

The aim of the present study was to determine the distribution and the structure (abundance, wet weight) of macrozoobenthos in the re-opened oxbow lake of the Słupia River. The data acquired will help to determine the quantitative changes in benthic macrofauna in revitalisation of oxbows exposed to various human activities. This paper is also intended to outline the distribution and the abundance of benthic fauna as a food base for animals, including fish species of economic importance.

MATERIALS AND METHODS

Study area

The oxbow is situated in the Słupia River catchment upstream of the city of Słupsk, northern Poland. It was created between 1915 and 1922 by cutting off the right-bank meander. The area of the oxbow amounts to 670 m^2 with a depth of 1 m. It is localized at 38 km of the river course. As a result of river straightening, a part of the river meander was disconnected from the main channel of the Słupia River. Since then, the newly formed oxbow, Osokowy Staw, has undergone advanced autogenic biogeochemical processes leading to its degradation. In July 2000, this hypertrophic and polymictic water body was reconnected to the main channel by installing two PVC pipes in the upstream and downstream arms to let the water freely flow into and out of the lake. The inlet and outlet pipes were 9 and 12 m long, respectively, with diameters of 160 mm to allow the passage of fish species inhabiting the Słupia River (Fig. 1).

The pipe lengths were related to the distance between the arm and the river channel. To avoid silting up, the pipes were installed at a slope of 20°. Flow measured at the inflow and outflow using a flowmeter (SEABED, UK), showed similar values (mean Q = 0.55 m^3/s), whereas in the middle part of the reservoir the flow was significantly lower and amounted to 0.13 m^3/s. The flow velocity and discharges showed temporal changes related to hydrological conditions. In months of accelerated water circulation, e.g. March–April when snow and ice melt occurs, average discharge values were up to 0.7 m^3/s (Fig. 2). The lowest flows occurred from late summer to mid-winter (January–February).

The flow initiated through the oxbow lake has significantly changed the grain size composition, as well as chemical properties of bottom sediments (Obolewski & Glińska-Lewczuk, 2006). The relatively small cross-section of the installed pipes allowed for the control of changes taking place within the water body. Moreover, the system was designed for *in situ* monitoring of ecological transformations from lentic to lotic habitats, e.g. the observation of macrozoobenthos, whose representatives are important in the assessment of ecological state of aquatic ecosystems, just like ichthyofauna or macrophytes.

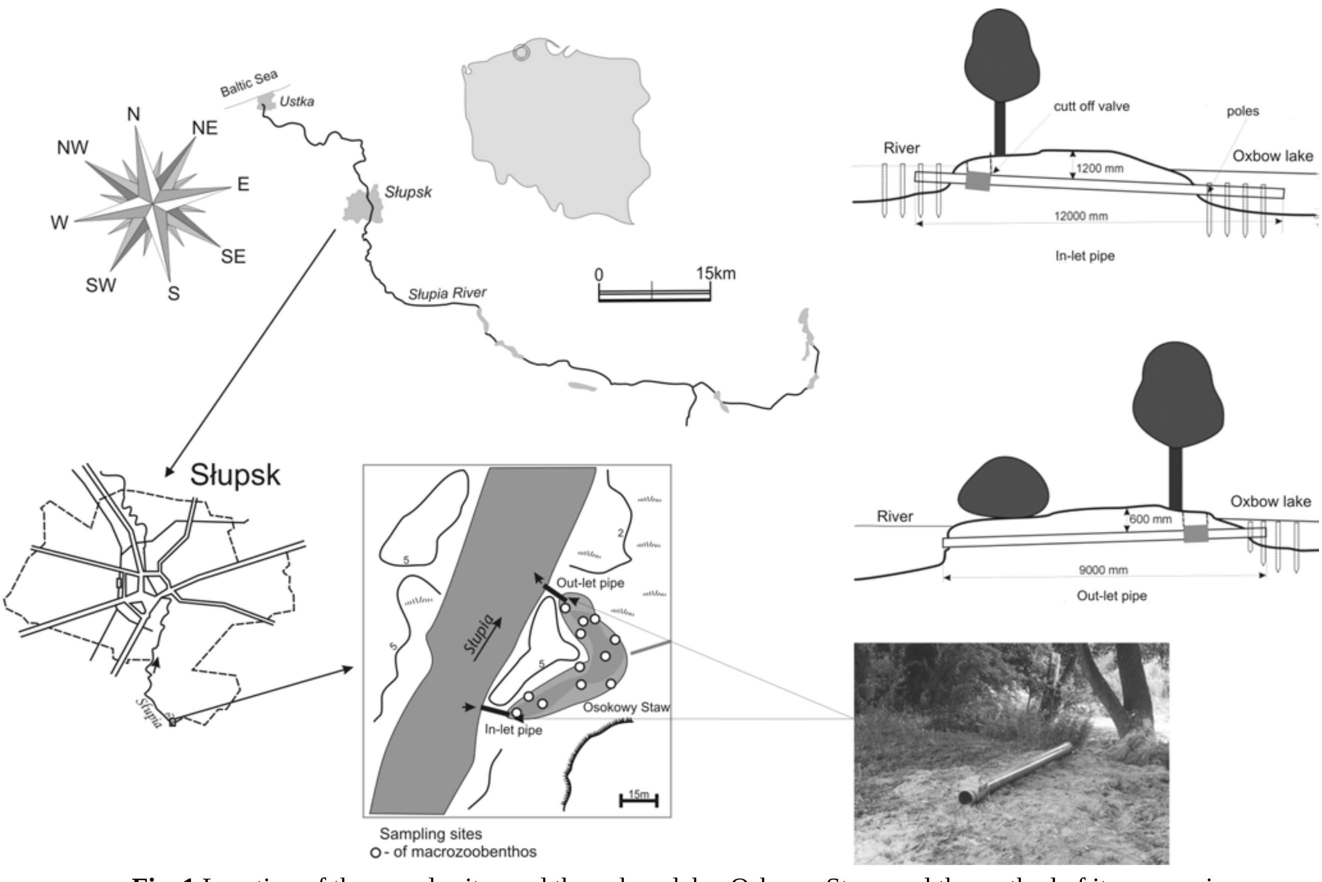

Fig. 1 Location of the sample sites and the oxbow lake, Oskowy Staw, and the method of its re-opening.

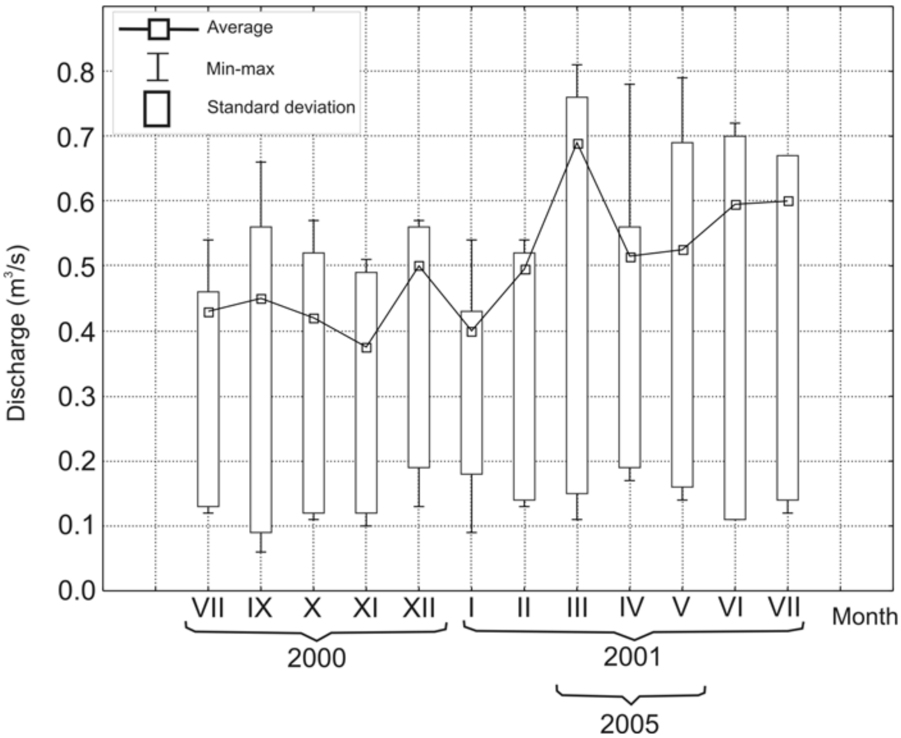

Fig. 2 Monthly changes in discharge (m^3/s) in the re-opened Osokowy Staw.

Soon after reconnection, pike (*Esox lucius* L.) and roach (*Rutilus rutilus* L.) were observed in the oxbow lake. The connection has caused rapid changes in the water level both in the river and the oxbow. The river effects were also manifested in similar oxygen content and temperature in the oxbow and river (Glińska-Lewczuk *et al.*, 2009, this issue).

Sampling

Macrozoobenthos of the disconnected Osokowy Staw was sampled seasonally from autumn 1999 until summer 2000 (altogether 20 samples). After re-opening, benthos was sampled from summer 2000 (20 samples) to spring 2005 (10 samples). Six sampling sites were established in the shallow zone of the bottom area that had no rooted plants such as nympheids or helophytes. The remaining four sites were located in a deeper zone.

At sites 4, 5 and 6, about 0.5 m deep, the bottom was covered with silt and fine sand. The bottom at sites 1, 2, and 3 was covered with mud locking a grainy mineral fraction. Sites I–IV (at a depth of 0.45–2.5 m) had a muddy bottom mixed with mollusc shell breccia. The bottom of sites in the river was composed of fine sand. The shallow water benthos samples (sites 1–6) were collected with a 80 cm^2 Wolnomiejski grab and from the central deeper water (sites I–IV) with a 225 cm^2 Ekman-Birge grab. Four grabs were collected for each sample at each site. Immediately after collection, the sediment was sieved through a 0.5 × 0.0197 mm mesh sieve and the sieve residue was fixed in 4% formaldehyde. Oligochaeta, Hirudinea, Trichoptera, Isopoda, Mollusca and Chironomidae larvae were identified to species, while other invertebrates were identified to a higher taxa level. The biomass was determined by weighing individuals belonging to different taxa with 0.01 g accuracy. Prior to weighing, the organisms were blotted with a filter paper. In the preliminary analysis the following factors were calculated: density (D_A) and biomass (D_m) dominance indices, frequency (Fr) and Shannon biodiversity index (H′).

DO, temperature, conductivity and pH were measured *in situ* using portable probes, previously calibrated (Hach-Lange HQD40). Two-litre water samples were collected directly into acid-washed polycarbonate bottles at a depth of 20 cm and placed in ice. On the same day, samples were filtered through Whatman® GF/F glass fibre filters (pre-combusted at 450°C for 4 h) to determine the amount of suspended, dissolved and ash-free solids. Ion chromatography was used to determine dissolved nutrient and ion concentrations (Cl^-, SO_4^{2-}, NO_3^-, NO_2^-, NH_4^+, PO_4^{3-}, HCO_3^-, Mg^{2+}, Fe^{2+}) while spectrophotometry (Hach-Lange cuvette tests) was used to determine total nitrogen and total reactive phosphorus (Hermanowicz *et al*., 1999).

Statistical analysis

To assess the general differences among wetland invertebrate data (grouped into classes/order or species) and environmental variables were subjected to non-parametric analysis of variance (Kruskal-Wallis test, $p \leq 0.05$). To reduce the effect of absolute values, the invertebrate abundance density data were square-root transformed ($\sqrt{X + 1}$), rare species were downweighted and environmental data were log-transformed ($\log_{10}(X + 1)$ (ter Braak & Šmilauer, 2002). To further identify patterns in the abundance of 23 invertebrate genera, direct ordination methods were used, such as Correspondence Analysis (CA) (ter Braak & Šmilauer, 2002).

To identify the primary environmental gradients affecting macro invertebrate assemblages, we applied Canonical Correspondence Analysis (CCA; ter Braak, 1995) to three environmental variables and 10 groups of macroinvertebrates. Because environmental gradients had not previously been evaluated in the study area, we ran a manual, forward-selection CANOCO procedure, which included variables that had a conditional effect significant at the level of 10% ($p \leq 0.1$). P-values were calculated using the Monte Carlo Permutation Test (ter Braak & Šmilauer, 2002). Once the main environmental gradients affecting the aquatic community structure were identified, the model was subjected to variation partitioning in order to quantify the amount of variation uniquely explained by each set of variables in the model (arms, seasons, time), the shared variance among them and the variance that remained unexplained (Borcard *et al.*, 2004).

Generalized additive models were used to test the relationship between the invertebrate community (measured by abundance and diversity) and key environmental variables (Wood, 2004). A non-parametric analysis of variance (Kruskal-Wallis test, $p < 0.05$) was used to identify differences in richness and abundance among types of connectivity.

Values of the Shannon-Wiener diversity index for invertebrates in different parts of the oxbow were calculated from local mean densities of the taxa identified and then were averaged.

RESULTS

Environmental conditions

Mean annual values, from the period of 1999–2005, for the measured physical and chemical parameters, and significant differences among hydrological connectivity oxbow lake are shown in Table 1. Differences were observed between years as well as between the river and oxbow lake mainly in inorganic ion concentration (HCO_3^-, NO_3^-, PO_4^{3-}). While the river is the main source of those constituents, leaf litter, sediment scouring, hillslope-runoff inputs, agricultural irrigation and rainfall also made substantial contributions. The closed oxbow lake had significantly higher conductivity than the re-opened reservoir. The re-opened Osokowy Staw had significantly higher DO concentrations in 2005. The shift of Osokowy Staw from a lentic to a lotic habitat was caused by permanent hydrological connection with the main river channel, which might have increased primary productivity through higher nutrient availability. Finally, the re-opened Osokowy Staw contained higher concentrations of both organic and inorganic elements in 2005.

Patterns of macro invertebrate abundance

Before re-opening of the Osokowy Staw oxbow lake, the macrozoobenthos was represented by four taxa. The overall mean macrozoobenthos density was 0.99 indie. 10^2 m^{-2}, and the overall mean biomass amounted to 0.3 g m^{-2} (Table 2). The most abundant taxa was *Asellus aquaticus* L.

Table 1 Water quality of “Osokowy Staw” and the Słupia River. Data are mean (SD) of monthly samples in 1999–2005.

	Słupia River		Closed Osokowy Staw	Re-opened Osokowy Staw		
	2000 (n = 6)	2001 (n = 12)	1999 (n = 12)	2000 (n = 9)	2001 (n = 12)	2005 (n = 10)
T (^{o}C)	9.0 (3.2)	8.8 (5.7)	15.5 (2.7)	15.2 (10.9)	10.9 (5.6)	12.8 (3.6)
pH	7.4 (0.3)	6.3 (0.6)	7.7 (0.4)	7.1 (0.06)	7.2 (0.08)	7.6 (0.15)
DO (mg/L)	6.2 (2.4)	7.2 (3.6)	3.6 (5.5)	5.2 (4.2)	8.8 (2.2)	8.5 (1.87)
COD (mg/L)	87.7 (15.4)	35.9 (10.8)	9.7 (1.63)	9.0 (1.4)	10.5 (0.5)	24.9 (2.97)
BOD (mg/L)			10.3 (0.51)	9.7 (0.21)	5.9 (0.90)	5.5 (0.65)
Conductivity (μS/cm)			943.3 (1036.3)	492.5 (237.0)	335 (47.5)	322.2 (45.9)
NO_3^- (mg/L)***	0.53 (0.24)	0.26 (0.19)	0.16 (0.05)	0.17 (0.15)	0.22 (0.23)	0.14 (0.18)
DON (mg/L)**	0.04 (0.06)	0.03 (0.07)	0.42 (0.23)	0.15 (0.04)	0.27 (0.10)	0.12 (0.21)
PO_4^{3-} (mg/L)**	0.18 (0.11)	0.10 (0.05)	0.84 (0.48)	0.60 (0.09)	0.52 (0.64)	0.57 (0.32)
DOP (mg/L)**	0.15 (0.09)	0.11 (0.04)	0.70 (0.48)	0.35 (0.28)	0.23 (0.17)	0.24 (0.01)
SO_4^{2-} (mg/L)	22.9 (6.01)	22.7 (3.77)	17.4 (9.71)	87.5 (90.1)	36.3 (3.1)	24.3 (1.0)
Cl^- (mg/L)***	1.78 (0.04)	5.16 (3.31)	44.8 (15.0)	21.0 (0.5)	20.9 (7.8)	11.1 (0.2)
HCO_3^- (mg/L)	164 (87)	125 (71.6)	191 (65.0)	289 (201.5)	167 (22.1)	184.5 (76.4)
Mg^{2+} (mg/L)**	5.24 (0.15)	5.38 (0.22)	4.17 (2.15)	15.08 (24.40)	8.25 (7.28)	11.09 (12.31)
Fe^{2+} (mg/L)	0.38 (0.25)	0.36 (0.36)	0.86 (0.46)	0. 39 (0.21)	0.45 (0.15)	0.33 (0.11)

T = temperature, DON = dissolved organic nitrogen, DOP = dissolved organic phosphorus. * = significant differences (nonparametric Kruskal-Wallis test, $p \leq 0.05$) among research periods between river and oxbow lake in the same year, ** = significant differences (nonparametric Kruskal-Wallis test, $p \leq 0.05$) among research periods in oxbow lake.

Table 2 Invertebrate composition and metrics in the Osokowy Staw and the Słupia River. Data are mean density and biomass (s.d.) from seasonal samples in 1999, 2001 and 2005.

	River Słupia		Closed Osokowy Staw				Re-opened Osokowy Staw			
	2001 (n = 24)		1999 (n = 20)		2000 (n = 20)		2001 (n = 20)		2005 (n = 10)	
R*	17		4		8		17		14	
N*	54.1		0.99		1.25		3.55		6.6	
B*	20.54		0.31		3.81		7.60		73.11	
H	0.74		0.35		0.82		0.95		1.50	
J	0.66		0.58		0.91		0.83		0.46	
	A	B	A	B	A	B	A	B	A	B
Nanidae	0	0	0	0	0	0	0	0	0.01 (0.01)	0.01 (0.01)
Oligochaeta*[1,2]	9.5 (11.71)	0.7 (0.75)	0	0	0.22 (0.12)	0.3 (0.02)	0.74 (1.23)	0.33 (0.76)	0.40 (0.23)	0.4 (0.32)
Hirudinea***[1,2]	1.1 (0.55)	0.5 (0.85)	0	0	0	0	0.20 (0.78)	1.8 (2.89)	0.70 (0.98)	6.9 (4.11)
Crustacea***[1,2]	4.0 (4.81)	0.6 (0.73)	0.75 (0.34)	0.3 (0.87)	0	0	0.44 (0.03)	0.2 (0.04)	4.80 (11.34)	53.0 (25.76)
Diptera*[1]	18.1 (11.8)	0.4 (0.57)	0.24 (0.21)	0.01 (0.51)	0.10 (0.31)	0.2 (0.05)	1.33 (2.45)	0.5 (5.4)	0.20 (0.13)	0.2 (0.12)
Trichoptera*[1]	1.1 (0.0)	0.02 (0.3)	0	0	0	0	0.04 (0.01)	0.07 (0.2)	0	0
Ephemeroptera	0	0	0	0	0.10 (0.2)	0.01 (0.01)	0	0	0	0
Hydrachnella	0	0	0	0	0	0	0	0	0.20 (0.35)	0.5 (0.31)
Gastropoda**[1]	0.4 (0.65)	0.04 (0.5)	0	0	0.23 (0.45)	0.2 (1.6)	0	0	0.02 (0.01)	0.1 (0.08)
Bivalvia***[1,2]	20.0 (20.51)	18.4 (17.1)	0	0	0.60 (0.52)	3.1 (2.2)	0.80 (1.11)	4.7 (5.37)	0.30 (1.0)	12.0 (15.87)

R = Invertebrate richness, N = Total invertebrate abundance, B = Total invertebrate biomass, H = Invertebrate Shannon diversity, J = Invertebrate evenness, * = significant differences (nonparametric Kruskal-Wallis test, $p \leq 0.05$) between river and oxbow lake in the same year (1 – density, 2 – biomass), ** = significant differences (nonparametric Kruskal-Wallis test, $p \leq 0.05$) in oxbow lake (1 – density, 2 – biomass).

which occurred at three sites and had a mean density of 0.75 indie. 10^2 m^{-2} (D_A = 76%) and biomass reached 0.3 g m^{-2} (D_m = 97%).

In the samples collected from the lentic oxbow, a high diversity of Chironomidae larvae was identified comprising *Procladius* Skue larvae (Fr = 25%) of the subfamily *Tanypodinae* as well as *Chironomus* sp. (Fr = 50%) of the subfamily *Chironomidae* and *Sergentia* sp. The *Chironomus* sp. was dominant in the northwestern zone of the reservoir, while *Procladius* on the eastern bank constituted 100% of all Chironomidae larvae. The Chironomidae density was 0.5 individuals per 10^2 m^{-2}. The densest assemblages of Chironomidae larvae occurred in the western zone of the reservoir.

In the re-opened oxbow, during the first year after connection, the macrozoobenthos was represented by 4–12 taxa. The mean macrozoobenthos density was 4.8 individuals per 10^2 m^{-2}, and the overall mean biomass amounted to 11.4 g m^{-2}. The abundance of macrozoobenthos in the opened oxbow was 6-fold higher and the wet weight was 2-fold higher in spring 2001 than in autumn 2000. The macrozoobenthos in Spring 2005 consisted of 12 taxa of different levels which was 3-fold higher than before the re-opening. In 2005 the overall mean density was 6-fold higher, while biomass was almost 20-fold higher than before re-opening (Fig. 3).

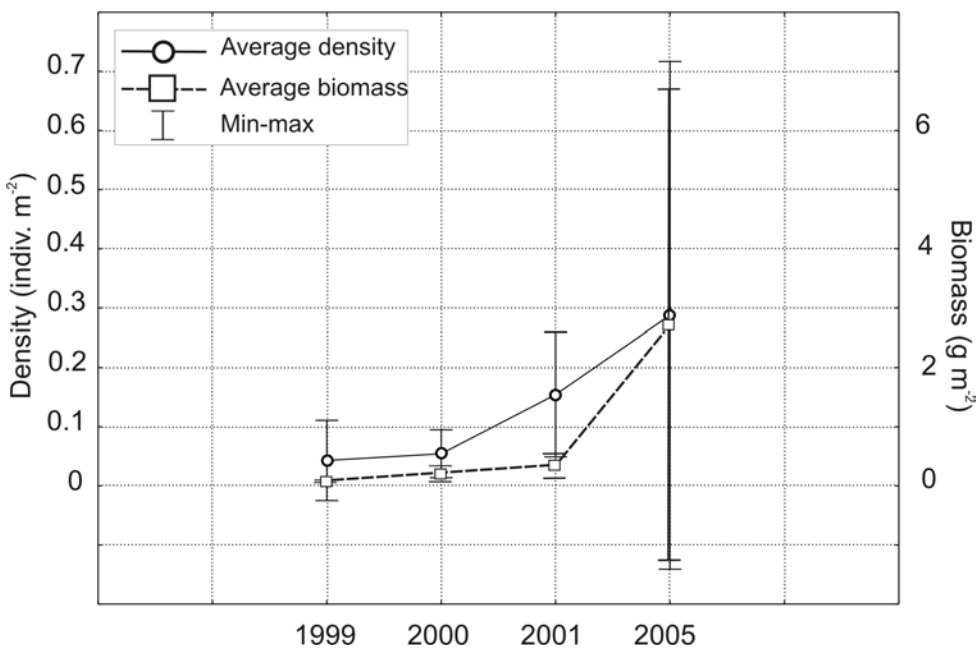

Fig. 3 Average density and biomass of invertebrates in the Osokowy Staw over time.

Oligochaetes were represented in the reopened oxbow lake biotope by 4 species belonging to the Tubificidae: *Limnodrilus hoffmeisteri* (Clap.), *Potamothrix bavaricus* (Oesch.), *Potamothrix hammoniensis* (Mich.) and *Potamothrix moldaviensis* (Vejd. Et Mr).

The overall mean Tubificidae densities ranged from 0.7 to 8.7 individuals per 10^2 m^{-2}, with the highest mean density occurring next to the inlet tube. The overall tubificid biomass at the opening of the oxbow lake remained stable and amounted to 1.1 g wet weight m^{-2}. In the deepest zone of the water body Oligochaetes were represented by only two Tubificidae species. In spring 2005 Oligochaetes were represented by one Tubificidae family, and its mean density was lower than in the first year after opening, while the overall mean biomass was higher (Table 2).

The Hirudinea were represented in the re-opened oxbow by three species: *Erpobdella octoculata* (L.), *Helobdella stagnalis* (L.) and *Glossiphonia complanata* (L.), with an overall mean density of 0.2 individuals per 10^2 m^{-2} (Table 2). They occurred numerously in the northeastern zone of the pond. The overall mean biomass was the highest in spring 2001 (5.5 g wet weight m^{-2}, 30% of the total wet weight in that season). In spring 2005 leeches were co-dominant. The mean Hirudinea density and biomass were 4-fold higher in the second year after re-opening compared to the first year after re-opening.

Crustaceans were represented by two species: *A. aquaticus* L. and *Gammarus fossarum* Koch. The overall mean density of isopods was high (4.0 indiv. 10^2 m^{-2}, Fr = 33%, D_A = 11%) while the overall mean biomass was low (0.2 g wet weight m^{-2}, D_m = 3%), (Table 2). In spring 2005 *A. aquaticus* was dominant compared to all the species in the oxbow as regards its density and biomass (D_A = 73%, D_m = 73%). After re-opening of the oxbow lake the mean density of isopods was 500-fold higher and the mean biomass was 53-fold higher (Table 2). The gammarids occurred only in winter 2000 next to the western bank. Their overall mean density and biomass were the lowest in the first period after re-opening.

Four species of Chironomidae larvae were identified after re-opening. The most common, present at sampling sites 1, 3 and 5, were *Chironomus* sp. (Fr = 50%), *Dicrotendipes* sp. (Fr = 11%) of the subfamily Chironomidae, *Polypedilum* sp. and *Sergentia* sp.

The Chironomid density was 0.7–26.0 individuals per 10^2 m^{-2} and the densest population occurred within the eastern zone of the reservoir. The highest density of Chironomidae larvae was observed at sampling site 3. The lowest density and biomass of Chironomidae larvae were measured in spring 2005.

Trichoptera larvae (*Odontocerum albicorne*) were observed in spring 2001 in the middle zone of the reservoir (northeastern bank) and had a mean density of 0.4 individuals per 10^2 m^{-2}, and biomass of 0.07 g wet weight m^{-2}. No Trichoptera larvae were observed in spring 2005.

Molluscs were represented in the opened oxbow by six species: *Bithynia tentaculata* (L.), *Lymnaea stagnalis* (L.) and *Anisus vortex* (L.) and bivalves comprising *Pisidium amnicum* (O.F.Müller), *P. subtruncatum* (Malm), *Sphaerium corneum* (L.). The gastropods mean density was 0.3 individuals per 10^2 m^{-2} and the bivalves mean density ranged from 2.7 to 18.7 individuals per 10^2 m^{-2}. The densest assemblages of gastropods and bivalves occurred within the oxbow next to the inlet and outlet tubes. In the central part of the reservoir three species of *P. subtruncatum* were observed. In the second year after re-opening, the mean density of bivalves decreased by 3-fold when compared to the first year, while the mean biomass was 2.5-fold higher.

The Shannon–Wiener biodiversity indices were calculated from overall mean densities. The biodiversity of macrozoobenthos in the closed oxbow lake was lower than in the re-opened Osokowy Staw (H′ > 1).

Influence of environmental variables on invertebrates

The relationships between environmental conditions and invertebrate taxa were analysed by RDA (Redundancy Analysis) (Fig. 4). The X- and Y-axis described 27.3% and 4.2% of eigenvalues, respectively. There appeared to be relationships between season and *Ephemera* sp. as well as *Dendrocoleum lacteum*, whereas the time since re-opening correlated with the density of *A. aquaticus*. Under lentic conditions we only identified two taxa. The rapid inflow of well-aerated and nutrient-rich water to the oxbow lake caused a temporary destabilization of the system. During the first year after re-opening there was a significant increase in the number of taxa, but the majority of these were unable to permanently inhabit the newly-formed ecosystem (Fig. 4). Only *D. lacteum*, Tubificidae and Hirudinea still existed four years after re-opening. However, Hydrachnellae appeared for the first time in 2005. Among the macrozoobenthos taxa identified, three were positively related to the increased flow, but the water movement and changes in environmental conditions (e.g. coarser substratum) were associated with the disappearance of four taxa (Fig. 4).

The results obtained were confirmed by PCA (Principal Component Analysis). The first PCA axis explained 48.3% of the total variance, while the second axis explained 18.3%. It indicated the role of time in stabilizing the system favouring the presence of Hydrachnellae and Hirudinea species. The PCA analysis showed a significant effect of season on Chironomidae larvae and

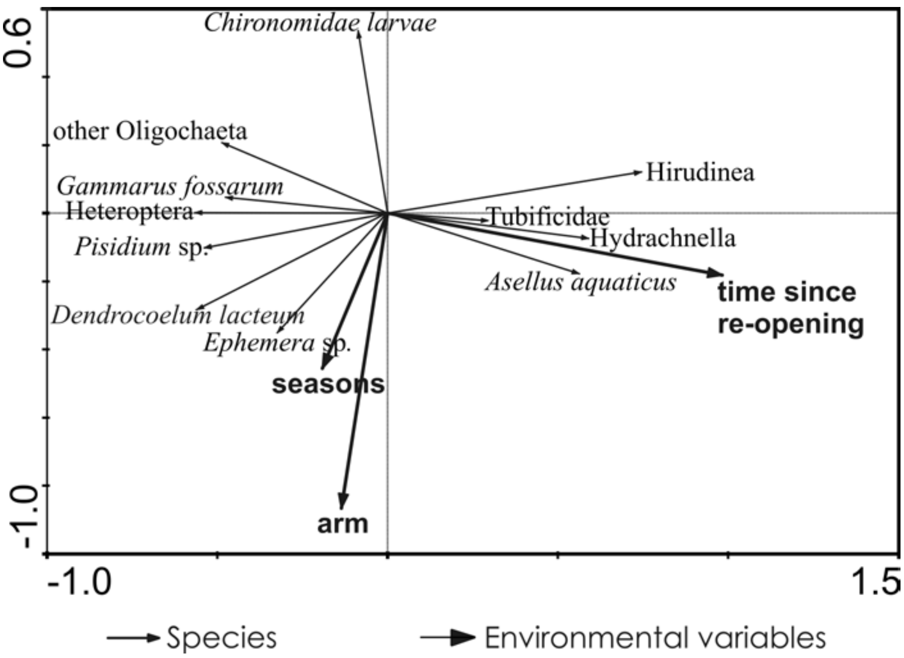

Fig. 4 Redundancy analysis (RDA) between environmental variables (nominal) and macrozoobenthos taxa.

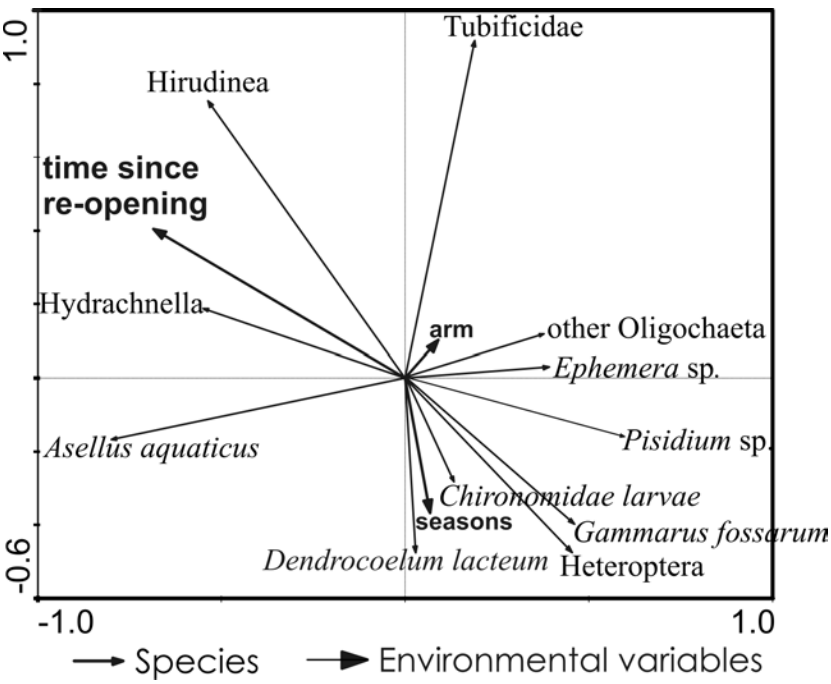

Fig. 5 PCA diagrams of the effects of re-opening on the presence of macrozoobenthos taxa in Osokowy Staw.

Dendrocoleum lacteum, and some effect on Heteroptera and *Gammarus fossarum* species (Fig. 5). In the case of *D. lacteum* only, the results of both PCA and RDA were identical, with no difference between years. CCA analysis showed no effect of location of a sampling site.

Taxa were identified which responded positively to the oxbow re-opening and which settled permanently in the new conditions. A distinct clustering of the benthos organisms was identified in relation to the environmental conditions over time. The input of river water to the oxbow lake caused a distinct effect on the benthic zooplankton, the majority of which appeared after the oxbow re-opened.

DISCUSSION

The taxonomic diversity, density, and biomass of macrozoobenthos in the Osokowy Staw oxbow lake was within the ranges typical for small freshwater reservoirs. Cut-off oxbow lakes create unfavourable conditions for the development and existence of benthos organisms. Poor environmental conditions, especially low DO concentrations, and intensive mineralization processes in the bottom water layer cause the disappearance of many species in cut-off river channels, even after their appearance in oxbow lakes after spring and summer river overflows (Slavik & Bartoš, 2001). This is proved by the low biodiversity (H′) and low density and biomass of bottom macrofauna in the Osokowy Staw, as well as by the similarity of the stands of similar environmental conditions and bottom. Only representatives of bottom macrofauna with a high tolerance of poor environmental conditions, such as larvae of dipterans, can freely penetrate the bottom of oxbow lakes. The cut-off Osokowy Staw was dominated by *Chironomus* sp. which represents a typical situation for strongly eutrophic ecosystems with a thick layer of bottom sediment (Ali *et al.*, 2002; Gallardo *et al.*, 2008).

The analysis of higher macrozoobenthos taxa in the River Warta oxbow lake (Jezierska-Madziar *et al.*, 2000) and in the Słupia oxbow lake (Osokowy Staw) showed considerable similarities. In both oxbow lakes there were five common taxa: Tubificidae, Hirudinea, Gastropoda, Trichoptera and Chironomidae while biodiversity indicators oscillated around H′ ≈ 2.5. In the oxbow lake investigated here a larger diversity of species was found in the re-opened lake, up to 12 taxa of macrozoobenthos. In addition to the taxa mentioned above, the oxbow lake bottom was inhabited by bivalves (*Pisidium* and *Sphaerium*), isopods (*A. aquaticus*) and gammarids of the *Gammarus* genus. In particular the occurrence of *Gammarus fossarum* is evidence of the improvement in oxygen conditions (Bachmann & Usseglio-Polatera, 1999). *Asellus* present in the cut-off Osokowy Staw did not adapt to the new environmental conditions (Arakelova, 2001). The inflow of river water had an especially big influence (increase in biomass and decrease in density) on the development of the Sphaeriidae species. They might have entered the water through the inlet and outlet pipes or been carried by waterfowl nesting on the banks of the oxbow lake (Marklund & Sandsten, 2002). As in the Osokowy Staw, other oxbow lakes have also been found to be dominated by Bivalvia, namely, *Pisidium subtruncatum* and *Sphaerium corneum*, but more species were observed in the other oxbow lakes (Piechocki, 2006), whereas there were only three species in the Słupia oxbow lake investigated. Representatives of Gastropoda occurred sporadically in the oxbow lake investigated. Only two species, *Bithynia tentaculata* and *Anisus vortex*, were noted in 2005 at sites with constant contact with river water. More typical species for oxbow lakes is *A. vortex*, whereas *B. tentaculata*, a typical river species, occurs occasionally (Lewin, 2001). The appearance of the representatives of caddis-flies influences the reduction of phytoplankton. It leads to the end of eutrophication in oxbow lakes that were made passable and did not create conditions for accumulating biogenic compounds brought by river waters (Hoffsten, 1999). After re-opening of the Osokowy Staw we also observed a 1.5-fold decrease in zooplankton abundance and a 22-fold increase in phytoplankton abundance (Obolewski, 2005).

The concentration of macrozoobenthos increased from 99 to 659 individuals per m^{-2} over time after the oxbow lake was re-opened. Wet mass, in turn, increased more than 240 times (from 0.03 to 73.1 g wet weight m^{-2}). River regulation processes, in this case incorporating oxbow lakes into the river channel network, improved conditions for organisms occupying the bottom of the water body under research (Drake, 1984). Also the similarity in terms of the concentration of control stands in the contact zone of the river and oxbow lake water can be observed. The inflow of river water does not influence considerably the way of forming the cluster of biomass around water contact zones in both ecosystems. Biomass in the oxbow lakes appeared to be controlled by

Bivalvia, which were carried with river water into the centre of the lake and then grouped themselves in places of moderate current or in terminal stretches, where water leaves an oxbow lake. A similar situation was observed in the river Słupia where most of the biomass was at the control sites with a muddy bottom and calm current.

Making oxbow lakes passable can especially affect the representatives of Bivalvia, Prosobranchia, Crustacea and Diptera (Bachmann & Usseliglio-Polatera, 1999). This is important as bottom fauna constitutes considerable food resources for fish which find food, shelter and spawning ground in newly-passable oxbow lakes (Slavík & Bartoš, 2001). Unfortunately, in the oxbow lake of the Słupia, and also in the Osokowy Staw, no detailed research on fish ecology was conducted and it is difficult to assess the role of benthos for fish fry and adult specimens of different fish species. Predatory species are expected to be particularly favoured by the re-opening of oxbow lakes and research carried out in the oxbow lakes of the Pilica and the Warta rivers in Poland showed that the majority of fish species are common to river–oxbow lake ecosystems (Penczak *et al.*, 2004). The biggest role in terms of food resources among the developing macrozoobenthos in the newly-passable Osokowy Staw was played by the isopods and gammarids which constituted food for salmonids and roach. An important role may be played by the representatives of Trichoptera, which are readily eaten by trout (Lehane *et al.*, 2001). Other fish species that may thrive in passable oxbow lakes are *Cyprinidae*, *Cobitidae*, *Siluridae*, *Esocidae* and *Percidae* (Golski *et al.*, 2004). Therefore, it can be consistently asserted that unblocking of the oxbow lake Osokowy Staw was very advantageous to the development of benthos, and indirectly to other components of the aquatic ecosystem, e.g. fish, phyto- and zooplankton (e.g. Hoffsten, 1999).

CONCLUSIONS

– From the results obtained in this study, as well as some literature data, it can be concluded that, as a result of the re-opening and improvement of environmental conditions (mainly increased DO concentration), macrozoobenthos re-established in the bottom layer in the Osokowy Staw.
– The re-opening had positive effects on the qualitative and quantitative composition of the aquatic ecosystem in the oxbow lake. The number of taxa dwelling on the lake bottom increased, along with a biodiversity indicator. The density grew over time and macrozoobenthos biomass increased rapidly after re-opening.
– The sampling sites in both arms of the oxbow were similar only in terms of biomass. It is difficult to notice the similarity of the stands in terms of density.
– Re-establishment of the benthic fauna is expected to constitute an important food source for fish in the newly-passable oxbow lake.

Acknowledgements This scientific study was financed by educational funds as research project no. PBZ/MNiSW/07/2006/07.

REFERENCES

Ali, A., Frouz, J. & Lobinske, R. J. (2002) Spatio-temporal effects of selected physico-chemical variables of water, algae and sediment chemistry on the larval community of nuisance Chironomidae (Diptera) in a natural and a man-made lake in central Florida. *Hydrobiol.* **470**(1–3), 181–193.

Arakelova, K. S. (2001) The evolution of individual production and scope for growth in aquatic sow bugs (*Asellus aquaticus*). *Aquatic Ecol.* **35**(1), 31–42.

Bachmann, V. & Usseglio-Polatera, P. (1999) Contribution of the macrobenthic compartment to the oxygen budget of a large regulated river: the Mosel. *Hydrobiol.* **410**, 39–46.

Borcard, D., Legendre, P., Avois-Jacquet, C. & Tuomisto, H. (2004) Dissecting the spatial structure of ecological data at multiple scales. *Ecology* **85**, 1826–1832.

Dermott, R. M. (1988) Zoobenthic distribution and biomass in the Turkey lakes. *Can. J. Fish. Aquat. Sci.* **45**(Suppl. 1), 107–114.

Drake, J. A. (1984) Species aggregation: the influence of detritus in a benthic invertebrates community. *Hydrobiol.* **112**, 109–115.

Gallardo, B., Garcia, M., Cabenza, A., González, E., González, E., Ciancarelli, C. & Comin, F. A. (2008) Macroinvertebrate patterns along environmental gradients and hydrological connectivity within a regulated river-floodplain. *Aqiut. Sci.* **70**, 248–258.

Glińska-Lewczuk, K., Kobus, Sz., Obolewski, K. & Chormański, J. (2009) *The Effect of Groundwater Supply on the Hydrochemical Diversity of Floodplain Lakes in the Temperate Climatic Zone*. IAHS Publ. 328. IAHS Press, Wallingford, UK (this issue)

Golski, J., Jezierska-Madziar, M., Pińskwar, P. & Gromadzińska-Graczyk, H. (2005) Initial characteristic of species composition ichtiofauna selected oxbows Warta River on section Konin – Poznań. In: *Oxbows Lake Essential Element of the River Ecosystem* (ed. by M. Jezierska-Madziar), 129–141. AR Press, Poznań, Poland.

Gromadzińska-Graczyk, H. (2005) Characteristics of epiphytic fauna and macrozoobenthos in selected oxbow lake of the Warta River. AR Press, Poznań, Poland (in Polish).

Hajduk, D. & Hajduk, Z. (1984) Hirudinea central and the lower Biebrza River and it oxbow lake. *Wiadomości Parazyte*. **XXX**(2), 241–248 (in Polish).

Higler, L. W. G. (1981) Bottom fauna and littoral vegetation fauna in Lake Maarsseveen. *Hydrobiol. Bull.* **15**, 82–86.

Hoffsten, P. (1999) Distribution of filter-feeding caddisflies (Trichoptera) and plankton drift in a Swedish lake-outlet stream. *Aquatic Ecol*. **33**(4), 377–386.

Jezierska-Madziar, M., Pińskwar, P., Gromadzińska, H., Galant, P. & Górecki, K. (2000) Qualitative and quantitative structure of the benthic fauna of an oxbow lake (cut-off meander) of the water river at Luboń in relation to the elevated concentrations of fluorine compounds. *Folia Univ. Agric. Stetin* **214** Piscaria (27), 93–106.

Kajak, Z. (1988) Benthos. In: *Ecology of Inland Waters*, 235–313. PWN-Polish Sci. Publ., Warszawa, Poland (in Polish).

Lehane, B. M., Walsh, B., Giller, P. S. & O'Halloran, J. (2001) The influence of small-scale variation in habitat on winter trout distribution and diet in an afforested catchment. *Aquatic Ecol*. **35**(1), 61–71.

Lewin, I. (2001) *Concentration of the Gastropods in Sweet Water in the Reservoirs and RIvers Ciechanowska Wysoczyzna*. UŚ Press, Katowice, Poland (in Polish).

Marklund, O. & Sandsten, H. (2002) Reduction of benthic macroinvertebrates due to waterfowl foraging on submerged vegetation during autumn migration. *Aquatic Ecol.* **36**(4), 541–547.

Obolewski, K. (2005) Reaction of phytoplankton on re-opening of the Słupia oxbow lake. In: *Oxbows Lake Essential Element of the River* (ed. by M. Jezierska-Madziar), 70–83. AR Press, Poznań, Poland (in Polish).

Obolewski, K. (2006) Oxbows – important element of river valleys in the example of the Słupia River. *Infrastruktura i Ekologia Terenów Wiejskich, PAN Kraków* **4**(2), 99–108 (in Polish).

Obolewski, K. & Glińska-Lewczuk, K. (2006) Contents of heavy metals in bottom sediments of oxbow lakes and the Słupia River. *Polish J. Envir. Stud.* **15**(2a), part II, 440–444.

Obolewski, K., Glińska-Lewczuk, K. & Kobus, Sz. (2009) Assessment of water quality influence on qualitative and quantitative structure of epifauna inhabiting *Stratiotes aloides* L. – case study for the Łyna oxbow lake. *J. Element.* **14**(1), 119–135.

Opaliński, K. W. (1971) Macrofauna communities of the littoral of Mikołajskie lake. *Pol. Arch. Hydrobiol*. **18**, 275–285.

Penczak, T., Galicka, W., Głowacki, Ł., Kruk, A., Kostrzewa, J., Marszał, L., Koszaliński, H. & Zięba, G. (2005) Meaning of oxbows lake to keeping the diversity and the abundance ichtiofauna in the river ecosystem. In: *Oxbows Lake Essential Element of the River* (ed. by M. Jezierska-Madziar), 95–128. AR Press, Poznań, Poland (in Polish).

Piechocki A. (2006) Species composition, frequency and density of the Sphaeriidae (Bivalvia, Heterodonta) in different freshwater habitats. *Heldia* **6**(3/4), 161–178.

Rasmussen, J. B. (1988) Littoral zoobenthic biomass in lakes, and its relationship to physical, chemical, and trophic factors. *Can. J. Fish. Aquat. Sci*. **45**, 1436–1447.

Slavík, O. & Bartoš, L. (2001) Spatial distribution and temporal variance of fish communities in the channelized and regulated Vitava river (Central Europe). *Envir. Biology Fishes* **61**(1), 47–55.

ter Braak, C. J. F. (1995) Ordination. In: *Data Analysis in Community and Landscape Ecology* (ed. by R. H. G. Jongman, C. J. F. ter Braak & O. F. R. van Tongeren), 91–173. Cambridge University Press, Cambridge, UK.

ter Braak, C. J. F. & Šmilauer, P. (2002) CANOCO Reference Manual and CanoDraw for Windows User's Guide: Software for Canonical Community Ordination (version 4.5). Microcomputer Power, Ithaca, NewYork, USA.

Wood, S. N. (2004) Stable and efficient multiple smoothing parameter estimation for generalized additive model. *JASA* **99**, 673–686.