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Colonisation by midges (Chironomidae, Diptera) of recently-created shallow ponds: implications for the restoration of lacustrine fringing wetlands

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Abstract – As part of a wetland conservation programme for the Southern shore of lake Neuchatel (Western Switzerland), an experiment was started in 1993 to slow down the terrestrialization of the wetland. Sediments were removed in a reed-belt adjacent to an existing pond, at three different depths (0.20, 0.30 and 0.40 m), thus creating a new shallow pond. The colonisation by aquatic vegetation and invertebrates was monitored during three years (1994–1996) to evaluate the effect of the creation of a new pond on the chironomid diversity and abundance through time. Chironomids (78 taxa) were among the most diverse group colonising this fringing wetland. Significant differences in abundance were found between the reed-belt and the pond habitats (existing and recently created ponds). As expected, the latter was more productive, especially during the first two years of colonization. However, the chironomid diversity in the newly created pond, measured by the rarefied richness, was intermediate between the reed-belt (low diversity) and the pre-existing pond. Ordination of the sites on the basis of their chironomid assemblages confirmed this trend. The depth at which the sediment was scrapped off, had no major influence upon the chironomid assemblages. Finally, within the three years after creation of open water habitats, the production of chironomids and consequently their availability for dabbling ducks and wetland birds, were enhanced. However, this time was not enough for the chironomid community to reach the diversity of the pre-existing pond, to which it is connected. The creation of shallow ponds seemed to be crucial to slow down terrestrialization processes in this lacustrine wetland and to have only slight effects on the chironomid communities.

Key words: Fringing wetland / pond / colonisation / wetland management / Switzerland

Introduction

Wetlands represent some of the most threaten ecosystems on the planet and pose some of the most contentious questions to both scientists and policy makers (Maltby *et al.*, 2000). They occupy about 6% of the world's land surface (Maltby and Turner, 1983) and occur on all continents. Estimated wetland loss in Europe is large, as exemplified by France (67%: 1900–1995), Germany (57%: 1950–1985) and Italy (66%: 1938–1984) (Maltby *et al.*, 2000). In Switzerland, a considerable reduction in wetland area occurred during the last century and induced negative effects resulting from a reduction in aquatic connectivity and from water level regulation. Protection of these dynamic ecosystems requires a sound management for the conservation and/or enhancement of the unique species assemblages associated with wetland habitats. Wetland management often incorporates mowing, burning or grazing to slow down vegetation encroachment, as well as the creation of open water habitats for biodiversity maintenance or productivity enhancement for fish and waterfowl (Rehfisch, 1994; Sanders, 2000). However, little information is available about the impact of such restoration measures upon different components of the aquatic biodiversity.

Wetland invertebrates are vital links between primary production and vertebrate consumers (Murkin and Wrubleski, 1988). Among them, midge flies (Chironomidae) belong to the most abundant aquatic insects in freshwaters marshes (Wrubleski, 1987), and are therefore very important food for fish and water birds (Murkin *et al.*, 1982; Maher and Carpenter, 1984; Batzer *et al.*, 1993). Their distribution and abundance are also accurate indicators of physical, chemical and nutrient conditions

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*Corresponding author: `brigitte.lods@vd.ch` of wetland habitats (Johnson *et al*., 1995). Chironomids also belong to the first colonizers of newly created or rewetted wetlands (Layton and Voshell, 1991; Batzer and Wissinger, 1996; Wrubleski, 1999, 2005).

The study site, the largest fringing wetland of Switzerland (the "Grande-Cariçaie") suffers, as many other such wetlands, from terrestrialization through forest expansion on its land-ward side. Pilot projects were developed to control invasion by shrubs and to prevent the degradation of reed-belts. Maintenance operations as the creation of ponds were also used as a method to enhance open water habitats (Gander, 2003; Sager and Clerc, 2006).

The objective of this study was to evaluate the influence upon chironomid community structure (composition, diversity and abundance) of the creation of a new shallow pond in a fringing wetland. Community succession was monitored in three habitat types: the new shallow pond, an adjacent reed (*Phragmites australis*) bed and two pre-existing ponds, one of which was connected to the new one. In term, we sought to assess the efficiency of the "reed-belt scraping" method of sediment removal in mitigating terrestrialization of freshwater marshes and to determine its usefulness for wetland restoration.

Site and methods

Study site

The study was carried out in the "Grande-Cariçaie", the largest fringing wetland of Switzerland, located on the South-eastern shore of Lake Neuchâtel. This wetland was created as a consequence of the lowering of the lake water level during the "first correction of Jura waters" (1880) and now forms an almost continuous belt of 42.5 km length and 480 m average width. After the second correction of Jura's hydrological system (completed in the 1970's), mean annual water level fluctuations were reduced below 1 m. Management programmes were set up in 1987 to prevent the drying up of the wetland, to restrain the advance of the forest, to ensure the survival of all types of fringing wetland species and to slow down the erosion of the lakeshore.

In January 1993, an experiment was started to slow down the terrestrialization of the wetland, by sediment removal in a reed-belt (*Phragmites australis* (Cavanilles) Steudel) adjacent to an older pond (pond B) created in 1988 ("Font" sector) (6°48'45"E, 46°50'30"N, altitude 430 m a. s. l.) (Fig. 1). Reed-belt rhizomes and sediments were removed on a surface area of 5300 m^2 and at three different depths (0.20, 0.30 and 0.40 m) in order to test the recolonisation rate of the reed-belt.

Mean annual air temperature is 10.0 °C, mean annual precipitation is 84 mm (data Swiss Forecasting Institute). Freezing of the open water areas did not exceed 10–20 days (January and February) during the sampling programme (1994–1996). Water physico-chemical parameters were monitored from January 1997 to June 1998 (Castella-Müller, 2004). Water was well mineralised (average electric conductivity: 468 $mS.cm^{-1}$), average nitrates and total phosphorus concentrations (NO_3-N: 0.11 $mg.L^{-1}$; Ptot: 0.03 $mg.L^{-1}$) were low. The area was connected with the lake either by seepage through the littoral sandy dunes or through the adjacent pond that receives lake water at high water levels (Table 1).

Sampling regime

Twelve sampling stations were selected in three different habitats in order to monitor the colonisation by midges: (i) the untouched reed-belt (RB) (five stations representing different degrees of connection with the adjacent pond and covering a moisture gradient, n° 2, 3, 5, 8 and 10); (ii) two older ponds created in 1988 (pond A (PA): station 1; pond B (PB): stations 11 and 12) considered as references; and (iii) the newly-created pond (NCP) (1993) (four stations, n° 4, 6, 7 and 9, scraped at three different depths) (Table 1). The adult chironomid fauna was monitored during three years (1994–1996) monthly or bi-monthly from April to late August. Each month, adults were collected using two emergence traps per station (1075 cm^2 each), left in place for five consecutive days.

After sorting, the male adult chironomids were stored in 70% ethanol and specimens were mounted on microscopic slides in a mixture of lactic acid, glycerol and polyvinylic alcohol (Reymond, 1994). Identification to species level was carried out when possible using Lehmann (1970), Pinder (1978), Lindeberg and Wiederholm (1979), Reiss and Säwedal (1981), Rossaro (1985), Contreras-Lichtenberg (1986), Wiederholm (1989) and Saether (1995). The female chironomids were only counted.

Data analyses

Given the heterogeneity of abundances between samples, the taxonomic richness was calculated using the rarefaction procedure (Heck *et al*., 1975; Krebs, 1999). Rarefaction simulates random draws of a fixed number of individuals within the samples (or combined samples) to be compared. The number of individuals drawn is based upon the least abundant sample. Rarefied richness is not an estimate of the total community richness, but it allows an unbiased comparison between samples of unequal abundances. It can also be regarded as a diversity measure, because, for a given number of observed taxa in a sample, rarefied richness will increase with the evenness of the distribution of abundance between taxa. Calculations were performed with the function "rarefy" from the "vegan" library in the R software (R Development Core Team, 2009). The function calculates the rarefied richness for a given number of individuals from the Hurlbert (1971) formula, together with a standard error following Heck *et al*. (1975).

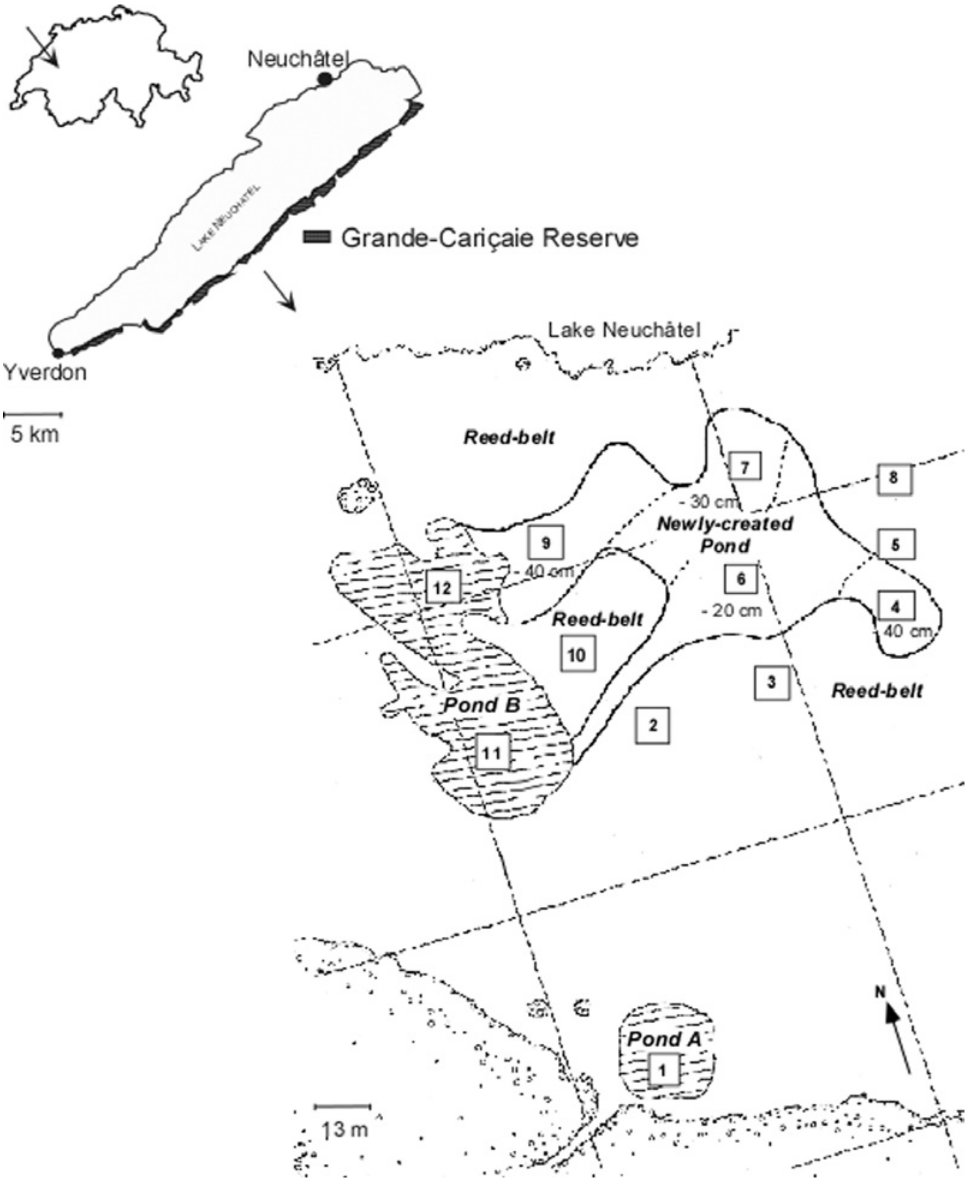

Fig. 1. Location of the sampling stations in the study area. The old ponds are A and B.

Table 1. General characteristics of the sampling stations and their dominant vegetation.

Site	Station	N°	Date of creation	Depth (m)	Surface area (m^2)	Dominant vegetation	Remarks
1	Reed-belt (RB)	10, 2	*ca.* 1880	0.00–0.20	/	*Phragmites australis*	Connected with the pond B from March to May
1	Reed-belt (RB)	3, 5, 8	*ca.* 1880	0.00–0.20	/	*Phragmites australis*	Permanently connected with pond B
2	Newly-created pond (NCP)	6	1993	0.2	5300	*Myriophyllum spicatum*	Permanently connected with pond B
2	Newly-created pond (NCP)	7	1993	0.3	/	*Typha latifolia*	Permanently connected with pond B
2	Newly-created pond (NCP)	4	1993	0.4	/	No vegetation	Permanently connected with pond B
2	Newly-created pond (NCP)	9	1993	0.4	/	*Myriophyllum spicatum*	Permanently connected with pond B
3	Pond B (PB)	11, 12	1988	1.00–1.50	3100	*Myriophyllum spicatum*	Connected with the lake
4	Pond A (PA)	1	1988	0.10–0.40	730	*Chara hispida*	Connected with the lake during floods

Table 2: Comparison of chironomid densities between sites and years. Results of the Mann–Whitney U test (significant results are shown in bold, $p < 0.05$). For abbreviations of sites, see Table 1.

Sites/Year	1994	1995	1996
1–2	**0.000**	**0.002**	0.205
1–3	**0.003**	**0.001**	0.050
1–4	**0.023**	**0.036**	0.169
2–3	0.348	0.502	0.422
2–4	0.454	0.621	0.336
3–4	0.149	0.679	0.768

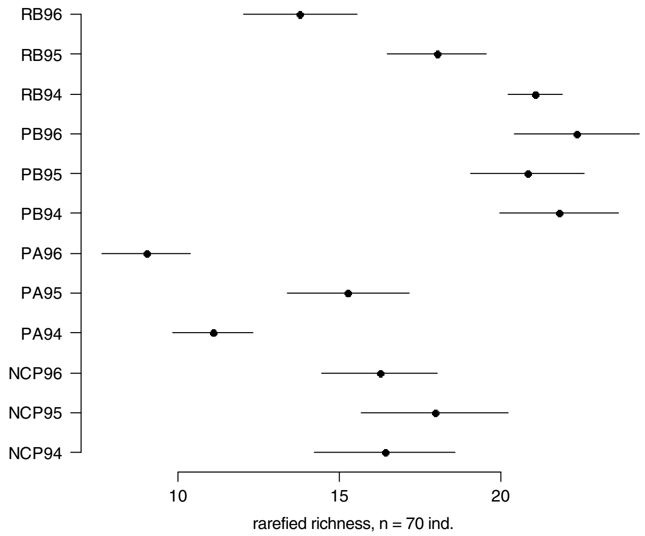

Fig. 2. Rarefied richness (average number of chironomid taxa for 70 individuals ± 1 SE), calculated for the four sites at each of the three years.

Mann–Whitney U-test was used to examine whether there were significant differences concerning number of taxa and density between habitats and depths. Correspondence analysis (CA), associated with between-class correspondence analyses (Dolédec and Chessel, 1987, 1989; Thioulouse *et al.*, 1997) were used to ordinate the samples in terms of taxonomic composition and to describe and quantify the differences between sites and dates. All multivariate analyses were carried out using the ade4 library (Chessel *et al.*, 2004) for the R software (R Development Core Team, 2009).

Results

Taxonomic composition, diversity and abundance

During the study period, about 7000 adult chironomids were caught, in which female made up 55 to 82% in the reed-belt (RB), but only 38 to 56% in the ponds. Seventy-eight taxa (59 identified to species level belonging to 34 genera) were identified: 10 Tanypodinae (9 genera), 16 Orthocladiinae (8 genera), 43 Chironomini (14 genera) and 9 Tanytarsini (4 genera) (Table 2). The rarefied richness (expressed as the average number of taxa for 400 individuals, all dates combined) had the highest value in pond B (PB) (44.1 ± 1.8), followed by the reed-belt (41.1 ± 0.9) and the newly created pond (NCP) (39.6 ± 2.4). Pond A (PA) had the lowest richness (29.9 ± 1.5). The differences were significant except between RB and NCP where the confidence limits around the mean overlapped.

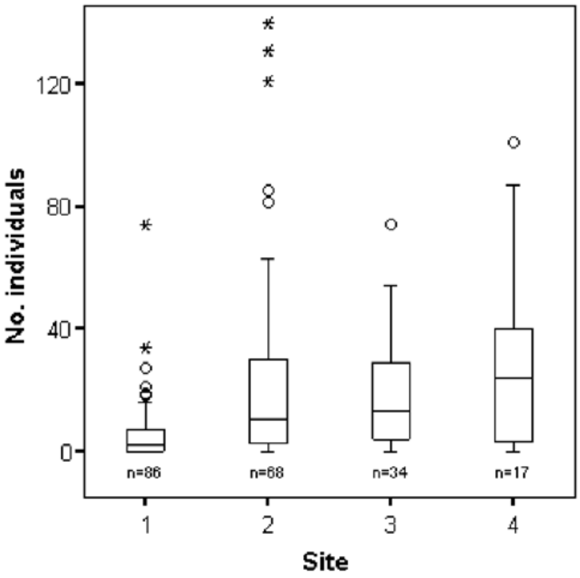

Fig. 3. Box-whisker graphs showing the distribution of chironomid abundance (individuals/0.2 m^{2}) in the four sites (1, reed-belt; 2, new-created pond; 3, pond B; 4, pond A) during April–late August 1994 to 1996; *n*: numbers of stations × dates. The box indicates the interquartile range (Q25–Q75) and contains the median (horizontal line).

When the dates were considered separately (Fig. 2), significant differences in chironomid diversity were measured between years only in pond A and in the reed-belt. This latter fact could be in relation with the greater variability of the water level during the three years sampling. There was no significant changes over the years in pond B and in the newly created pond.

Only six species had a relative abundance above 5% of the total number of individuals: *Tanypus punctipennis* (12.4%), *Xenopelopia nigricans* (5.2%) (Tanypodinae), *Cricotopus sylvestris* (6.1%), *C. intersectus* (5.0%) (Orthocladiinae), *Tanytarsus excavatus* (9.2%) and *T. mendax* (7.2%) (Chironominae-Tanytarsini). Species of the Chironominae-Chironomini contributed less than 3%. *Endochironomus dispar* and *Polypedilum sordens* dominated, followed by *Chironomus pallidivittatus* and *Parachironomus parilis*.

Overall, the mean density of chironomids emerging from the pond habitats (the pre-existing ponds (PA, PB) and the newly-created one (NCP)) was always higher (64.9–163.4 individuals.m^{-2}) than from the reed-belt (9.9–43.5 individuals.m^{-2}) (Mann–Whitney *U*-test, $p < 0.001$) (Fig. 3). However, the temporal trend showed that the chironomid density was significantly higher in the three pond sites than in the reed-belt during the two first years.

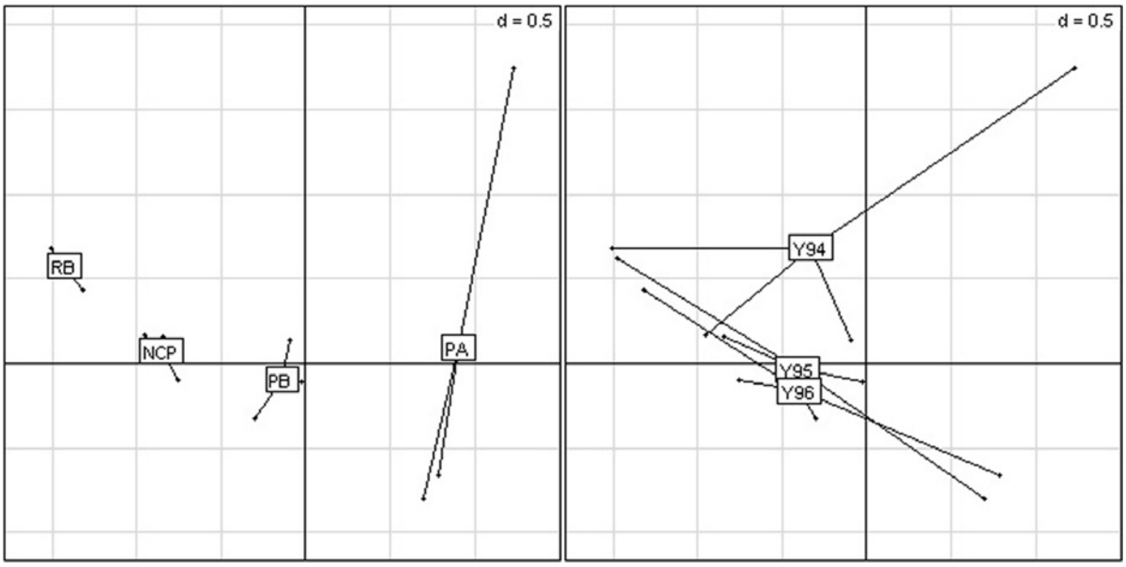

Fig. 4. Ordination of the 12 site∗years by correspondence analysis (first factorial plane, horizontal axis = 21% of variability, vertical axis = 17%). Left: site∗years grouped according to the four sites, right: site∗years grouped according to the three years.

In 1996, the difference was only slightly significant between the reed-belt and pond B. This could be related to low densities in the three ponds during the third year of study (Table 2).

Ordination of the four sites on the basis of their chironomid assemblages over time

The ordination of the 4 sites∗3 years by a correspondence analysis (Fig. 4) showed a clear separation of the sites along the F1 axis. The between-site variability accounted for 42% of the total faunal variability and was significant (permutation test, $p = 0.001$). The chironomid composition in NCP was intermediate between RB and the adjacent PB, even after three years. The between-year effect accounted for 18% of the faunal variability and was not significant ($p = 0.71$). The highest between-year variability was observed in the small pond A.

The faunal data table was reorganized according to the first axis scores of the ordination (Table 3). It revealed contrasted trends in the six dominant species. *Tanypus punctipennis* was present in all sites but with higher densities in NCP; its decreased with time after sediment removal suggesting its pioneer character. An increase in density through years was evident for *Cricotopus intersectus*, *C. sylvestris* and *Tanytarsus mendax*. They could be considered as secondary colonisers of the new open water habitat. *Tanytarsus excavatus* was a characteristic species of the permanent pond, associated with submerged vegetation and it occurred in large numbers in pond A. *Xenopelopia nigricans* was localized almost exclusively in the flooded reed-belt (stations 5 and 8) together with *Zavrelimyia* sp. (station 3) at a lower abundance and *Microtendipes diffinis* in July in stations 2 and 10 when the reed-belt had drying up. Most of these characteristic species were bi- or multivoltine except for *Xenopelopia nigricans* which was univoltine.

Effect of the depth of sediment removal upon the community structure

The depth of sediment removal (20, 30 and 40 cm) in the newly-created pond habitat had no major influence upon the chironomid abundances (Fig. 5). The chironomid density at these three depths was not significantly different (Mann–Whitney U-test, $p > 0.05$) during the three years.

Discussion

Twelve species collected during the course of this study had never been recorded in Switzerland previously (see Table 3) (Heiri, 1996; Lods-Crozet, 1998; Otto and Schiegg, 1999). This quite large number highlights the lack of systematic studies of this family in Switzerland. Taxonomical uncertainty remained about a species close to *Metriocnemus brusti* because of the low number of collected specimens (Saether, pers. comm.). Most of the 78 taxa were already collected in adjacent countries, *e.g.* France (Serra-Tosio and Laville, 1991; Laville and Serra-Tosio, 1996), Austria (Janecek *et al.*, 2002) and Germany (Samietz, 1996), but this work represents a first reference list of chironomid species in a fringing wetland for Switzerland.

Almost all the species collected in the reed-belt and the different ponds were relatively eurytopic. However, distinct preferences for shallow stagnant or slow-flowing habitats (*e.g.* pools, ponds, shallow lakes, littoral zones of lakes) are indicated for them in the literature (Reiss, 1968; Moller Pillot and Buskens, 1990). *Tanypus punctipennis* is

Table 3: Mean number of adult male Chironomidae (no. individuals per emergence trap), collected per site and per year (1994–1996). * New for the Swiss fauna; see Table 1 for abbreviations of sites.

Site – year	RB94	RB95	RB96	NCP94	NCP95	NCP96	PB96	PB94	PB95	PA95	PA96	PA94
Chaetocladius (Kieffer)	0.03											
Dicrotendipes notatus (Meigen)	0.03											
Limnophyes Eaton	0.03											
Nanocladius Kieffer	0.03											
Paratendipes albimanus gr. (Meigen)	0.03											
Chironomus (*Camptoch.*) *pallidivittatus* (Malloch)*		0.04										
Microtendipes diffinis (Edwards)*		0.81										
Zavrelimyia Fittkau		0.08										
Paratendipes nudisquama gr.*	0.06		0.04									
Psectrotanypus varius (Fabricius)	0.06		0.04									
Neozavrelia Goetghebuer			0.04									
Xenopelopia nigricans (Goetghebuer)	0.43	1.27	2.92	0.07	0.1	0.05	0.1	0.07				
Metriocnemus brusti var. Saether*		0.08	0.08		0.05							
Chironomus cingulatus Meigen		0.04	0.12	0.04		0.05						
Corynoneura carriana Edwards*	0.51			0.32	0.1		0.1					
Microtendipes Kieffer	0.03	0.04		0.14								
Cladotanytarsus mancus (Walker)	0.43	0.23	0.56	0.18	0.25	0.2		0.21	0.2			
Polypedilum (*Tripodura*) *bicrenatum* Kieffer	0.03	0.08	0.28	0.04	0.75	0.15						
Parachironomus Lenz				0.07								
Endochironomus dispar (Meigen)		0.04		0.04	0.45							
Clinotanypus nervosus (Meigen)					0.1							
Cryptochironomus rostratus Kieffer					0.05							
Demicryptochironomus vulneratus (Zetterstedt)					0.1							
Cryptochironomus albofasciatus (Staeger)				0.04	0.15	0.05						
Tanypus punctipennis Meigen	0.11	0.69	3.32	11.11	5.95	1.8	1.6	0.93	2.1	0.4	0.2	0.43
Cryptochironomus psittacinus (Meigen)						0.05						
Chironomus riparius Meigen		0.04	0.08						0.1			
Cricotopus van der Wulp				0.14	0.05			0.07				
Cricotopus (*Isocladius*) *sylvestris* (Fabricius)	0.03	0.12		1	5.7			0.29	1.6			
Cricotopus (*Cricotopus*) *annulator* Goetghebuer			0.2	1.07	2.8	0.9	0.4		1.3			
Cricotopus (*Isocladius*) *intersectus* (Staeger)		0.08	0.32	0.29	4.25	0.8	0.2		1.5	0.2		
Tanytarsus mendax Kieffer		0.08		0.75	3.6	4.75	0.2			0.2	1	
Chironomus dorsalis authors not Meigen		0.04	0.08				0.3					
Microchironomus tener (Kieffer)					0.2	0.2	0.3					
Acricotopus lucens (Zetterstedt)				0.04	0.35		0.1	0.07	0.1			
Corynoneura scutellata Winnertz			0.04					0.07				
Cladopelma viridula (Linnaeus)				0.71	0.15	0.05	0.1	0.07	0.1		0.2	
Cryptochironomus Kieffer				0.07				0.07				

Table 3: Continued.

Site – year	RB94	RB95	RB96	NCP94	NCP95	NCP96	PB96	PB94	PB95	PA95	PA96	PA94
Polypedilum (*Pentapedilum*) *uncinatum* (Goetgh.)*					0.15				0.1			
Paratanytarsus tenellulus (Goetghebuer)*		0.04					0.1	0.07				
Corynoneura edwardsi Brundin			0.04	0.07	0.4	0.95	1.3	0.07	0.4	0.2		
Procladius choreus gr. (Meigen)	0.06	0.12	0.16	0.11	1.95	0.05		0.29				0.71
Polypedilum (*Tripodura*) *scalaenum* (Schrank)					0.05		0.2					
Parachironomus arcuatus (Goetghebuer)				0.21		0.6	1	0.07			0.2	
Glyptotendipes Kieffer	0.03			0.14				0.21	0.2			
Polypedilum (*Pentapedilum*) *sordens* (van der Wulp)*			0.24		0.3	1.1	0.6	0.07	3.1			
Parachironomus biannulatus (Staeger)							0.1					
Tanypus vilipennis (Kieffer)				0.29						0.2		
Paratanytarsus intricatus/tenuis	0.03			0.46		0.05		0.79	0.2	0.2		
Polypedilum (*Polypedilum*) *nubeculosum* (Meigen)	0.09		0.08	0.43	1.05	0.55	0.7			0.6		0.57
Corynoneura celtica Edwards					0.05		0.2		0.2			
Einfeldia longipes (Staeger)						0.15	0.1	0.5	0.1			
Psectrocladius (*Allopsectrocladius*) *obvius* (Walker)				0.04	0.2				0.1	0.2		
Phaenopsectra Kieffer					0.05		0.2	0.07	1			
Parachironomus monochromus (van der Wulp)								0.07				
Glyptotendipes gripekoveni (Kieffer)					0.05		1.1		2.4	0.2		
Microtendipes pedellus (De Geer)	0.03	0.12	0.52		0.15	0.2	0.3		1.5		1.4	
Paramerina cingulata (Walker)*				0.07	0.3	0.05	0.3	0.43	1.4	0.4	0.2	
Monopelopia tenuicalcar (Kieffer)*	0.03	0.19		0.14	0.1			1.21	0.4	0.4		0.29
Dicrotendipes tritomus (Kieffer)					0.05			0.64	0.4	0.2		
Paratanytarsus intricatus (Goetghebuer)*		0.04	0.04	0.61	0.1	0.1	0.3	1.29	3.2	1.2	3	
Endochironomus tendens (Fabricius)			0.12	0.11	0.4	0.3	3.7		0.8	5.6	0.4	0.29
Chironomus s. str. Meigen	0.03			0.04			0.1					0.14
Chironomus plumosus (Linnaeus)				0.25	0.15	0.15	0.1	0.21	1.9	5.8		
Corynoneura lobata Edwards									0.1	0.2		
Ablabesmyia phatta (Egger)					0.3			0.36	1.2	1.4	1.2	0.57
Cladopelma edwardsi (Kruseman)				0.11	0.05	0.4	0.2	0.14		6.4		
Dicrotendipes nervosus (Staeger)			0.04		0.05			1.14	1.1	1		2.14
Dicrotendipes modestus (Say)									0.1	1.4		
Dicrotendipes lobiger (Kieffer)				0.04						0.8	0.2	
Polypedilum (*Polypedilum*) *cultellatum* Goetgh.										0.2		
Tanytarsus excavatus Edwards		0.12			0.8	0.05	0.2		2.5	14	20.8	4.29
Tanytarsus nemorosus Edwards	0.03									0.8	0.2	0.14
Parachironomus parilis (Walker)				0.07	0.2		0.1	0.21	2.1	0.2		4.57
Polypedilum (*Pentapedilum*) *tritum* (Walker)								0.07			0.2	0.29
Polypedilum (*Pentapedilum*) Kieffer					0.05							2.86
Tanytarsus van der Wulp												8.43
Total number of taxa	22	22	22	35	42	26	30	29	31	25	13	14

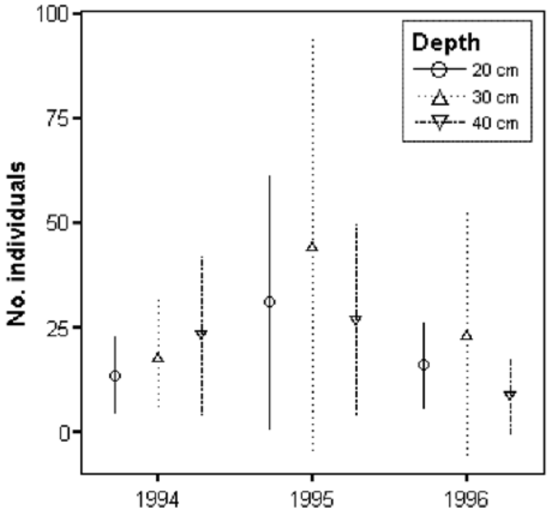

Fig. 5. Mean chironomid abundance (individuals/0.2 m^2 ± SD) in the newly-created pond at three depths during three years.

characteristic of stagnant waters, eutrophic and hypertrophic habitats (Caspers and Heimbach, 1995*)*. *Cricotopus sylvestris* is a ubiquitous species but often associated with plants (Mackey, 1979; Caspers and Heimbach, 1995). *Xenopelopia nigricans* was found as larvae in small stagnant waters and eutrophic lake littoral (Ferrarese, 1983; Wiederholm, 1989) but in this study the species colonised almost exclusively the reed-belt habitat (see Table 3). Several other taxa were found only in the reed-belt habitat which could be considered as a semi-terrestrial habitat (*Psectrotanypus varius, Zavrelimyia, Chaetocladius, Limnophyes, Metriocnemus*). Chironomid richness and abundance were lower in the reed-belt habitat, as observed by Wrubleski (1987) in marshes. On the other hand, *Chironomus pallidivittatus* had its highest abundance in the open water habitat (ponds), as in the open water areas of a North Dakota wetland studied by King and Wrubleski (1998).

The diversity of chironomids in this lake-fringe wetland (78 taxa) is not exceptional when compared to wetlands from other regions, but it is larger than usually reported for temporary ponds (Bazzanti *et al*., 1997; Leeper and Taylor, 1998). Wrubleski and Rosenberg (1990) collected 84 taxa of chironomids using emergence traps in a semi-isolated wetland along the shore of Lake Manitoba, Canada. The same number of species was collected in peatlands in north-western Ontario by Rosenberg *et al.* (1988). As in our study, Stagliano *et al.* (1998) found 31 genera of chironomids emerging from a small wetland in the southeastern USA and Botts (1997) 34 genera of chironomid larvae in a Lake Erie coastal wetland.

Adult chironomids are good dispersers (Batzer and Wissinger, 1996) and are often the earliest colonists to arrive in newly created wetlands (Barnes, 1983; Layton and Voshell, 1991). Their life-history strategies allow them to rapidly colonize the early phases of newly inundated wetlands. However, the chironomid pool, from the adjacent lake littoral zone (about 40 m) did not seem to play a very active role as reservoir for the colonisation of the wetland and the connected ponds. Only ubiquitous taxa like *Procladius choreus* gr., *Cricotopus, Cryptochironomus, Demicryptochironomus vulneratus, Polypedilum nubeculosum* gr. and *Cladotanytarsus mancus* gr. were common to both the adjacent lake and the studied ponds (Lods-Crozet, unpublished data).

Cricotopus species, well represented in the studied wetland, are recognized as pioneer in new habitats (Oliver, 1971; Titmus, 1979). The fact that the dominant species are bi- or multivoltine (Reiss, 1968; Lindegaard and Brodersen, 2000), emerging from early spring to mid-autumn, enhances the colonisation process by aerial dispersal of ovipositing adults. Feeding groups like collector-filterer larvae living on detritus and algae prevailed during the phase of colonisation (Maher and Carpenter, 1984). The earliest communities in the recent pond reflected this trend (*Tanypus punctipennis, Cricotopus annulator, Chironomus pallidivittatus, Cladopelma edwardsi, Paratanytarsus, Tanytarsus mendax*). Furthermore, the desiccation resistance of both eggs and larvae (Delettre, 1990; Tokeshi, 1995) contributes to the rapid settlement of chironomid assemblages.

Several studies reported higher numbers of invertebrates in newly flooded marsh habitats than in adjacent older wetlands (Whitman, 1974; Danell and Sjöberg, 1982). In this study, the same trend was observed with a significantly lower abundance in the reed-belt habitat than in the newly created pond. However, the recently created pond had comparable densities with the older pond B during the three years of colonisation. In term of diversity, it seems that three years of colonisation was not sufficient to provide a the level of chironomid diversity that can be expected from the adjacent existing ponds, or even from the reed-belt that was subjected to water level fluctuations and seasonal drying out. The time effect was also not demonstrated, since the rarefied richness showed no significant differences between years. A similar result was established by Ruhi *et al.* (2009) in recently created ponds in Spain.

The experimental scraping of the reed-belt soil at three different depths had no major influence on the chironomid assemblages, but the zone scraped down to a depth of 20 cm was important as a breeding site for two rare species, the dragonfly *Aeschna affinis* and the mayfly *Caenis lactea* (Gander, 2003). It is also relevant to note that three bird species associated with the reed-belt and considered as key species in this wetland reserve, colonised the new created pond. This demonstrated that the sediment removal experiment has helped to restore the integrity and function of the reed-belt.

Because European wetlands are threatened and constantly reducing in extent, management strategies tend to promote their conservation and restoration. Whitman (1974) suggested that optimal conditions for invertebrate production in newly created wetlands occur during the first 1.5 to 4 years and he recommended drawdown between years 5 to 7 to improve food and cover for waterfowl. In the present study, the creation of a new shallow pond in the reed-belt of a lacustrine wetland was beneficial for chironomid abundance, but our results must be considered as preliminary because they are based only upon a three year period, which is not long enough to confirm chironomid diversification associated with the creation of the new open water habitats. Nevertheless, this management option could be considered as relevant because it slows down terrestrialization processes, compensates for wetland habitat destruction and the subsequent loss of species, enhances chironomid abundance and consequently their availability as food for dabbling ducks and shorebirds.

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