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Restoration of Skjern River and its valley—Short-term effects on river habitats, macrophytes and macroinvertebrates

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ABSTRACT

The lower 19 km of the Skjern River was restored and transformed into a 26 km long meandering river. Three survey reaches and one control reach upstream of the restoration area were surveyed to assess the short-term effects of the restoration on river habitats, macrophytes and macroinvertebrates. The reaches were surveyed before the restoration in 2000 and again after the restoration in 2003. Morphological adjustments were evident in the re-meandered river and the habitat structure (depth, current velocity and substratum) became more diverse. The macrophyte coverage was 34% before the restoration. Restoration included removal of dense near bank vegetation stands of *Glyceria maxima*, and in 2003 re-colonization of the restored reaches had resulted in 24% macrophyte coverage. Species composition and growth patterns changed significantly in the edge habitat and the dominant macrophyte *G. maxima* was replaced by *Elodea canadensis* and *Sparganium* sp. Macroinvertebrates rapidly colonized the restored reaches and increased the community diversity. Only one taxon, Heptageniidae, significantly increased in abundance after the restoration and a more even distribution of taxa developed on the restored reaches. Biological communities will continue to develop over the coming years as the river becomes more physically stable. Hence the macroinvertebrate and macrophyte communities will adjust and colonization from upstream sources and other systems will probably increase biodiversity.

1. Introduction

Stream and rivers constitute a dense network of flowing water with a large interface to the surrounding landscape. For this reason they are highly susceptible to various anthropogenic pressures and this has significantly affected biodiversity and caused the extinction of some species (Sand-Jensen, 2001). Natural morphological characteristics and the biota of streams and their riparian areas have been strongly altered and in many cases lost due to human manipulation of the landscape (e.g. Phillips, 1995; Sparks, 1995). One way of counteracting the morphological degradation of streams and rivers is to restore the lost physical features (e.g. Mitsch and Jørgensen, 2003; Ormerod, 2003; Pedersen et al., 2006a). This has been widely applied in the US and Western Europe on different scales and with the use of a multitude of approaches (Holmes, 1998; Hansen, 2000; Bernhardt et al., 2005).

Restoration can be defined as "a complete structural and functional return to a pre-disturbance state", *sensu* Cairns (1991). Many river restoration projects in lowland areas aim at re-creating morphological features and thus increase habitat diversity. Because lowland streams are usually located in

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areas used for agricultural production or urban settlements a complete restoration is rarely feasible.

The Skjern River restoration project included re-meandering of the river to approximately the course the river had before the major channelization in the 1960s. The idea behind the project was to recreate the old meandering channel and the natural dynamic hydrological and physical processes within the river. By restoring the processes and some morphological features the biotic communities were left to colonize the new river (Pedersen et al., 2006b). Restoration will be used throughout this paper as used in Denmark although it is recognized that in the strictest sense the restoration of the lower part of the Skjern river is river rehabilitation and not a complete restoration to pre-disturbance conditions.

Re-meandering of stream channels has been widely undertaken as a restoration measure in lowland areas. This type of project results in an immediate rehabilitation of some aspects of natural stream channel morphology as new meanders are actively created and coarse substrata added (e.g. Friberg et al., 1994; Biggs et al., 1998). Few studies have documented the ecological effects of river restoration (Palmer et al., 2005). However, the few existing effect studies have shown some distinctive short-term patterns: the biota generally recovers rapidly after the restoration (1–2 years) due to high resilience in the biotic communities (Friberg et al., 1994; Kronvang et al., 1997; Biggs et al., 1998). Recovery appears to be related to stream size (catchment area) and hence colonization opportunities from upstream refuge areas (Hansen et al., 1999). The results also indicate that macrophytes could play a key role in the recovery (Friberg et al., 1998). Macrophyte recovery generally appears to be slower than that for macroinvertebrates, but Pedersen et al. (2006a) found that macrophyte communities in restored streams were similar to unimpacted reference streams if restoration had taken place more than 3 years prior to sampling.

The water quality of most Danish streams has improved markedly during the recent two decades (e.g. Kronvang et al., 1996, 1997). However, this improvement has not been matched by an overall improvement in stream ecosystem quality. This lack of improvement is probably due to the poor physical conditions of many lowland streams (e.g. Iversen et al., 1993). The primary reason for the deterioration of streams and riparian areas in Denmark has been the change in land use from forested land to intensively cultivated land, 63% of Denmark is presently being arable farmland and only 12% forest. Cultivation of farmland has resulted in extensive straightening and culverting of streams and drainage of riparian wetlands. Currently more than 90% of the 35,000 km of natural Danish streams have been physically modified (e.g. Iversen et al., 1993). One way of improving the physical stream quality is by active re-meandering and rehabilitation of substratum and habitat conditions in the rivers and streams. Numerous Danish restoration projects have focused on re-meandering since the late 1980s (Iversen et al., 1993; Hansen, 1996, 2000; Kronvang et al., 1997).

The Skjern River is the largest river in Denmark with an average annual discharge of 35 $m^3 s^{-1}$. Throughout the last 200 years, the lower river valley has been used for agricultural production. Increased demand for fertile agricultural soils in the late 1960s led to the draining of the wetlands and channelization of the river. During the 1970s the negative effects of the channelization and draining scheme set in and plans for restoring the river and the valleys were initiated in 1987. In 2000–2002 the lower part of Skjern River (19 km) was restored into a 26 km meandering river with natural discharge and sediment dynamics.

The overall aim of the present study is to investigate the short-term effects of the Skjern river restoration on river habitats, macrophytes and macroinvertebrates. We addressed this aim by sampling the Skjern River prior to the restoration in 2000 and again after completion of the restoration works in 2003. We hypothesise that: (1) the restoration has an instant effect on the habitat diversity. (2) The altered habitats will result in higher macroinvertebrate diversity and abundance. (3) We also hypothesise that the restoration will alter abundances and composition of the macrophyte community. (4) The re-colonization potential in a large river, like the Skjern River is high, and therefore we expect the macroinvertebrate community to recover rapidly. (5) Furthermore, we do not expect the diversity of the biotic communities to increase because dispersal of new species between catchments is a slow process, especially in larger rivers which are usually located far from each other.

2. Materials and methods

2.1. Study sites

Short-term effects of the restoration on river habitats, macrophytes and macroinvertebrates were examined in three 300 m-long survey reaches along the lower part of Skjern River (R1, R2, R3). Prior to restoration a 300 m control reach (C4) upstream from the restoration area was selected for comparison (Fig. 1). All four sites were sampled once before and after the restoration in September 2000 and 2003, respectively. Three supplementary 100 m-long control reaches (C5, C6, C6) located in Omme river and in Skjern River upstream of C4 were used to evaluate the re-colonization of macroinvertebrates after the restoration (Fig. 1).

2.2. Changes to the overall morphology

Restoration of the Skjern River comprising 46 new meanders, cross sections and riffles resulted in significant changes to the overall morphology of the cross sections (Fig. 2). The banks of the channelized river were dominated by a dense growth of *Glyceria* species. This vegetation was removed as part of the construction of the new cross sections and the slope of new banks was generally lower than in the channelized river. After the completion of the restoration work the new river has experienced marked morphological adjustments which have caused erosion and deposition in the cross sections (Fig. 2). The average cross sectional area was decreased by 20% at reach R3 and 30% at reach R2 (Table 1). Gravel was introduced to the newly created riffles as part of the restoration scheme. This increased the gravel coverage on the river bed by 5%.

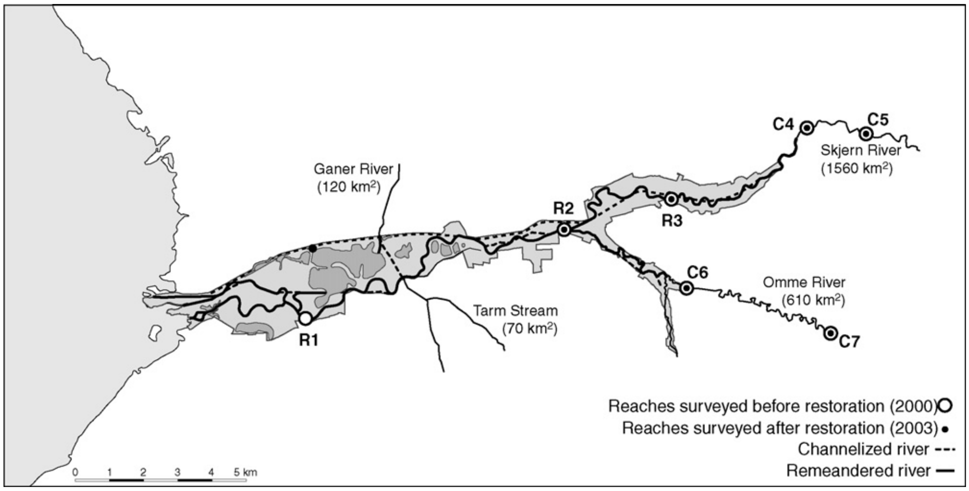

Fig. 1 – Map of the restoration area showing the position of the three restored reaches (R1, R2 and R3) and the control reach just upstream of the restoration area (C4). The straightened and the re-meandered reaches are also shown. The area affected by the restoration of rivers and raised groundwater level is shaded.

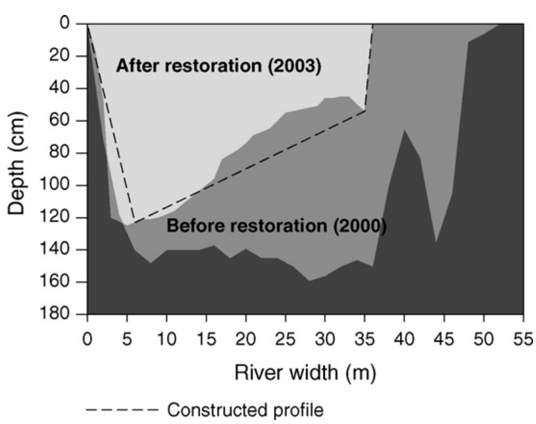

Fig. 2 – Example of the changes to the cross-sectional profiles in Skjern River. The cross sectional area has generally decreased by approximately 30%. The morphology of the profiles has changed from the constructed rectangular shape to a more natural physical appearance.

2.3. *Morphology and in-stream habitats*

Six transects were placed at equal intervals along the 300 m reach. Each transect was divided into 1 m × 1 m quadrats across the entire width. The detailed cross section survey enabled comparison of cross sections before and after the restoration. All measurements were performed from a boat anchored to a wire with 1 m markings and spanning the width of the river. This ensured that the boat was kept in place, and that all measurements were made at exact positions across the river. The number of quadrats varied between 405 and 572 among the reaches, depending on the river width. A glass-bottom hydroscope was used to facilitate underwater observations of substratum characteristics.

The depth was measured to the nearest cm at the centre of each quadrat and the current velocity was measured 10 cm above the stream bed using an electromagnetic current meter (Nautilus, OTT Instruments, Germany). The dominant substratum on the river bed in each quadrat was visually assessed using an approximation to the Wentworth-scale (Wentworth, 1922): Stone (>64 mm), gravel (2–64 mm), sand (0.5–2 mm), silt (<0.5 mm), mud (<0.1 mm, black colour), peat and hard clay.

Table 1 – General characteristics (medians) of the geomorphology of the Skjern River in 2000 before the restoration and in 2003 after the restoration

	Control reach C4		Restored reach R3		Restored reach R2	
	2000	2003	2000	2003	2000	2003
Median width (m)	20	20	32	26*	49	34*
Median depth (cm)	149	145	128	98*	132	83*
Median current velocity (cm s^{-1})	33	32	30	41*	29	38*
Dominant substratum	Sand	Sand	Sand	Sand	Sand	Sand

The overall morphological changes could not be evaluated at site R1, therefore data are only available for control reach C4 outside the project area and the two restored reaches R2 and R3.

* Significant differences (Mann–Whitney *U*-test, $p < 0.05$) between the median values in 2000 and 2003.

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The presence of coarse particulate organic matter (CPOM) or fine particulate organic matter (FPOM) in each quadrat was also registered. Mean depth and mean current velocity of a reach was calculated from all measurements on the reach. Mean stream width was calculated from measurements of the wetted width in all five transects.

2.4. Macrophytes—in-stream vegetation

The 1 m × 1 m quadrats used for the morphological survey was also used for the macrophytes. A cover score was allocated to each macrophyte species present in the quadrat using the following scale: 1 = 1–20%, 3 = 20–40%, 3 = 40–60% 4 = 60–80% and 5 = 80–100%. Macrophyte species were identified to the species level whenever possible using standard taxonomic keys (Hansen, 1981; Moeslund et al., 1990). If identification to species level could not be achieved due to absence of seasonal diagnostic features, then the record was only performed to genus level. To achieve total coverage values for each species present, at each site, the sum of quadrat coverage values was divided by the total number of quadrats investigated multiplied with the maximum score (in this case 5). Hereafter, these values were multiplied with 100 to reach percentage coverage for the species. Total macrophyte coverage was calculated as the sum of total coverage values of present species.

2.5. Macroinvertebrates

Three samples were collected on all substrata covering more than 5% of the river bed at each reach. Before the restoration sand and macrophytes were sampled. Mud was always associated with macrophytes and is thus included in the macrophyte samples. After the restoration gravel and macrophytes were sampled. Samples were collected using kick sampling or diving if the depth exceeded 1 m. Sampling was undertaken using either a Surber-sampler or a hand net, both fitted with a frame with an area of 500 cm^2 to ensure equal sample areas irrespective of the method used. Macroinvertebrates were retained in 0.5 micron nylon net. All samples were kept separately and preserved in 96% ethanol. The samples were sorted, identified and counted in the laboratory using standard sorting keys. Macroinvertebrates were identified to either species or genus level. Diptera were identified to family level, except for chironomids which were identified to sub-family level.

2.6. Study design

The habitats and the biological communities were surveyed once before and once after the restoration. Site (C4) was used as the control site for all three survey sites (R1, R2, R3) to check if any hydrological and morphological changes occurred in the river upstream of the restoration area. Traditional BA or BACI ANOVA analysis was not possible because only replicate samples from each reach and no time series data were available (Stewart-Oaten et al., 1986; Smith, 2002). As only the physical conditions were altered as a consequence of the restoration, and the design included independent checking for physical difference at the control site (C4), standard statistical comparative tests were applied to analyse short-term changes to biotic communities after the restoration of Skjern River.

2.7. Data analysis

The physical habitat variables between control reaches and the survey reaches R2 and R3 were compared before and after the restoration. Physical characteristics were not normally distributed and therefore medians were compared by means of Mann–Whitney pair wise *U*-test on depth, current velocity and river width (Conover, 1980).

The distribution of depths, current velocities and substrata was tested for differences between the channelized and the re-meandered condition using Chi-square tests. The distribution of macrophyte coverage among different depths and the coverage of macrophyte species before and after the restoration were tested using the χ^2-test (Snedecor and Cochran, 1989).

In each sample, macroinvertebrate community structure and diversity were expressed in several ways. Total abundance and abundance of Ephemeroptera, Plecoptera and Trichoptera (EPT) and Shannon diversity (H′) along with taxonomic richness and EPT taxa richness were calculated for each sample (Washington, 1984). The macroinvertebrate community structure before and after the restoration was compared in two ways: (1) by treating all samples from each year separately and (2) by aggregating samples to the reach level according to substratum distributions and then comparing aggregated communities between 2000 and 2003. For the first comparison the community parameters (abundance, species richness, Shannon diversity, EPT taxa riches and EPT abundance) were calculated for each sample and the mean values for 2000 and 2003 were then compared using standard t-tests on log-transformed data. The second comparison was performed by calculating substratum-weighted means of the community variables for each reach and then comparing the 2 years using standard *t*-tests on log-transformed data. Linear regression analysis on the log-transformed rank-abundance relationships was used to analyse changes to the overall species distribution (Southwood and Henderson, 2000). Biotic effects of the changes to in-stream habitats were analysed by comparing the abundances of six selected taxa using *t*-tests on log-transformed data. The selected taxa represented three taxa occurring in mud substratum and in association with macrophytes in Danish streams (Tanytarsini indet, Orthocladiinae indet and Simulidae indet) and three taxa associated with gravel and stones (*Baetis* spp., *Elmis aenea* and Heptageniidae indet). Taxonomic richness was also estimated as the overall species richness (S_{max}) calculated from all samples on the three sites R1, R2 and R3 in 2000 and 2003 using the first order Jack-knife estimate based on re-sampling of the species lists (Palmer, 1990). Confidence intervals for S_{max} were calculated from Smith and Van Belle (1984).

The effect of the restoration on the macroinvertebrate communities on the restored sites was also analysed by means of Bray–Curtis similarity between samples. Taxonomic similarity was calculated from all macroinvertebrate samples from both years. Four groups of samples were compared using the ANOSIM procedure in the Primer software package (Primer E-Ltd, 2001). The four groups were: (1) control 2000, (2) control 2003, (3) samples from the channelized river in 2000 and (4) samples from the restored reaches in 2003. The variation in similarities within the groups was also analysed and compared, using a standard *t*-test (Snedecor and Cochran, 1989).

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3. Results

3.1. *River morphology and habitats*

The discharge and channel slope were not affected by the restoration. The decrease in the cross sectional area therefore directly influences the current velocity, which increased by about 30%, from 29 and 30 to 38 and 41 cm s^{-1} in reach R2 and R3, respectively. Similar changes to the in-stream-physical environment did not occur on the control reach C4 in the period (Table 1).

The current velocity distribution changed significantly after the restoration (Fig. 3; Kolmogorov–Smirnoff test, $p < 0.001$). Overall depth distributions on the reaches affected by restoration increased significantly in heterogeneity, and the dominant depth interval decreased from 100–160 to 40–140 cm (Fig. 3; Kolmogorov–Smirnoff test, $p < 0.001$). The dominant substratum in the Skjern river was sand which covered 58 and 68%, respectively, before and after the restoration. The introduction of gravel to the restored riffles was reflected in an increase in gravel coverage of 5% from 4 to 9%. The vegetation and associated mud substratum was located along the river edge and covered 12% in 2000. In 2003 this had almost disappeared as a consequence of the restoration (Fig. 3).

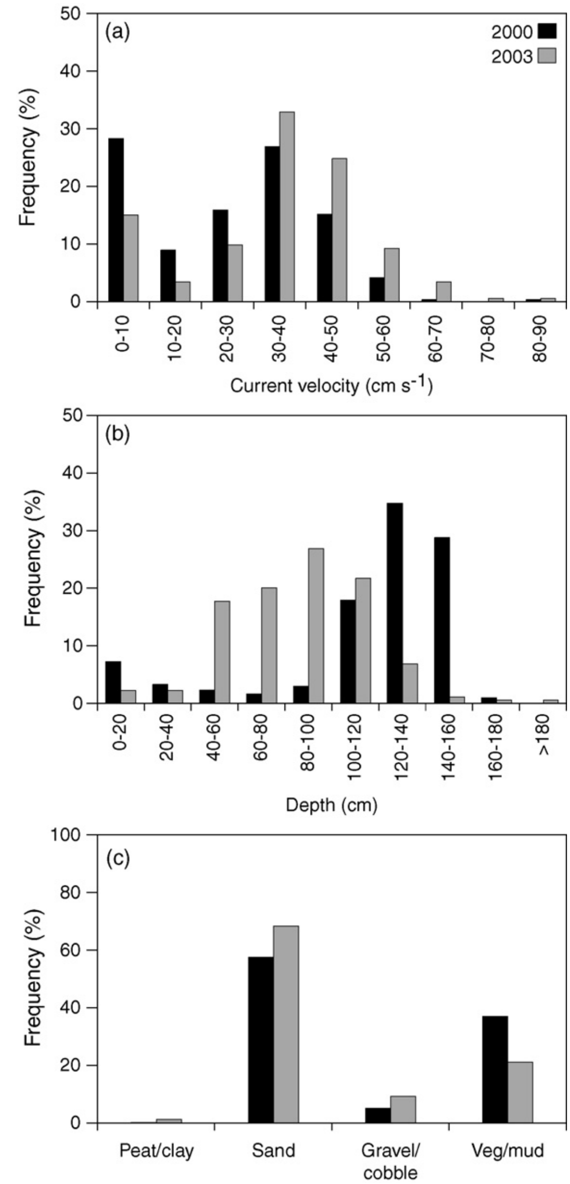

Fig. 3 – Changes to the morphological parameters in the Skjern river following re-meandering and restoration of riffles as measured in 18 transects on three reaches before (2000) and after the restoration (2003). (a) Distribution of current velocities, (b) distribution of depths and (c) distribution of different substrata on the river bed—including the channel vegetation.

3.2. *Macrophytes—in-stream vegetation*

Average macrophyte coverage was significantly lower in 2003 (24%) after the restoration than before the restoration in 2000 (34%). No significant changes were detected in the macrophyte coverage on the control reach between 2000 and 2003 (Table 2). The lower macrophyte coverage after the restoration was primarily attributable to a decrease in cover in the shallow edge habitat where *Glyceria maxima* stands were removed, and to a lesser extent by a lower coverage in the central part of the river (Figs. 4 and 5). Current velocities of 0–10 and 30–40 cm s^{-1} dominated within the *G. maxima* stands along the edge and in the middle of the channel constituted, respectively. After restoration the velocity distribution shifted towards higher current velocities, mainly around 30–60 cm s^{-1}.

The average number of species present in the 1 m^2 quadrats was identical in 2000 and 2003. However, if the edge habitat (first 2 m from the bank) was removed from the analyses, a significant decrease in the average number of species present was observed (Table 2). Total species richness increased from 28 in 2000 to 40 in 2003. *G. maxima* and *Glyceria fluitans* and *Phragmites* sp. dominated the macrophyte community before the restoration. The re-meandered channel and the edge habitat were dominated by colonizers such as *Elodea canadensis* and *Sparganium emersum* dominate (Table 3 and Fig. 5). *G. maxima* was more evenly distributed in the quadrats in 2003 than in 2000 where extended growth in the edge habitat shifted the distribution towards dominance of high coverage (Fig. 5). In contrast, the coverage of both *E. canadensis* and *S. emersum* generally increased after the restoration and both species were found at more varied depths than before the restoration.

G. maxima was primarily found on sandy substratum in 2003, whereas it was primarily associated with mud substratum in 2000. *E. canadensis* and *S. emersum* were associated with a mixture of sandy and mud substratum in 2000. In 2003, however, *E. canadensis* was primarily found in quadrats with sand whereas *S. emersum* was still found on a mixture of sand and mud. All three species occupied identical depths in 2003 whereas they varied in depth preferences in 2000.

3.3. *Macroinvertebrates*

Two years after the restoration of the Skjern River, the macroinvertebrate mean sample or habitat weighted mean diversity and abundance had reached pre-restoration levels

Table 2 – Mean plant coverage and number of macrophyte species in the surveyed 1 m^2 quadrats

	Control reach		Restored reaches (R1, R2, R3)	
	2000	2003	2000	2003
River				
No. of. species	0.9 (0–8)	0.9 (0–9)	0.7 (0–7)	0.6 (0–12)
Coverage (%)	32 (0–300)	36 (0–260)	34 (0–220)	24 (0–320)[a]
River excluding edge zone				
No. of. species	0.4 (0–5)	0.4 (0–5)	0.6 (0–7)	0.2 (0–4)[a]
Coverage (%)	15 (0–180)	17 (0–180)	25 (0–200)	10 (0–220)[a]

Ranges of means are given in parenthesis. The number of observations varied between 405 and 472, depending on actual width on sites.
[a] Significant differences in coverage or number of species between the surveys in 2000 and 2003.

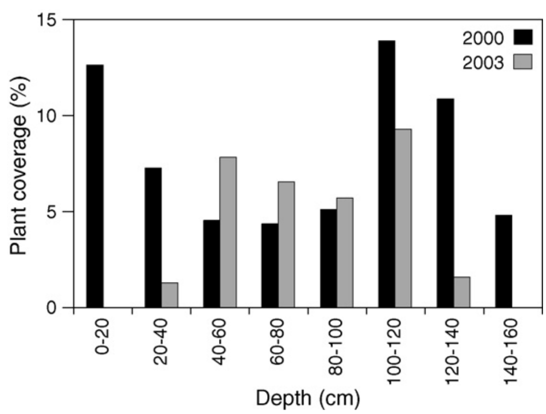

Fig. 4 – Plant coverage at different depth classes before (2000) and after the restoration (2003). Data are aggregated from site R1, R2 and R3. Coverage is in percentage for each depth class and therefore not additive due to differences in depth-class distributions in the river.

and no significant differences were found when data from 2000 and 2003 were compared (t-tests, $p > 0.05$; Table 4). The range in community variables was generally lower after the restoration compared to the situation before the restoration and the standard deviations were significantly higher in 2000 compared to 2003 (F-test, $p < 0.01$). A total of 97 macroinvertebrate taxa were found during the 2-year survey in Skjern River. Overall species richness was 63 before the restoration in 2000 and 76 after the restoration in 2003. Mean species richness per sample was also identical in 2000 and 2003, but total Jack-knife estimated species richness was significantly higher (95.9) after the restoration than before the restoration (86.4) (Table 4).

The macroinvertebrate community was more consistent in 2003 compared to 2000, indicating a more homogeneous distribution of taxa and individuals between reaches and samples (Table 5 and Fig. 6). Before the restoration, Simuliidae indet. was the dominant taxon, approximately two times more abundant than Orthocladiinae indet. In 2003 the three most

Table 3 – Frequencies of occurrence of the ten most abundant plant species in Skjern River in 2000 and 2003

Species	2000	Species	2003
Glyceria maxima	30.1	*Elodea canadensis*	18.8
Glyceria fluitans	12.0	*Sparganium emersum*	15.4
Phragmites sp.	11.2	*Batrachium* sp.	8.2
Scirpus lacustris	9.2	*Callitriche* sp.	5.0
Elodea canadensis	5.1	*Glyceria maxima*	4.4
Sparganium erectum	4.9	*Sagittaria sagittifolia*	3.3
Sparganium emersum	4.1	*Potamogeton natans*	2.8
Potamogeton natans	3.6	*Phalaris arundinacea*	1.9
Phalaris arundinacea	2.2	*Glyceria fluitans*	1.7
Potamogeton sp.	2.0	*Lemna minor*	1.6

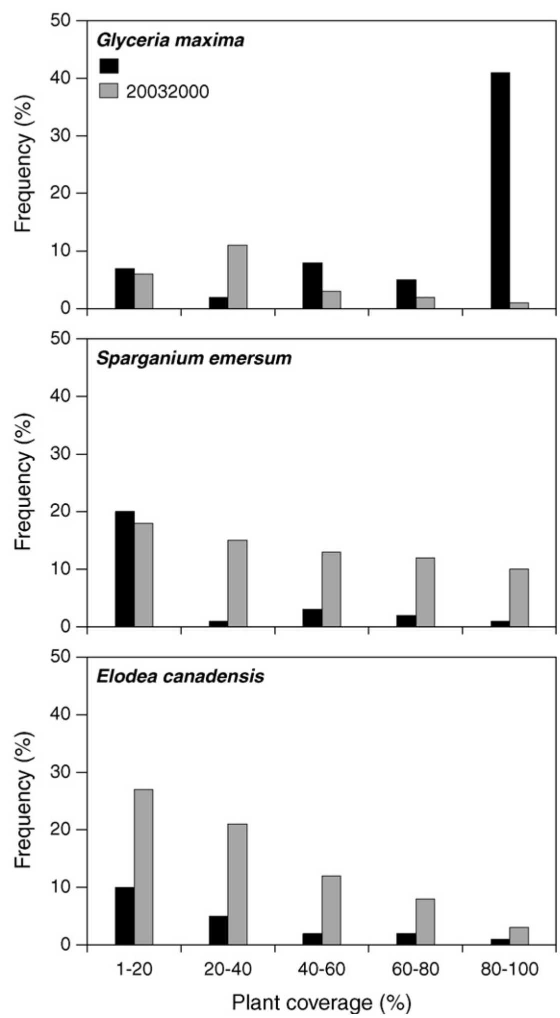

Fig. 5 – Coverage of three frequently occurring plant species in Skjern River before and after the restoration.

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Table 4 – Macroinvertebrate communitry variables before and after restoration of the Skjern River

Samples	Before restoration (2000) ($n = 16$)		After restoration (2003) ($n = 20$)		p-Value
	Mean	Range	Mean	Range	
Species richness	14.7	1–40	19.7	1–33	0.12
Abundance	761	1–3505	335	1–851	0.37
Shannon diversity (H′)	0.43	0–0.92	0.62	0–0.87	0.05
EPT taxa	6.8	0–22	10.4	0–20	0.10
EPT abundance	149	0–1060	192	0–673	0.09
Habitat-weighted	**Before restoration (2000) ($n = 3$)**		**After restoration (2003) ($n = 3$)**		**p-Value**
	Mean	Range	Mean	Range	
Species richness	8.1	6.0–9.8	8.2	3.4–11.6	0.83
Abundance	364	219–625	118	40–200	0.11
Shannon diversity (H′)	0.28	0.11–0.55	0.36	0.11–0.55	0.70
EPT taxa	3.5	2.5–4.6	4.0	1.0–5.8	0.88
EPT abundance	70	35–99	66	3–147	0.52
Species richness (S_{max})	86.4 [84.0–88.9]		95.9 [94.7–97.2]		

Upper part of the table shows mean values based on individual samples from the 2 years. Lower part of the table shows means of community variables based on habitat weighing. All p-values are from t-test on log-transformed data. Total Species-richness was estimated by first order Jack-knife re-sampling of samples. For comparison 95% confidence limit values are also included.

Table 5 – Frequency of occurrence of the 10 most abundant macroinvertebrate taxa in Skjern River before the restoration in 2000 and after the restoration in 2003

Taxa	2000	Taxa	2003
Simuliidae indet.	45.2	Orthocladinae indet	16.7
Orthocladinae indet.	21.1	*Brachycentrus maculatus*	13.6
Brachycentrus maculatus	14.3	*Heptagenia sulphurea*	11.7
Tanytarsini indet.	5.5	*Gammarus pulex*	6.0
Oligochaeta indet.	1.5	*Elmis aenea*	5.7
Pisidium sp.	1.5	Oligochaeta indet.	4.9
Gammarus pulex	1.3	Simuliidae indet.	3.5
Elmis aenea	1.0	Corixinae indet.	1.8
Taniopteryx nebulosa	0.9	*Baetis rhodani*	1.4
Asellus aquaticus	0.6	Chironomini indet.	1.4

dominant taxa, Orthocladiinae indet., *Brachycentrus maculatus*, and *Heptagenia sulphurea* were almost equally abundant (Table 5). Of the ten most common taxa in 2000 and 2003 approximately 80% are identical, which can be attributed to a slightly different distribution of habitats between the 2 years.

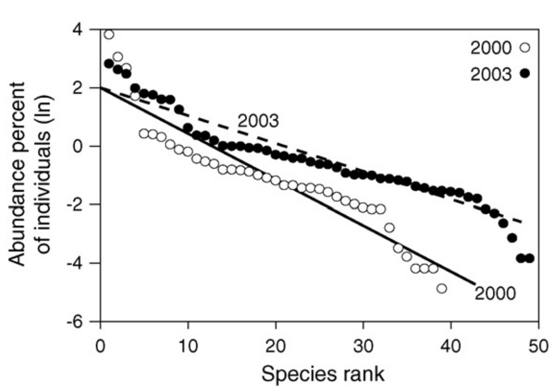

Fig. 6 – Macroinvertebrate rank-abundance relationships on the channelized reaches (2000) and on the restored reaches (2003). The slopes of the linear regression lines are significantly different at the 5%-level.

Simuliidae indet. and Tanytarsini indet. dominated in samples collected in *Glyceria maxima*. These taxa along with Orthocladiinae indet. decreased in abundance after the restoration (Fig. 7). The riffle beetle *Elmis aenea* and the mayflies *Baetis* ssp. (six species including the dominating species *B. rhodani* and *B. niger*) and Heptageniidae (two genera and five species including the dominating species *Kageronia fuscogrisea*, *Heptagenia sulphurea*, and *Heptagenia flava*) increased in numbers after the restoration. However, only the increase in abundance of Heptageniidae was significant (Fig. 7).

The macroinvertebrate community had reached pre-restoration levels on the restored reaches in 2003. The number of EPT taxa, total species richness, ASPT and BMWP on the restored reaches (R1, R2, and R3) were comparable to the upstream control sites (C4, C5, and C6) in 2003 (Table 6). The Danish stream fauna index value was 7, which corresponds to the highest biological quality on all control and restored reaches in 2003.

There was no significant difference (Table 7; pair wise ANOSIM, $p = 0.214$) in macroinvertebrate similarity between samples from the control reach in 2000 and 2003, and also between restored reaches in 2003 and the control reach the same year. In contrast, there was a significant difference ($p < 0.05$) in similarity between the control and pre-restored channelized reaches in 2000 and between the pre-restored channelized reach and the same reaches after the restoration in (pair wise ANOSIM Table 7; Fig. 8).

4. Discussion

4.1. *Short term-effects of restoring the Skjern River*

The diversity in the in-stream habitats increased after the Skjern river restoration. The construction work and increased

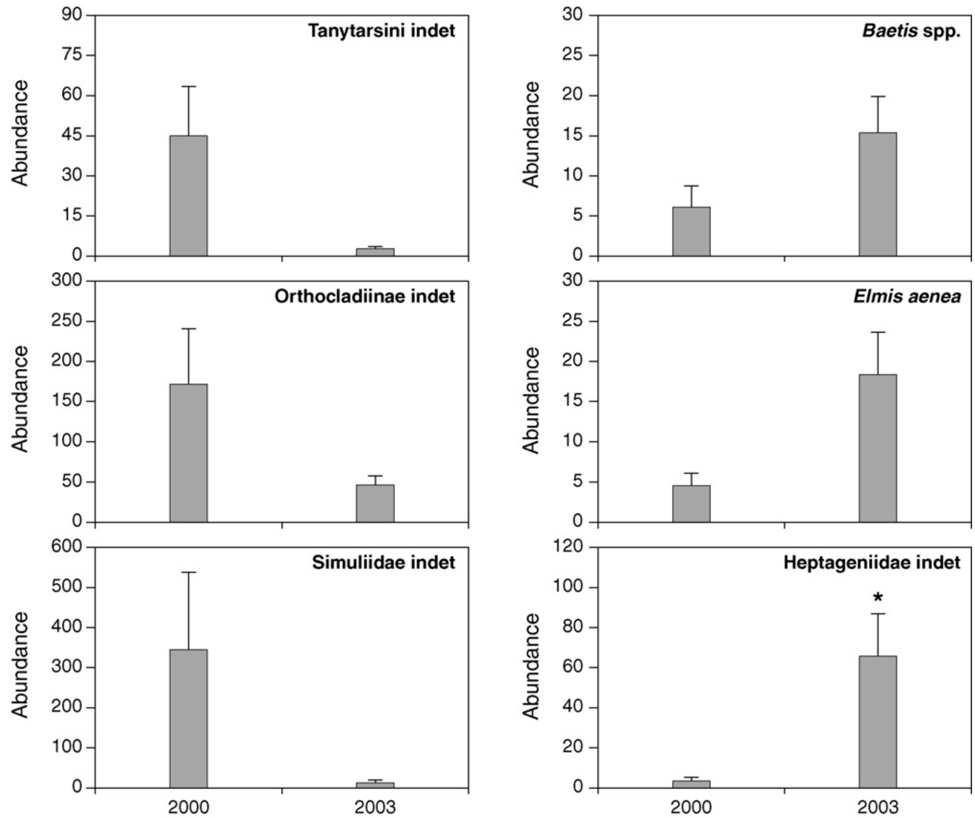

Fig. 7 – Comparisons of mean sample abundance of six individual macroinvertebrate taxa before ($n = 16$) and after the restoration ($n = 20$) of Skjern River. *Significant differences between 2000 and 2003 (t-test, log-transformed data, $p < 0.05$).

habitat diversity influenced the biological components. Plant coverage decreased from 34 to 24% and primarily influenced the coverage of *Glyceria maxima*. The initial short-term colonization of the river resulted in more complex growth patterns. The macroinvertebrate community recovered quickly and the species composition changes significantly in the restored river. Some taxa (e.g. Heptageniidae indet.) responded to the increased coverage of coarse substrata and were thus more abundant after the restoration.

4.2. *Assessing the short-term effects of river restoration projects*

Short-term adjustments to the river morphology and habitats are multiple and in many ways counteractive: the restoration results in an instant increase in the habitat diversity and morphological variability (positive) but at the same time sediment is mobilised and deposited due to morphological adjustments in the newly created channel (negative). These morphological adjustments potentially decrease the possibilities of assessing the short term effects of the restoration. The biotic response to the morphological adjustments can obscure any effects on the biotic communities (e.g. Friberg et al., 1998).

The limited availability of time series of biological data from the channelized river and problems of establishing a sufficient number of controls reaches influenced the monitoring strategy in Skjern river. Applying traditional BACI design to the monitoring programme was not possible and this has to be taken into account when the short-term results are inter-

Table 6 – General characteristics of the macroinvertebrate communities in the lower part of the Skjern River in 2003

Reach	DSFI	Taxa	EPT taxa	BMWP	ASPT
C4	7	45	15	203	6.8
R3	7	45	21	174	6.0
R2	7	48	17	200	6.5
R1	7	48	16	174	6.0
Omme river 5 km upstream R2 (C6)	7	41	17	140	5.6
Omme river 5 km upstream C6 (C7)	7	30	15	149	6.5
Skjern river 2 km upstream R1 (C5)	7	28	13	125	6.6

Several indicators are used to characterise the communities at the four surveyed reaches and additional three upstream reaches. Danish stream fauna index (DSFI) is a water quality metric ranging from 1 (poor) to 7 (high quality) (Skriver et al., 2000). BMWP and ASPT are also water quality indices primarily used in the UK. High BMWP and ASPT scores indicate high quality (e.g. Hawkes, 1997).

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Table 7 – ANOSIM test of similarity between samples from the control reach and the three channelized reaches in 2000 and restored reaches in 2003

Groups	*R*-statistic	*p*-Value	No. of permutations	No. of cases where ≥*R*
r2000, c2000	0.849	0.022[a]	45	1
r2000, r2003	0.307	0.002[a]	999	1
c2000, c2003	0.219	0.214	28	6
r2003, c2003	0.138	0.150	999	149

R-statistic for the global test was 0.349 and $p = 0.001$.
[a] Significant differences in pair-wise ANOSIM tests between the groups.

preted. Conclusions about the actual effects should thus be made with caution (Stewart-Oaten et al., 1986; Smith, 2002).

4.3. *In-stream habitats*

The changes to the river morphology cross generally increased the variability in depth, substratum and current velocities, indicating an initial increase in habitat diversity in the new channel. These effects to the variability in the habitats are similar to results from other restoration projects in lowland rivers (Friberg et al., 1994, 1998; Kronvang et al., 2000). The increased mobilisation and deposition of sediment as a consequence of morphological adjustments in the channel is likely to continuously change the morphology of the cross sections and thus the distribution of habitats (Carling, 1988; Kronvang et al., 1998a,b; Pedersen et al., 2006a). These adjustments are likely to carry on for several years until the river has reached a new state of equilibrium with runoff regime and the surrounding landscape (Frissell et al., 1986; Carling, 1988; Knighton, 1998). By recreating a more natural overall morphology and restoring the physical processes in the river, the potential for development of higher habitat diversity on both the small scale (patch) and on the large scale (reach) has been established (Frissell et al., 1986).

4.4. *Macrophytes*

Macrophytes colonizing the edge habitat after the restoration were found in more complex growth patterns (more plants in each survey quadrat) compared to the channelized river.

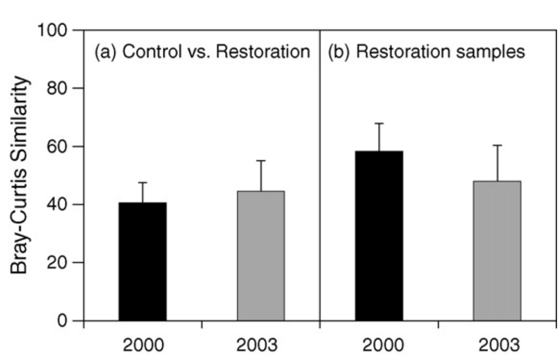

Fig. 8 – (a) Bray Curtis similarity among samples on the control reach and the channelized and restored reaches. (b) Bray Curtis similarity among samples on the channlized reaches in 2000 and among the samples on the restored reaches in 2003.

This finding is similar to the results reported by Pedersen et al. (2006a) from 10 restored medium-sized Danish streams. The dominance of *E. canadensis* and *Sparganium* sp. is probably due to the ability of the species to easily disperse from upstream areas and settle in newly created habitats where light availability favours rapid underwater growth (Nichols and Shaw, 1986; Barrat-Segretain et al., 1998; Pedersen et al., 2006a). The increased species richness and changed dominance pattern may mark a transition to a more natural plant pattern with high species richness on the banks (e.g. Nilsson, 1986, 1992). Increased species diversity on river banks and edge habitats has been reported in other studies of river restoration (e.g. Baattrup-Pedersen et al., 2000).

Plant re-colonization was intermediate in the Skjern River compared to colonization rates found in other studies (Barrat-Segretain and Amoros, 1996; Henry et al., 1996; Baattrup-Pedersen et al., 2000). Thus total plant coverage after the restoration (24%) was approximately 10% lower than before the restoration (34%) whereas a re-colonization of 10% was reported for a small Danish headwater stream 2 years after restoration (Baattrup-Pedersen et al., 2000). In contrast, Henry et al. (1996) found total re-colonization in the river Rhone 3 years after disturbance. Furthermore, propagules first colonized the edge habitat and then the remaining part of the river (Henry et al., 1996).

Re-colonization of river reaches after disturbance is primarily associated with two processes: in-stream dispersal from plant fragments and lateral colonization from the banks and riparian areas into the river. Aquatic plant dispersal mainly relies on vegetative production and is therefore usually dependant on downstream transport of either whole plants or fragments (Sculthorpe, 1967; Barrat-Segretain, 1996). Hence re-colonization is dependant on the presence of potential colonizers from upstream reaches and thus on community diversity in the entire catchment (Rosenzweig, 1995). Artificial mats aimed at protecting the banks from fluvial erosion were placed after restoration. This has slowed down the lateral colonization by terrestrial and amphibious plant species in the edge habitat. This means that the primary colonization route during the first years after the restoration has to be through the river (Henry and Amoros, 1996; Biggs et al., 2001). The lateral colonization will probably increase in importance as the riparian vegetation develops and the artificial erosion-control mats disintegrate.

The mobilisation of sediment from morphological adjustments in rivers may lead to unstable conditions unfavourable for macrophyte establishment (Biggs, 1996). Despite rapid establishment of a diverse macrophyte community in the edge

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habitat, colonization of the remaining river will probably be slow, primarily due to the high sediment mobilisation and unstable sandy substratum. The macrophyte community is not yet stable and may therefore be severely influenced by natural disturbance events such as high winter discharge which may further prolong the colonization. Given the scale of the restoration work and the subsequent morphological adjustments, a stable macrophyte community will take many years to develop. However, the proximity of large upstream refuge areas may ensure rapid colonization as the morphological instability decreases in the future.

In lowland streams and rivers, macrophytes are a very important feature and we consider the establishment of a diverse community a key factor in stabilising the river morphology. When a stable macrophyte community has colonized the reach, it will help stabilize the morphology (Jones, 1994; Sand-Jensen, 1997).

4.5. *Macroinvertebrates*

Rapid re-colonization of the restored reaches was similar to results from other studies of river restoration (e.g. Friberg et al., 1994; Biggs et al., 1998; Laasonen et al., 1998). The primary reason being availability of downstream drift as a re-colonization source (Matthaei et al., 1997). The macroinvertebrate community in the Skjern River upstream of the restored reaches is very diverse and the reaches support a high diversity in refuge areas. This result indicates that large rivers are highly resilient and recover quickly after disturbance. Even in medium-sized rivers the resilience is high and colonization is a rapid process (Friberg et al., 1994, 1998; Hansen et al., 1999). In contrast, small headwater streams lack upstream refuge areas and establishment of the macroinvertebrate community must primarily rely on immigration from other systems (Milner, 1996; Hansen et al., 1999). Recovery from disturbances is linked to availability of stable substrata which serves as refuge areas (e.g. Matthaei et al., 1999). These were present on the upstream reach (C4) as gravel stones and macrophytes and could thus act as sources for the colonization of the restored reaches.

In the Skjern River the average abundance has reached the levels found in the channelized river despite changes to the available habitats. There, however, indications of slightly lower abundances after the restoration. *Glyceria maxima* dominated the channelized river and was associated with high abundances of certain species attached to the plants (e.g. Simuliidae indet.), or species living in the soft sand and mud substratum (e.g. Tanytarsini indet.) below the plants (Merritt and Cummins, 1996). The only taxon to show a significant response to the restoration was the mayfly family Heptageniidae. This family has previously been shown to be a rapid colonizer on stones and gravel in restored rivers (Friberg et al., 1994).

The similarity results indicate the existence of significantly different communities in the channelized compared to the restored river. Moreover, dissimilarity increased among the samples in the restored reaches, indicating a shift in the species composition after the restoration. This response may be viewed as a result of the availability of a more varied habitat template in the restored reaches compared to the channelized river.

The macroinvertebrate community will potentially benefit from the restoration, which can increase community persistence by creating a higher variety of refuge areas for the benefit of more species (Boon, 1988; Mackay, 1992). As the channel morphology stabilizes over time and the habitat diversity increases, the macroinvertebrate community will have a template where biotic interactions will become increasingly important (Southwood, 1977; Hildrew and Giller, 1994). Being the most species-rich river system in Denmark, the re-colonization potential after the restoration is very high and the first indications of a potential increase in species diversity can be seen in these short-term monitoring results. Diversity in the restored part of the Skjern River will possibly increase as colonization from neighbouring systems will continue and new habitat niches are created from the increasingly stable environment.

5. Conclusion

The restoration increased the habitat diversity and resulted in more varied depth and current velocity distributions on the restored reaches in Skjern River in 2003 compared to the channelized reaches in 2000. The *Glyceria maxima* community along the river edge disappeared and replaced by a less dense edge habitat with more species.

Total macrophyte coverage decreased from 34% before the restoration to 24% after the restoration. Community composition and species growth patterns in the edge habitat became more complex after the restoration as the habitat diversity increased.

A diverse and species-rich macroinvertebrate community developed on the restored reaches. The species composition changed after the restoration and was significantly different from the community on the channelized reaches. Only the mayfly family Heptageniidae increased significantly after the restoration.

The short-term monitoring of the restoration of the Skjern River indicated minor changes to the overall biodiversity of the restored river. Initially the plant diversity in the edge habitat increased and the macroinvertebrate community was altered to reflect changes to the habitats.

6. Recommendations

From the results of the restoration of Skjern River it is evident that the initial instability associated with re-meandering of the river masks some of the effects of the restoration. Morphological adjustments in the new re-meandered channel have increased the short-term sediment mobilisation. This instability and associated fluctuations will influence the colonization of macrophytes and macroinvertebrates until the morphology and physical processes have reached a new equilibrium with the runoff regime and surrounding landscape. The full morphological and biological effect of the restoration will therefore become increasingly visible over the next years and decades as the morphology stabilises and the biotic communities mature.

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For future restoration projects it is therefore recommended to have time series before and after the restoration in order to analyse the results using a true BACI design. It is also recommended to plan the monitoring in such a way that the physical instability is accounted for the sampling strategy, e.g. by sampling several years or decades after the restoration.

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