10 years After The Largest River Restoration Project In Northern Europe

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Published in:
Ecological Engineering

DOI (link to publication from Publisher):
10.1016/j.ecoleng.2013.10.001
10.1016/j.ecoleng.2013.10.001

Publication date:
2014

Document Version
Early version, also known as pre-print

G Model
ECOENG-2759; No. of Pages 9
ARTICLE IN PRESS

Ecological Engineering xxx (2013) xxx–xxx

Contents lists available at ScienceDirect

Ecological Engineering

journal homepage: www.elsevier.com/locate/ecoleng

10 years after the largest river restoration project in Northern Europe: Hydromorphological changes on multiple scales in River Skjern

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ARTICLE INFO

Article history:
Received 26 February 2013
Received in revised form 22 August 2013
Accepted 7 October 2013
Available online xxx

Keywords:
River restoration
Long-term
Physical condition
Floodplain
Low-land rivers

ABSTRACT

The lower river Skjern (Denmark) historically contained a large variation in habitats and the river ran through large areas with wetlands, many backwaters, islands and oxbow lakes. During the 1960s the river was channelized and the wetland drained. A restoration during 2001–2002 transformed 19 km of channelized river into 26 km meandering river. The short-term effects of this restoration have previously been reported and for this study we revisited the river and with new data evaluated the long-term (10 years) hydrological effects of the restoration. The evaluation was done on three different scales: (1) in-stream habitats, (2) channel stability and (3) re-connection with the floodplain. In-stream habitats had changed little over the past 10 years and the habitats today showed close similarity with the habitats recorded immediately after the restoration. Measurements of channel stability showed that erosion and sedimentation have changed the cross-sectional profiles over the last 10 years, resulting in a net input of sediment to the lower reaches of the river. However, the change of channel form was a slow process and predicted bank retreat over a 100 year period was only up to 6.8 m. Hence the formation of lost habitats (islands, backwaters and oxbow lakes) is a very slow process and the spontaneous development of these habitats will take centuries. Furthermore, the evaluation also showed that the restoration re-connected the river with its floodplain and large areas of riparian areas are today periodically flooded, but that the flooding is controlled and tamed due to the restoration design. The restoration of River Skjern has therefore failed to re-create the natural habitats formerly present and the natural dynamic processes that shape these habitats are slow. To speed up this process we therefore recommend restoration engineering using a natural guiding image when restoring lowland rivers in the future and through this restoring the lost habitats and the dynamic processes characteristic of natural rivers.

1. Introduction

Habitat degradation is a serious threat to biodiversity (Dobson et al., 1997; Vitousek et al., 1997; Wilcove et al., 1998) and aquatic ecosystems are among those most severely impacted (Allan and Flecker, 1993; Sala et al., 2000). Over centuries, streams, rivers and their floodplains have been modified (e.g. Sparks, 1995; Kronvang et al., 1998; Bernhardt et al., 2005) as a result of land drainage, flood plain urbanization, flood defence and navigation (European Environment Agency, 1998). In North-western Europe, modification and channelization of watercourses have been particularly extensive and have left less than 10% of lowland streams in Great Britain, the Netherlands and Denmark in their natural physical state (Brookes and Long, 1990; Verdonschot and Niiboer, 2002). Thus, extensive damage has been caused to the river ecosystems with a widespread loss of habitats for biota, and the biodiversity of European rivers and floodplains is today significantly reduced.

As a consequence of the widespread damage to stream and river ecosystems, and based on a growing recognition of the conservation values within them, the number of river restoration projects has increased substantially in recent years (Bernhardt et al., 2005). River restoration efforts have primarily focused on channel re-configuration, and in-stream habitat improvements increasing heterogeneity, by re-meandering and adding physical structures such as wood, boulders and artificial riffles (e.g. Larson et al., 2001; Kasahara and Hill, 2008; Miller et al., 2010). However, during the last 10 years there has been a growing scientific and management-oriented recognition of the importance of restoring the natural processes of river ecosystems (Williams, 2001;

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http://dx.doi.org/10.1016/j.ecoleng.2013.10.001

Kondolf et al., 2006). This paradigm shift has resulted in a transition from small-scale engineering-dominated restoration approaches toward catchment-scale approaches that focus on enhancing both in-stream habitats and re-connection of the river with its floodplain and through this restoring freshwater wetlands (Hillman and Brierly, 2005; Kondolf et al., 2006). Therefore, there is a need to focus on the entire freshwater ecosystems including the riparian wetlands and through this the restoration of ecosystem processes and functioning which is vital to sustain the services these systems provide (Loomis et al., 2000). Scientific evaluations of catchment-scale restoration projects are however rare, especially studies that monitor the long-term responses (Friberg et al., 1998; Feld et al., 2011). Long-term evaluations are highly relevant as such studies can help us to advance the science of river restoration (Wohl et al., 2005) and ultimately help us achieve a higher rate of restoration success (Palmer et al., 2005).

The overall aim of this study was to evaluate the longer-term effects of restoring River Skjern, Denmark. This restoration project is the largest river restoration project in Northern Europe to date, aiming to enhance the nutrient retention capacity of the river by re-creating a natural hydrology in the river valley including re-connection of the river and the riparian wetlands and to enhance biodiversity by restoring the physical and hydrological dynamics of the river and the floodplain (Pedersen et al., 2007a). River Skjern is located in Western Denmark and drains a catchment of 2490 km^2. Land use in the catchment is dominated by agriculture and the geology of the area is a combination of sandy outwash plains and mostly sandy moraines (Smed, 1982). The river has the highest discharge of any Danish rivers (annual mean 35 m^3/s) why the river is of high regional importance as a biodiversity hotspot (Ovesen et al., 2000; Andersen et al., 2005). From historical maps dating back to the 1800th century the lower 10 km of the river can be classified as anastomosing with numerous channels between low relatively stable vegetated islands (Miall, 1977; Richards, 1997). During the late 1960s the lower 19 km of the river was channelized and riparian wetlands were drained as a result of increasing demand for agricultural production. River channelization and drainage was at the time considered a prerequisite condition for agricultural growth in the Danish society. However, 25 years later the area had lost its agricultural value and in 1987 the Danish government initiated plans to restore the area. The restoration was conducted in 2000–2002 and resulted in the transformation of the lower 19 km of channelized river into 26 km of meandering river (Pedersen et al., 2007a). The short-term effect of the restoration on in-stream habitats, macrophytes and macroinvertebrates has previously been reported (Pedersen et al., 2007b), however, the longer-term effects are unknown. The aim of this study was therefore to re-visit the River Skjern and analyze the development in channel morphology and habitats 10 years after the completion of the restoration. We investigated morphological development using three different spatial scales: (1) in-stream habitats, (2) channel stability and (3) the re-connection of the river with its riparian wetlands. We hypothesized that significant changes have occurred during these 10 years and that the River Skjern has developed into a river system with near-natural hydromorphology. The term hydromorphology is used as defined by Ŝípek et al. (2009), for discussion see Vogel (2011).

2. Methods

2.1. In-stream habitats

The short-term effects of the restoration on in-stream habitats have previously been evaluated in three 300 m long reaches along the restored River Skjern (R1, R2, R3, Fig. 1) based on a comparison with a 300 m long control reach (C, Fig. 1) located upstream of the restoration area (Pedersen et al., 2007b). All four reaches was sampled once before the restoration (2000) and again immediately after the restoration (2003; Pedersen et al., 2007b). After the restoration the location of R2, R3 and C remained at the same location as pre-restoration, while, as a results of the restoration and filling-up of major parts of the channelized river, reach R1 was in 2003 moved from the northern drainage channel to the newly excavated river channel located app. 2 km south (Fig. 1). For this study, we re-sampled these four reaches in 2011 providing data for a long-term evaluation (app. 10 years) of the restoration on in-stream habitats. Identical surveying methods were used in all three years to allow for cross-year comparison, for a detailed description of the methodology, see Pedersen et al. (2007b). In brief, six transects were placed equally spaced along each of the four reaches and each transect was divided into 1 m × 1 m quadrats across the entire width. A GPS was used to exactly identify location of each transect. At each quadrat, depth (to nearest cm), current velocity (at 10 cm above the stream bed), dominating substrates (using seven categories according to the Wentworth-scale (Wentworth, 1922) and macrophyte coverage (%) was recorded. Recording of in-stream variables was done in September 2000, August 2003 and September 2011 and it was aimed to collect data at similar discharge levels. However, the summer 2003 was drier than normal and mean monthly discharge for August 2003 was 12.7 m^3/s, while mean monthly discharge for September 2000 and September 2011 was 19.4 m^3/s and 19.1 m^3/s, respectively.

To evaluate the long-term changes to in-stream habitats we divided the recordings from each transect into two groups two groups termed a "Vegetated zone" and a "Main current zone". The first group was defined as quadrats with depths from 0 to 130 cm, often located along the edges of the river channel supporting vascular macrophytes, as these rarely occur at depths larger than 130 cm. The second group was quadrats with depths larger than 130 cm, often located in the mid-channel and being without vascular macrophytes. We preformed this a priori separation of the data to obtain a river-zone-specific evaluation of the physical changes during the 10 year period because these two main channel zones are expected to form different in-stream habitats for plants, macroinvertebrates and fish. For this study, we calculated a number of in-stream parameters using the transect data recorded in 2000, 2003 and 2011. For each transect, we calculated Coefficients of Variation (CV) for depth, mean current velocity and mean macrophyte coverage separate for the two habitat zones. In addition, we used substrate recording to calculate percent occurrence of four substrate types (peat, mud, sand and gravel) and produced transect means for each substrate type separately for the two zones. Finally, we used three different variables (domination, diversity and score) to describe changes in substrate for each transect divided into the two zones according to O'Hare et al. (2006). A value between 1 and 4 was allocated to the four substrate categories with values increasing with particle size. Domination was defined as the dominant substrate, i.e. the category occurring in most quadrats, diversity was the number of categories occurring and score the weighted average of the categories present in each transect. All physical variables used to evaluate changes to in-stream habitats for the four reaches and the two different zones are summarized in Table 1.

To investigate the effect of the restoration on in-stream habitats and the long-term development in the habitats we preformed Principal Response Curve analyses (PRC, Van den Brink and Ter Braak, 1999). The analyses were done with year 2000 as reference points (Van den Brink et al., 2009) thus enabling us to investigate the change of in-stream habitats relative to the physical condition

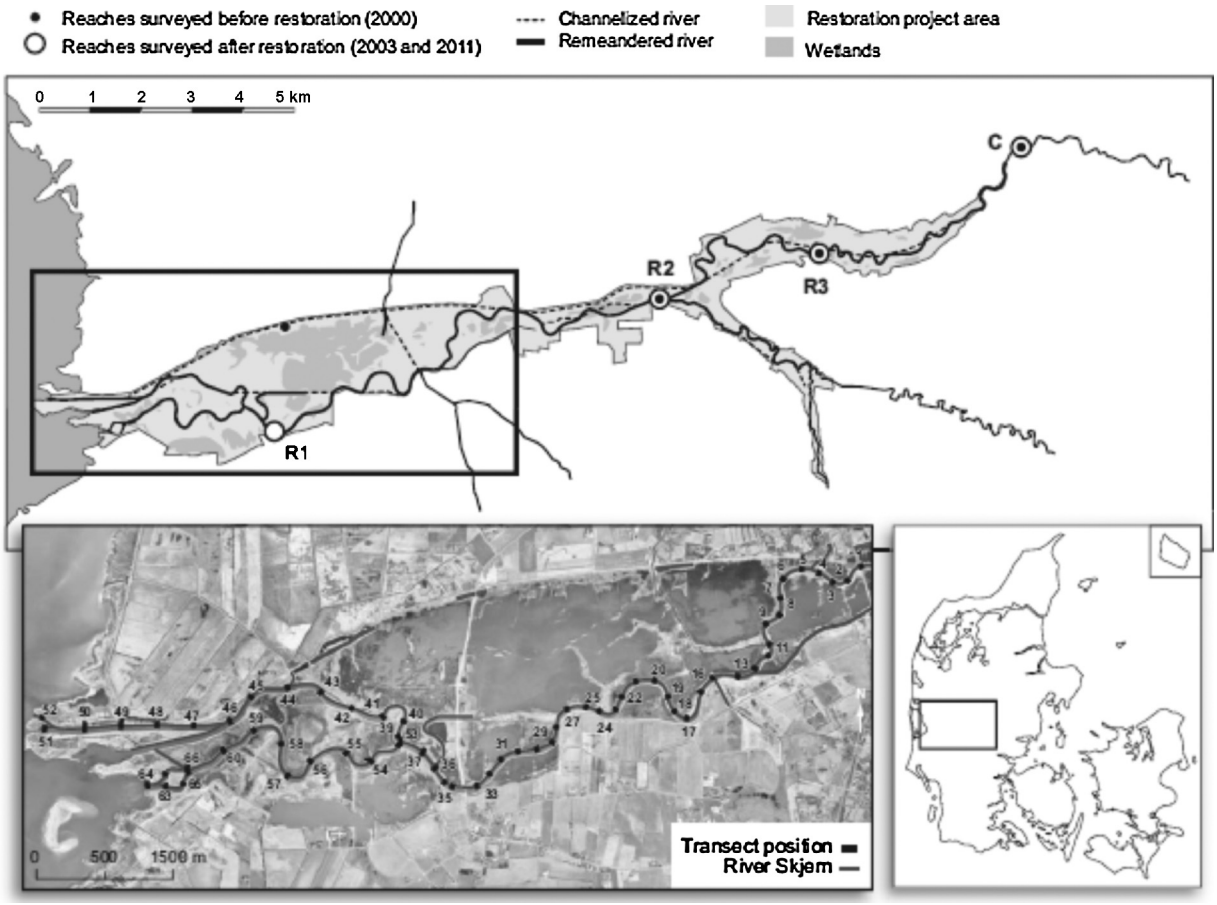

Fig. 1. Map showing the study area, study sites for in-stream habitats (R1, R2, R3 and C), and the position of transects used to measure channel stability (1–66).

before the restoration. All analyses were done using R (version 2.15, R Core Team, 2012) and we performed the analyses separately for each of the four river reaches and separately for the two different habitat zones. The PRC analyses with year 2000 as a fixed reference point were performed in a series of steps. First we calculated a Redundancy Analysis (RDA) using the RDA function (vegan package). All in-stream variables (Table 1) were *z*-standardized prior RDA. We then extracted the scores of the first RDA axis for each transect, calculated mean RDA scores for each year and calculated the difference in mean RDA scores between the reference year (2000) and years 2003 and 2011. This difference in RDA scores was then plotted against time to produce the final PRC

Table 1
Physical variables (mean with range) included in the analysis of in-stream habitats for the vegetated zone (a) and the main current zone (b). See text for further explanations.

	R1	R2	R3	C
(a)				
CV depth	0.36 (0.13–0.64)	0.31 (0.06–0.68)	0.42 (0.13–0.70)	0.41 (0.05–0.83)
Current velocity (m/s)	–	25 (2–37)	30 (2–51)	–
Macrophyte coverage (%)	78 (14–132)	47 (0–108)	57 (0–143)	132 (0–260)
Peat (%)	4 (0–30)	0.01 (0–0.01)	0.01 (0–0.01)	0.01 (0–0.01)
Mud (%)	22 (0–53)	18 (0–87)	13 (0–66)	23 (0–67)
Sand (%)	74 (30–100)	78 (13–100)	79 (16–100)	67 (33–100)
Gravel (%)	(0–8)	3 (0–40)	7 (0–77)	9 (0–51)
Dominance_substrate	2.8 (2–3)	2.8 (2–3)	2.9 (1–4)	2.8 (2–3)
Diversity_substrate	2.1 (1–4)	1.6 (1–2)	1.8 (1–3)	1.7 (1–3)
Score_substrate	271 (200–300)	284 (213–340)	292 (250–377)	285 (233–350)
(b)				
CV depth	0.08 (0.01–0.14)	0.06 (0.03–0.11)	0.06 (0.01–0.15)	0.07 (0.02–0.12)
Current velocity (m/s)	–	–	38 (25–54)	35 (27–46)
Macrophyte coverage (%)	28 (0–140)	6 (0–20)	14 (0–46)	13 (0–28)
Peat (%)	0.4 (0–5)	0	0	0.01 (0–0.01)
Mud (%)	2 (0–7)	2 (0–7)	0	1.4 (0–7)
Sand (%)	95 (75–100)	93 (60–100)	73 (0–100)	84 (72–95)
Gravel (%)	3 (0–17)	5 (0–40)	27 (0–100)	15 (0–28)
Dominance_substrate	3 (3)	3 (3)	3.3 (3–4)	3 (3)
Diversity_substrate	1.8 (1–4)	1.5 (1–2)	1.4 (1–2)	2.1 (2–3)
Score_substrate	300 (294–313)	303 (293–340)	327 (300–400)	314 (300–328)

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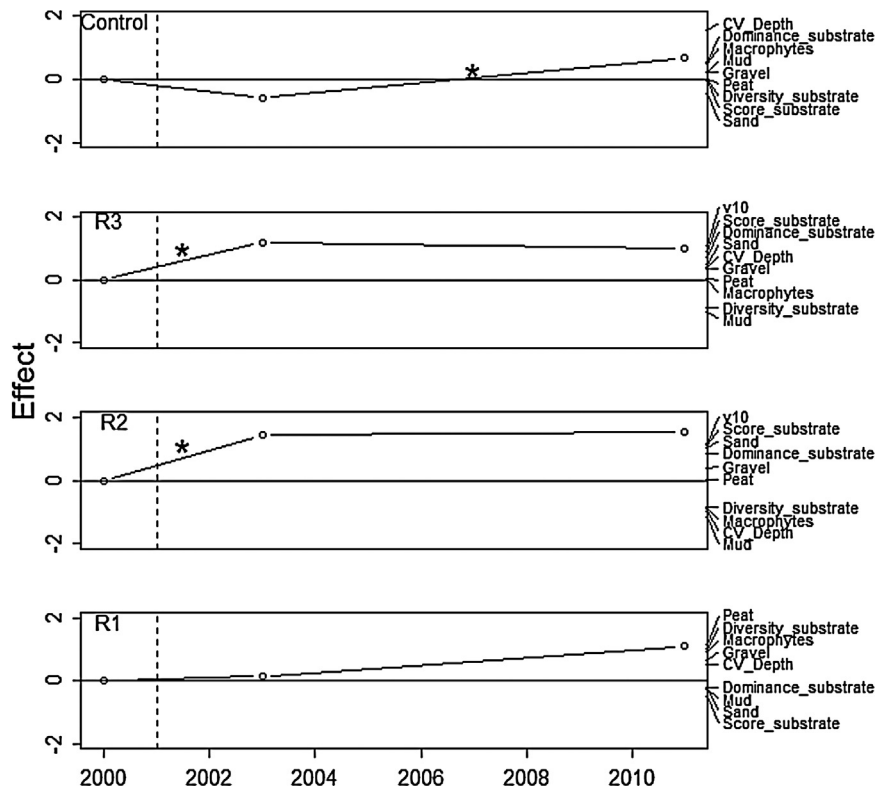

Fig. 2. Principal response curve analyses of the development of in-stream habitat parameters in four different sites of the vegetated zone (depths 0–130 cm) of River Skjern: control upstream the restored reaches (R1, R2, R3).

plot. Current velocities (v10) were not measured in the vegetated zone of the control reach and both zones of R3 why the analysis were performed without this variable for these. Following PRC analyses we used one-way ANOVA's to test for a significant effect of year. We used the RDA scores for each transect of the first axis for 2000 and the difference in scores between the reference year (2000) and years 2003 and 2011 in the ANOVA's. When the global one-way ANOVA was significant, we used Tukey pair-wise comparisons to compare years.

2.2. Sedimentation and erosion

To study the stability of the restored river channel and investigate how sedimentation and erosion affects the cross-sectional profile of the river, 65 transects were surveyed along the lower 10 km of the river (Fig. 1). Detailed measurements of these 65 cross-sections were performed immediately after the restoration (in 2001) and again in 2011. All cross sections were surveyed using RTK GPS equipment with a vertical error <20 mm (Leica Geosystems, 2010) and for each year *x*, *y* and *z* coordinates were obtained for 20 points per transect. After measurements all data points were transformed into profile lines. Erosion and sedimentation volumes were then calculated for each transect as the difference in height of the riverbed (*z* coordinate). Calculations were done separately for three cross-sectional compartments: (i) the left bank, (ii) the right bank and (iii) the river bed. The erosion and sedimentation was estimated as both a gross and a net amount and to determine the erosion and sedimentation between two consecutive transects we used nearest neighbor interpolation. In addition, we calculated bank retreat for both left and right bank for each transect.

2.3. Re-connection with the floodplain

To study the connectivity between the River Skjern, and its surrounding riparian areas river, water levels were modeled using the one dimensional hydraulic model MIKE11 (DHI, 2011). The hydraulic model was based on trans-sectional profiles no. 1–33 (Fig. 1) measured in 2001. The upstream model boundary was based on measured daily river flows covering the period 2001–2011 (Ovesen et al., 2000) and the downstream model boundary was based on sub-daily measurements of sea surface water levels at Bork Harbor located 9 km south west of the river outflow. The model was calibrated against measured water levels by adjusting a time series of daily Manning *M* numbers calculated at the most downstream permanent river flow gauging station on the River Skjern (Thodsen et al., 2006). When the modeled water levels were higher than the river banks the extent of the flooding was estimated based on a 1.6 m resolution LIDAR DEM (KMS, 2010) and the combined flooded area were calculated. Riparian wetland areas being flooded were estimated based on the 90th flow percentile in the river during the period 2001–2011.

3. Results

3.1. In-stream habitats

When comparing RDA scores, we found no significant overall change to in-stream habitats in the vegetated zone of the control reach from 2000 to 2003 ($P = 0.493$; Fig. 2) however, from 2003 to 2011 there was a significant change ($P = 0.0008$; Fig. 2). This change was primarily caused by an increase in CV Depth, indicating more heterogeneous stream profiles with larger variation in depth in

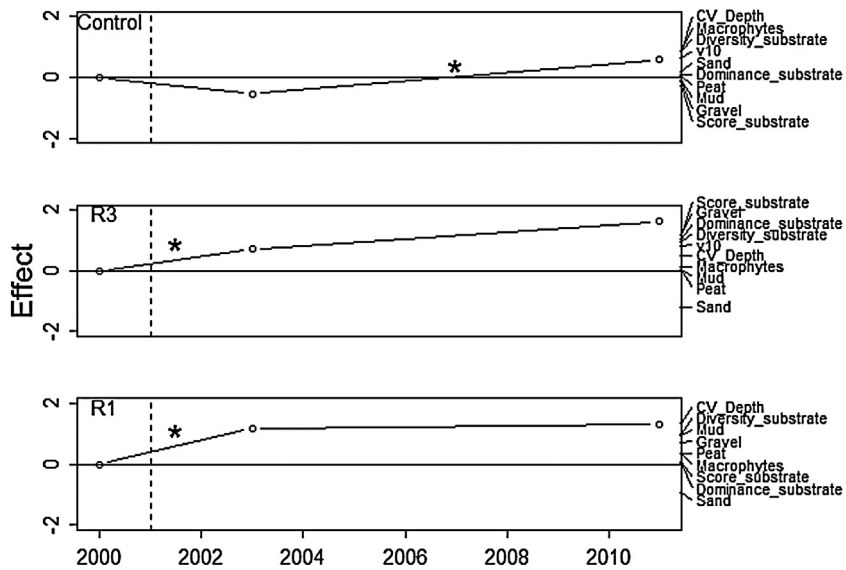

Fig. 3. Principal response curve analyses of the development of in-stream habitat parameters in four different sites of the current zone (depths >130 cm) of River Skjern: control upstream the restored reaches (R1, R2, R3).

2011 compared to 2003. In the vegetated zone of R3 and R2 the restoration resulted in a significant change to in-stream habitats ($P = 0.022$ and $P = 0.026$, respectively). In this zone of R3, the restoration resulted in higher current velocities (v10), a coarser substrate (Score_substrate) although still dominated by sand, a higher coverage of gravel, a reduction in coverage by mud and consequently a lower substrate diversity (Fig. 2). In the vegetated zone of R2, the restoration also resulted in increased current velocities (v10), a coarser substrate (Score_substrate) although still dominated by sand, a relatively higher coverage of gravel, lower coverage of mud and a lower substrate diversity (Fig. 2). In addition, the analysis showed that the overall significant changes to in-stream habitats of the vegetated zone of R3 were also related to a reduction in macrophyte coverage (Fig. 2).

The observed changes to in-stream habitats in the vegetated zone of R3 and R2 were stable over time showing no change from 2003 to 2011 ($P = 0.906$ and $P = 0.722$; Fig. 2). The in-stream habitats of the vegetated zone in the most downstream reach (R1) was not significantly affected by the restoration as only very small changes occurred from 2000 to 2003 (Fig. 2). From 2003 to 2011 there was some development in in-stream habitats but these changes were not significant ($P = 0.897$; Fig. 2).

Similar to the vegetated zone, we found no significant overall change to in-stream habitats in the main current zone of the control reach from 2000 to 2003 ($P = 0.460$; Fig. 3) and a small but significant change from 2003 to 2011 ($P = 0.011$; Fig. 3). This change was primarily driven by an increase in CV depth, macrophyte coverage and substrate diversity over the period (Fig. 3). In the main current zone of R3 and R1, the restoration resulted in a significant change to in-stream habitats ($P = 0.02$ and $P = 0.011$ respectively; Fig. 3), primarily caused by a higher variation in depth (CV depth) and higher coverage by gravel and consequently a higher substrate score, a higher substrate dominance score and a higher substrate diversity (Fig. 3). As found for the vegetated zone of R3 and R1, the observed changes to in-stream habitats following restoration, stayed stable over time in the main current zone of R3 and R1 ($P = 0.161$and $P = 0.897$, respectively; Fig. 3). No analysis was performed for R2 for the main current zone as only very few quadrates had depths over 130 cm.

3.2. Sedimentation, erosion and bank retreat

The survey of 65 cross-sectional transects along the lower 10 km of River Skjern in 2001 and again in 2011 revealed marked morphological adjustments through erosion and sedimentation in the newly restored stream channel (Fig. 4). There was large variations in the morphological adjustments among transects during the 10 year period (Fig. 5). Average sedimentation and erosion volume (±SD) measured at the left bank amounted to $231 \pm 360\,m^3$ and $274 \pm 325\,m^3$, respectively, resulting in a net erosion of $43\,m^3$ sediment. For the right bank, average sedimentation and erosion volume (±SD) was $192 \pm 340\,m^3$ and $536 \pm 740\,m^3$, respectively, yielding a net erosion of $344\,m^3$ sediment. However, the most pronounced morphological changes were observed at the river bed. Average sedimentation (±SD) for this part of the cross-sectional transects was $2259 \pm 2333\,m^3$ sediment and average erosion (±SD) was $707 \pm 996\,m^3$ resulting in a net sedimentation of $1552\,m^3$ sediment. Combined, the net balance resulted in a gain of $1166\,m^3$

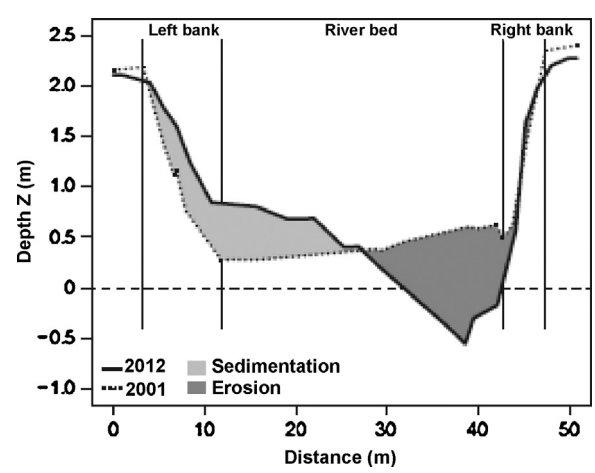

Fig. 4. Example of cross-sectional adjustments of the River Skjern between 2001 and 2012.

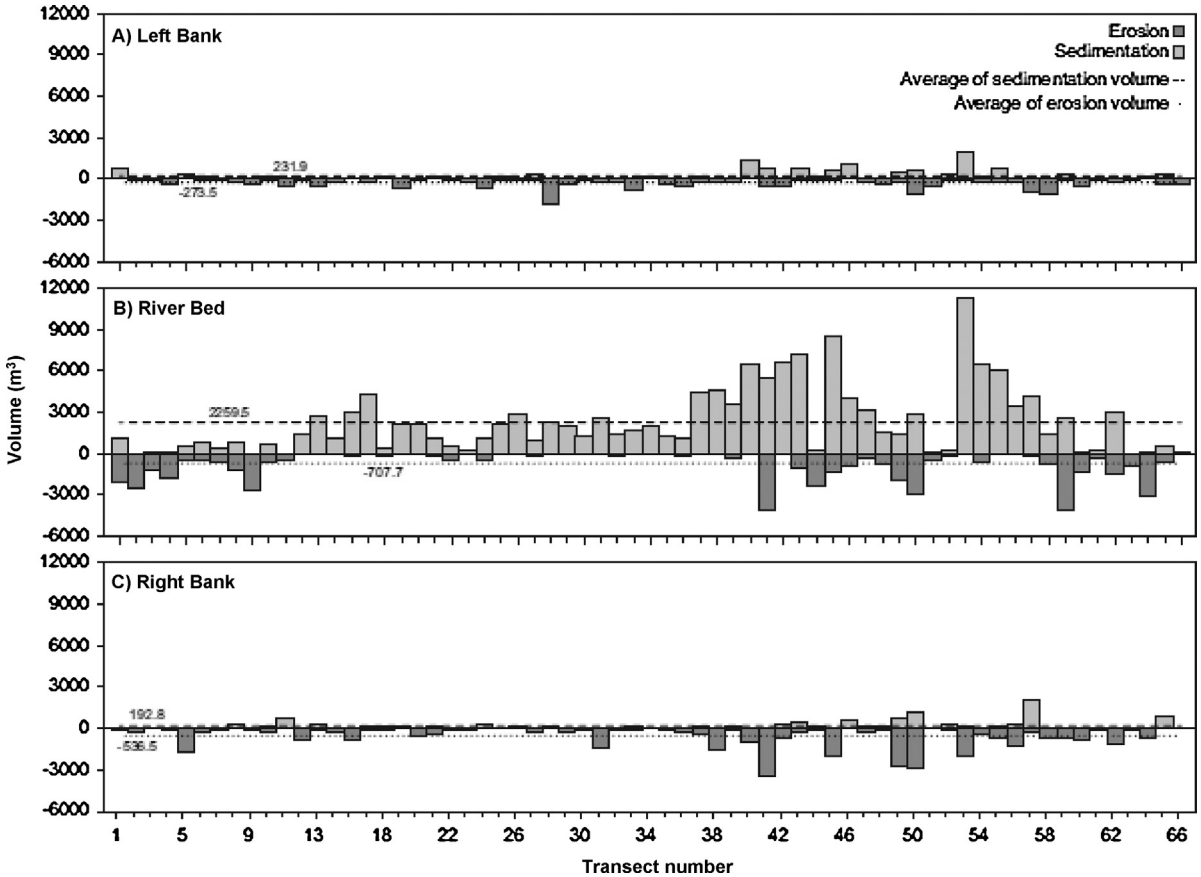

Fig. 5. Erosion and sedimentation in the lower 10 km of the restored River Skjern over a 10 year period (2001–2011).

sediment over the 10 years in the restored river channel. The net result is a slightly wider and shallower river in 2011 than in 2001.

There was large variation in bank retreat among transects (Fig. 6) and average bank retreat for left banks was 39 cm and 68 cm for right banks. This corresponds to a forecasted bank retreat of 3.9 m for left banks and 6.8 m for right banks over a 100 years period.

3.3. Re-connection with the floodplain

Before the restoration the channelized river had very little or no connection with the floodplain due to protection of the adjacent agricultural areas with dikes along the river channel and extensive drainage through pumping stations and ditches. The restoration immediately re-connected the river with the floodplain and based on the 2001 survey of the river channel cross-sections the MIKE11 hydraulic model revealed that 611 ha riparian areas located between transect 1 and 33 (Fig. 1) were flooded for 10% of the period (2001–2011). The modeling also revealed that flooding was most pronounced during winter as there was connection between the river and the floodplain for 30.5% and 25.2% of January and February, respectively. During summer months flooding was very rare and occurred for less than 1% of May, June, July and August.

4. Discussion

Evaluating the success of river restoration projects is often made impossible due to the lack of both pre- and post-monitoring data (Kondolf and Micheli, 1995; Lake, 2001; Palmer et al., 2005). This lack of data has for decades hampered progress in our scientific and practical understanding of what defines a successful river restoration project. The restoration of River Skjern is an exception to this general rule as both pre- and post-monitoring data over short- and longer-term exist. These data therefore provides us with the opportunity to evaluate and analyze success and use the results to improve future river restoration projects. However, the lack of monitoring data is not the only reason halting our understanding of river restoration success. Another important factor is the lack of agreed criteria for judging success (Palmer et al., 2005). The restoration of River Skjern was no exception and restoration goals concerning hydromorphology were vaguely defined as the aim to "restore the physical and hydrological dynamics of the river and floodplain" (Pedersen et al., 2007a). We therefore choose to evaluate the hydromorphological outcome of the restoration on multiple scales in order to effectively determine if the restoration had allowed for a more dynamic and natural river.

4.1. In-stream habitats

We found immediate changes to in-stream habitats following restoration and there were significant differences at reach-scale in-stream habitats between 2000 and 2003 for most of the restored reaches and for both habitat zones – primarily due to increase in current velocity (through narrowing of the channel) and addition of coarse sediment during restoration. The channelized stream had been constructed in order to secure effective drainage of agricultural areas and the channel was therefore wide and very straight. Consequently, the variation in depth was relatively small. The restoration decreased the width of the stream channel and consequently increased current velocity. The only reach where the

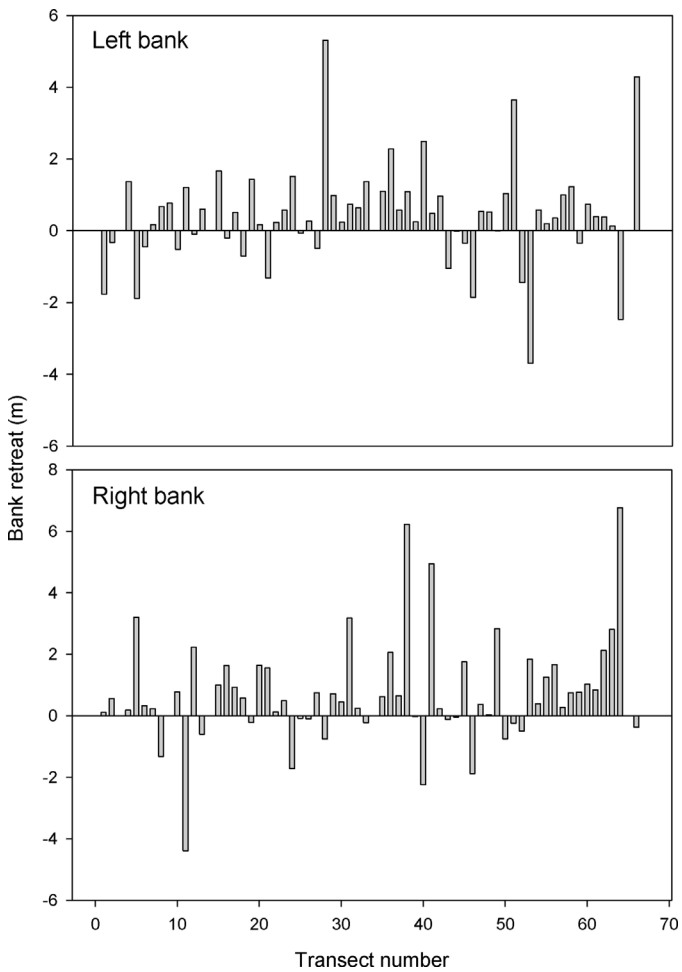

Fig. 6. Bank retreat in the lower 10 km of the restored River Skjern over a 10 year period.

restoration did not have an immediate impact on in-stream condition was the reach located most downstream (R1). At this reach, the restoration only decreased average transects width from 47.7 m to 45.7 m (4% decrease), while the decrease in average width was 29% and 17% for reaches R2 and R3, respectively. The addition of gravel is a common restoration practice in Denmark (Pedersen et al., 2009; Kristensen et al., 2011) and R2 and R3 also received gravel as part of the restoration. Gravel was not added to R1 which together with the limited reduction in width reflects the downstream location of this reach. Excessive addition of coarse material and further narrowing of the stream channel would therefore not have been in accordance with the natural river type for this location.

10 years after the restoration, we found that the in-stream habitats had gone through very little changes compared to the 2003 situation. The condition of all restored reaches was not significantly different from the condition created 10 years before. By nature, rivers and river channels are temporally variable (Rosgen, 1996) and natural rivers are characterized by continuous changes to channel form and habitats (Petts et al., 1995). These natural structural processes are important for river biodiversity and biofunction (Elosegi et al., 2010) and therefore ultimately for the ecological quality of rivers. However, defining the natural level of dynamic hydromorphological processes in rivers are not easy (Sear and Newson, 2003; Newson and Large, 2006) and relatively stable periods might be interspersed with periods of disturbance or change. Large changes to in-stream habitats in River Skjern over the time spand of our evaluation would probaply require more dynamism than is possible at the moment. This is primarilly due to the lack of large structuring elements (e.g. large woody debris; LWD) in the river – elements that can be transported downstream and create erosion of banks and streambed and consequently change the habitats. The question is therefore if the restoration of the lower River Skjern has created an artificially stable condition that leaves little room for the dynamics that forms natural rivers and creates habitats for a large variety of flora and fauna? Definitive answers to this question are difficult due to the relatively short time span of our evaluation however, it is very likely that increasing the occurrence of LWD in the rivers would increase the dynamism of the river. This would require a change to the current management strategy of Danish rivers where LWD are removed from the channels. Furthermore, it is importance to acknowledge the fact that we performed a transect-scale analysis of the long-term development in physical conditions. Had we done the evaluation on a smaller scale (e.g. quadrate-scale or microhabitat scale) temporal changes would most likely have been larger as temporal dynamics increase with decreasing spatial scale (Frissell et al., 1986).

4.2. Sedimentation, erosion and bank retreat

Immediately following restoration, the newly created stream channel of River Skjern experienced marked morphological adjustments because of the unconsolidated state of the river bed and banks (Pedersen et al., 2007b). This phenomenon has been reported for other river restoration projects as well (e.g. Sear et al., 1998). The present study has documented further morphological adjustments to the channel and 10 years later erosion and sedimentation has altered the cross-sectional profiles significantly. Along the investigated 10 km of river there was a surplus of sediment indicating a decrease in bed sediment transport capacity compared to further upstream. Over time the surplus of sediment and deposition along the lower parts of the River Skjern will lead to increased flooding that potentially will erode flood channels that again potentially could develop into permanent channels and recreate the formerly existing anastomosing river planform. Furthermore, the surplus of sediment can also over time lead to the formation of fluvial islands – a habitat type that existed historically in the river. It must be estimated that the time horizon of a new naturally developed anastomosing river and fluvial islands pattern is long, probably centuries. There are yet no safe indications on this development being ongoing. In other river types (braided rivers) the formation of islands occurs over 10–20 years (Gurnell et al., 2001) and is typically initiated when LWD lodges on a shallow section of the river (Ward et al., 2002). The formation of islands process is probably slower in lowland rivers and LWD is completely absent from most Danish rivers (including River Skjern) as trees are often not allowed to grow along river channels due to their extensive use for farming and any wood are routinely removed through river maintenance practices (Kristensen et al., 2012). Abandoning this practice and allowing trees along rivers and LWD in the rivers would benefit the creation of dynamic river channels (Gurnell et al., 2005) and eventually speed up the formation of fluvial islands.

In general, is the dynamics of river channels dependent on which scale the morphological changes occur (Frissell et al., 1986). At small scale (e.g. sediment rearrangement), changes occur over hours or days (Elosegi et al., 2010). Changes at a larger scale, such as lateral migration of river channels that leads to meander cut-off and the formation of oxbow lakes can take centuries (Gilvear and Bravard, 1996; Hooke, 2004). We measured the lateral migration rates (bank retreat) of River Skjern to be 3.9 cm/year and 6.8 cm/year for the left and right bank, respectively. Compared to reported migration rates for another Danish river (River Odense)

the rate for River Skjern is relatively high (Kronvang et al., 2012) but the rate is relatively low compared to lowland rivers from other regions (Gilvear and Bravard, 1996). Meander cutoff and formation of oxbow lakes in the restored river can therefore not be expected within the next centuries. Historical maps confirm that oxbow lakes and fluvial islands were features of the lower River Skjern before channelization and drainage however, these channel characteristic were not re-created during restoration (Pedersen et al., 2007a). There has been extensive research on the roles of channel complexity on biodiversity and ecosystem functioning (e.g. Hutchinson, 1959; Beisel et al., 1998; Aldridge et al., 2009; Elosegi et al., 2010) providing evidence for a positive relationship between complexity and biodiversity. Given the extensive periods of time needed before spontaneous re-formation of islands and oxbow lakes in River Skjern and the fact that these features were present historically, active re-creation of them during restoration would have increased chances of ecosystem recovery. We did not include biological samples in the current evaluation of the River Skjern restoration however, it is likely that species inhabiting shallow and slow-flowing areas (e.g. backwaters or oxbow lakes) have not made a full recovery due to the scarcity of these habitats today.

4.3. Re-connection with the floodplain

The restoration of the lower River Skjern re-connected the river with the floodplain and large areas are frequently flooded and some areas are even permanently water filled (Andersen et al., 2005). Especially the birdlife has responded positively to this change and the number of breeding species in the area increased from 7 to 31 between 2000 and 2003 (Andersen et al., 2005) however, this number has declined in recent years (Holm, T.E., unpublished data). During flooding, the water delivers sediment to the floodplain which contains seeds and the re-connection can therefore help to increase the possibility of re-colonization by plant species to the riparian areas (Baattrup-Pedersen et al., 2012). Furthermore, flooding can also increase retention of phosphorous and nitrogen (Baldwin and Mitchell, 2000; Kronvang et al., 2007) that otherwise would end up in the marine recipient and potentially cause environmental problems. The benefits from re-connection of rivers with their floodplain are therefore plentiful and there are many more than the ones mentioned here (Tockner et al., 2000). However, the flooding along River Skjern does not follow a natural pattern. It has previously been estimated that discharge in River Skjern have to be above 40 m^3/s before the river is connected with the floodplain (Andersen et al., 2005). This threshold value is a consequence of a desire to limit water flow through riparian wetlands and the construction of the restored river channel therefore included a designated in-flow area. This design was used to minimize the loss through predation of downstream migrating Atlantic salmon smolts (*Salmo salar*) in the wetlands (Andersen et al., 2005). However, this design limits the flooding frequency and the amount of water that flow into the floodplain (Andersen et al., 2005). Furthermore, the limited connection probably also has consequences for the morphological dynamics of the restored river channel as the channel is fixed in places to maintain the design.

5. Conclusion

With this study we have highlighted the value of long-term monitoring following river restoration and present data from an evaluation 10 years after one of the largest river restoration projects in Europe. Long-term monitoring is rare and this study therefore provides us with a unique opportunity to improve our knowledge about restoration success (Palmer et al., 2005). We evaluated the restoration on three different morphological scales (in-stream, channel morphology and re-connection with the floodplain) and how the morphology had hanged over the last 10 years. We found that in-stream habitats had changed very little since the immediate change following restoration and that the relatively stable stream channels were created during restoration. Along this line, we also measured the dynamic processes shaping river channels (erosion and sedimentation) and found that the rate of changes are relatively slow and the spontaneous creation of lost habitats (islands, backwaters and oxbow lakes) will take centuries. Moreover, although we found that the restoration had re-connected the River Skjern with its floodplain, this re-connection was tamed and controlled. If the aim of restoring River Skjern was to bring back the lost habitats, and the flora and fauna associated with these, we can therefore concluded that the restoration is not yet a success. Rivers are dynamic and the processes that will shape the lost habitats are slow and even within the scope of the present study (10 years), which is relatively long-term, the success can therefore be evaluated. However, to speed up this process we recommend restoration engineering using a natural guiding image when restoring streams and rivers in the future and through this restoring the lost habitats and the dynamic processes characteristic of natural rivers. We did not evaluate the biological recovery of River Skjern following restoration in the present study, but recently collected data (Wiberg-Larsen, unpublished data) suggest that the many macroinvertebrate and macrophytes associated with islands, backwaters and other slow-flowing areas have not returned to the restored river – partly because the habitats have not been re-created. Only when doing this we can expect a return of the lost species.

Acknowledgements

This study was funded by the 15. Juni Foundation and the EU project REFORM. We wish to thank Sandra Hille for statistical advice and Uffe Mensberg and Henrik Stenholt for assistance in the field.

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Please cite this article in press as: Kristensen, E.A., et al., 10 years after the largest river restoration project in Northern Europe: Hydromorphological changes on multiple scales in River Skjern. Ecol. Eng. (2013), http://dx.doi.org/10.1016/j.ecoleng.2013.10.001