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REVIEW

Notes on the ecological restoration of fen meadows, ombrogenous bogs and rivers: definitions, techniques, problems

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Summary

1 Definitions of the current technical terms in the field of restoration ecology are followed by a discussion of restoration efforts in three ecosystem types: fen meadows, ombrogenous bogs and rivers.

2 The main steps in restoring species-rich oligotrophic wet meadows on formerly fertilized grassland or arable land are the removal of excessive nutrients, the correction of the water table and the re-introduction of species.

3 Restoration of ombrogenous bog depends mainly on the successful re-establishment of suitable hydraulic conditions which will often give rise to a spontaneous recolonization by typical bog species; details of the artificial re-introduction of *Sphagnum* species are also given.

4 In restoring riverine ecosystems, for example by recreating meanders, apart from controlling flood hazards consideration will have to be given primarily to a variable design but consequences on the sediment transport should be carefully studied. Vegetation can in most cases be left to natural succession, whereas providing unhindered up- and downstream migration and resting places for animals is an important issue.

Keywords: nutrient removal, soil removal, species re-introduction, time scales, water table adjustment

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Introduction

During the last decade the number of projects involving ecological restoration has multiplied in great numbers. The dramatic loss of biotopes and its accompanied loss of species has led to new paradigms in nature conservation – mostly defensive strategies to preserve and save nature from further destruction are being replaced by more affirmative ones: sites that have suffered from various forms of destruction are now being restored, rehabilitated or reclaimed. This article has two objectives: (1) to define the meaning of terms most commonly used in this new branch of conservation biology (cf. Bradshaw 1997), and (2) to provide an overview over the most important applications of ecological restoration of wetlands. While a similar article by Pfadenhauer & Klötzli (1996) is based primarily on experience gained in projects of continental Europe we here include more work from non-European countries.

Whereas in the past primarily technical issues, with questions such as: "What is good restoration?" were focused, the interest now is shifting towards a more reflective attitude. Higgs (1997) has pointed out that restoration projects should not merely satisfy biological criteria, but that historical, cultural and social issues should also be considered. Thereby successful restoration should ultimately help to establish healthier relationships between people and the ecosystems in which they live. Although still dealing with mostly technical matters regarding the restoration of fens, bogs and riverine systems, we hope to contribute to this goal.

DEFINITIONS

"Restoration" (German *Regeneration*) by its strictest definition implies a return to a former or original state (Webster's New Collegiate Dictionary 1983). The degree of match between former (or original) and the restored state thereby is not explicit but it is implicit that a restored object has substantial similarity with a former condition (Wheeler 1995). The Society for Ecological Restoration (SER) defines ecological restoration as "the intentional alteration of a site to establish a defined indigenous, historic ecosystem. The goal of this process is to emulate the structure, functioning, diversity, and dynamics of the specified ecosystem". Similarly, Lewis (1990) defined restoration as return "from a disturbed or totally altered condition by some action of man" but "for restoration to occur it is not necessary that a system be returned to pristine conditions". Finally, the National Research Council (1992) defined restoration of aquatic ecosystems as the "re-establishment of pre-disturbance aquatic functions and related physical, chemical and biological characteristics". Because a complete return to predisturbance conditions is hardly ever possible, restoration usually means "returning an ecosystem to a close approximation of its condition prior to disturbance" (National Research Council 1992). The ecosystem state after restoration should be self-sustaining (requiring minimal maintenance or management or no maintenance at all), and the natural dynamic ecosystem processes should operate effectively again (Henry & Amoros 1995). A continuum of restoration efforts can be recognised, ranging from restoration of localised highly degraded sites to restoration of entire landscapes for production and/or conservation reasons (Hobbs & Norton 1996). Restoration of ecosystems functions are believed to be more important than restoration of the precise structure (Bradshaw 1997).

"Rehabilitation" (German *Renaturierung*) is a broad term that may be used to refer to any attempt to restore elements of structure or function of an ecological system, without necessarily attempting complete restoration to any specified prior condition, for example replanting of sites to prevent erosion (Meffe & Carroll 1994). Rehabilitation often involves the provision of new chemical or physical structures, that enhance the formation of a specific community (Gore & Shields 1995). Rehabilitation is sometimes used informally as a general term for the re-creation of unspecified wildlife interest (Wheeler 1995).

"Reclamation" typically refers to rehabilitative work carried out on the most severely degraded sites of such lands disturbed by opencast mining. Although in most cases no full restoration is achieved reclamation is a necessary first step in this direction. So far, the disciplines of restoration and reclamation have developed quite independently, however, some communication has been established recently (Meffe & Carroll 1994).

"Ecological recovery" is a term more or less synonymous with ecological succession and implies letting the system return to a previous state on its own.

"Recreation" attempts to reconstruct an ecosystem, wholesale, on a site so severely disturbed that there is virtually nothing left to restore (Meffe & Carroll 1994).

Restoration of fen meadows

Intensive agricultural production in the last few decades has led to a drastic loss of fens in central Europe. With shifting priorities in agricultural politics and high surplus production farmers are nowadays increasingly motivated to decrease productivity on previously intensively farmed areas. Since intensification of agriculture in this century has been identified to be one of the major causes for loss of species (Kaule 1991; Blab 1993) the reversal of this process appears to be of key importance in contemporary nature conservation (Tesch 1992). In central Europe several large-scale fen restoration projects are currently under way, e.g. in northern Germany (Pfadenhauer 1995; Pfadenhauer & Klötzli 1996: Friedländer Grosse Wiese, Drömling, Havelland, Dümmer), The Netherlands (van Diggelen *et al.* 1994: Gorecht area), and in Switzerland (Buttler 1985, 1992; Enz 1996: Nussbaumer Seen, Grande Cariçaie, Reusstal).

PROBLEMS

The re-establishment of typical fen vegetation on previously fertilised land is not an easy task. Depending on soil conditions nutrient reserves will often be very difficult to exhaust and many typical fen species that are able to compete in an oligotrophic environment may not be able to establish for a long time after fertilisation has stopped (Hegg 1984; Klötzli 1991). In situations where intensive agricultural production was only of short duration, after successful soil exhaustion and the restoration of suitable hydrological conditions, the desired species composition may establish spontaneously from the soil seed bank (Putwain & Gillham 1990; Bellairs & Bell 1993; Maas & Schopp-Guth 1995; De Bruijn & Hofstra 1997). In most cases, however, species will have to be re-introduced with either commercial seed mixtures or by the superficial distribution of whole plants cut at seed maturity in nearby locations. Special attention has to be paid to correcting the water table in restoring former wetland sites that were used as agricultural land after drainage. A detailed hydrological analysis may be helpful in assessing the restoration prospects of degraded wetland areas (Grootjans & van Diggelen 1995; van Diggelen *et al.* 1995).

REMOVING EXCESS NUTRIENTS

The successful removal of excess nutrients from a soil depends on its texture, the sorption capacity, the volume of its pores and the depth of soil formation. With increasing contents of silt and clay the time required to exhaust a soil will also increase because of the ability of such soils to immobilise P and K. On a sandy soil dry matter yields within eight years had dropped by fifty percent (Oomes & Mooi 1985); in clay-rich soils, however, much longer periods may be required. Also in purely organic peat soils that are unable to accumulate P- or K-reserves, excessive nutrients can be removed as quickly as one to five years (Kapfer 1988). Standing biomass appears to be a better indicator for the degree of soil exhaustion than concentrations of N, P and K in the soil (Schiefer 1984; Oomes & Mooi 1985; Bakker 1989). Kapfer (1988) found that grassland soil could be considered to be exhausted as soon as the biomass falls below 3.5–4 t dry matter per hectare and year.

Increased frequency of cutting will obviously remove more nutrients from the soil than one cut per year (Oomes & Mooi 1985; Egloff 1986; Pfadenhauer *et al.* 1987; Kapfer 1988) however, it will weaken some of the main species in fen meadows, particularly species that flower late in the year such as *Molinia caerulea, Lysimachia vulgaris, Mentha aquatica, Succisa pratensis* and *Parnassia palustris*. Thus, in areas where such species are present a compromise must be found between the speed of soil exhaustion and the conservation of certain species (Kapfer 1988).

Unwanted nutrients can also be exported by removing the top soil (Beltman *et al.* 1996). Kapfer (1988) found that in comparison to frequent cutting, export of nutrients by scraping in peat soils did not accelerate the restoration process. Conversely, in sandy soils the removal of a layer of only 5 cm caused a soil exhaustion that could only be reached after a long period of cutting and removing the hay (Oomes & Mooi 1985). The depth of the layer to be removed depends on the nutrient concentration at different soil depths; scraping to a depth of *c.* 30 cm may be sufficient in most cases (Pfadenhauer 1991).

RAISING THE WATER TABLE

Water tables can be raised simply by backfilling drainage ditches or by the construction of dams, or in lake shore areas by raising the level of the lake (Klötzli 1988; Eggelsmann 1989; Pfadenhauer 1994). Raising of the water table is only suitable where large connected areas are available for restoration; shrubs and other woody plants are to be removed beforehand (Pfadenhauer 1991). In order to stimulate the formation of peat the ground water table should be raised up to the soil surface and fluctuations should be minimised (Pfadenhauer 1991). A 30-cm raise of the water table in the Reuss valley, Switzerland, led to the following transformations: Molinieta developed into small sedge meadows dominated by *Carex hostiana,* whereas original small-sedge meadows were transformed into tall-sedge meadows; communities originally dominated by tall sedges developed into reed belts (Phragmiteta) (Klötzli 1988).

A raise of the water table will also influence the physical soil properties and nutrient dynamics in peat soils. Schmidt (1995) found that 20 months after rewetting the surface of a fen peat soil of 4.4 m depth had risen by 18 cm, the ratio of large pores had increased by 2%, middle sized pores by 1%, and the hydraulic conductivity had increased from 0.23 to 0.30 m d^{-1}. While it is generally believed that in peat soils rewetting will lead to lower concentrations of plant-available nutrients due to reduced mineralization, evidence from field studies is controversial, as it has sometimes been observed that mineralization can occur under completely waterlogged conditions (Koerselman & Verhoeven 1995). While Meissner *et al.* (1995) reported significant correlations between the level of the water table and contents in nitrate and ammonia, Eschner & Liste (1995) found not only no reduction but even a slight increase of nitrate and ammonia after three years of rewetting. Berendse *et al.* (1991) compared mineralization at high and low water levels and found that mineralization was not very strongly reduced in the wet compartment (132 *vs.* 155 kg N ha^{-1}). Hauschild & Scheffer (1995) found in incubation experiments that the optimum for nitrate formation in calcareous peat soils was at 70% water capacity while the optimum for acidic soils was at 100%, suggesting that in acidic peat soils a raise of the water table will not necessarily lead to a reduction of the nitrate formation. An additional problem of raising the water table is that the availability of phosphates is expected to increase. Thus, it is indispensable that the most limiting nutrients are determined before restoration projects are planned (Koerselman & Verhoeven 1995). The issue of nutrient dynamics of fen soils in relation to the water table is also discussed by Grootjans & van Diggelen (1995) and references therein and by Bakker & Olff (1995).

An alternative to the raising of the water table is the lowering of the soil surface by removing a layer of top soil. The removal of top soil can be a suitable measure in isolated small areas where both an export of nutrients as well as a correction of the water table is required. Scraping is suitable only in situations where fluctuations of the water table are minimal (Pfadenhauer 1991). In order to prevent a re-introduction of nutrients the scraped top soil should not be accumulated in vicinity of the restored areas (Pfadenhauer 1987).

RE-INTRODUCTION OF SPECIES

After corrections of the nutrient conditions and the water table it is crucial for the restoration process that a site-specific species composition is developed. If the previous intensive agricultural use was less than ten years typical fen meadow species may still be present in the soil seed bank (Pfadenhauer *et al.* 1987; Maas 1988). Among these only species with a robust protective outer seed layer, e.g. *Juncus* and *Carex*, are likely to be activated after nutrient export and the raising of the water table. Species with other seed types, e.g. *Gentiana* spp., *Parnassia palustris*, *Pinguicula vulgaris*, *Primula farinosa* will retain their germination capacity for only a short time and therefore will often have to be re-introduced artificially (Maas 1988). Arguments against the artificial introduction of species, however, have been put forward on grounds of the danger of the mixing up of local varieties (Schönfelder 1980).

In situations where no commercial seeds are available fresh plant material may be cut in a nearby habitat and spread in thin layers at the restoration site. Cutting should occur during humid weather to impede the fall-out of seeds, and no time should be lost when the cut material is transported to the new site in order to prevent high temperatures caused by fermentation (Voser & Kobe 1995). Care should be taken to not apply the material too thickly, otherwise dormancy of the seeds, triggered by shortage of light, may be induced (Maas 1988); thus, the area cut and the area of destination should be of approximately the same size (Voser & Kobe 1995). In order to ensure proper germination the straw should be removed at the latest in spring of the following year. The content of seeds may not be optimal in such material because only some species will be ripe at the time of cutting others will already have fallen out. Alternatively, seed material could be collected from the fall-out at the bottom of hay stacks (Schiechtl 1973). Since typical straw meadow species prefer open conditions some superficial harrowing or ploughing of the soil may be beneficial when seed material is brought in. Finally, species can be transplanted individually or in groups, a rather laborious alternative (Klötzli 1981; Pfadenhauer 1987). Ideal time for sowing is in spring; transplantation is best done in winter (Voser & Kobe 1995).

Ombrogenous bogs

Ombrogenous bogs (raised bogs and blanket bogs) occurring predominantly in the cool temperate zones of Eurasia and America were once much more widespread than they are today. Historically, great losses occurred particularly in continental Europe and the British Isles through reclamation for agriculture and forestry whereas in present times commercial peat extraction is generally regarded as the greatest threat to residual areas (Money 1995). Currently, in the UK, peat production is estimated at 1.76 million m^3 annually (Bather & Miller 1991). In Germany and Switzerland, only *c.* 10% of the original bog area has remained unmodified (Grünig *et al.* 1986; Eggelsmann 1989). As ombrogenous bogs offer very special conditions many plant and animal species only survive in this particular environment, possibly owing to reduced competition. All but a limited number of species are excluded by the prevailing high water table, low pH, and the low concentrations of many essential nutrients. A number of species are relicts from times of glaciation (Göttlich & Kaule 1990). Ombrogenous bogs are also important in their hydrological function of water retention (Ingram 1983), a fact that is increasingly recognised in the United States where wetland restoration is discussed primarily as a measure of flood control (Hey & Philippi 1995). Finally, bogs serve as pollen archives that have gained recent significance by efforts to study climate change (e.g. Frenzel *et al.* 1991) and nutrient deposition in historic and prehistoric times (e.g. Görres 1991).

One of the main concerns in bog restoration projects is the creation of hydrological conditions suitable for the re-establishment of bog vegetation. In most bog remnants or cut-over bogs the water table is usually lower than in undisturbed habitats while fluctuations of the water table are mostly higher (Lavoie & Rochefort 1996). This may have been caused either by superficial drainage, by peat cutting in surrounding areas leading to lateral and downward water loss or by changes in the hydrophysical properties of exposed peat layers (e.g. lowering of water storage capacity). Another cause for low water tables may be the replacement of the original *Sphagnum* vegetation by a vegetation dominated by shrubs and grasses which will enhance water extraction from deeper peat layers (Schouwenaars 1995). Thus, the choice of appropriate restoration strategies will depend on proper identification of the hydrological problems involved.

The aspect of second importance in bog restoration relates to the re-establishment of the typical bog vegetation. Spontaneous re-establishment of *Sphagnum* vegetation in cut-over areas is usually poor. Success of re-establishment after correction of the water table often depends on the method of peat extraction: chances for regrowth of bog species are much better in areas where peat was extracted by block-cutting than where the method of surface milling was applied (Poschlod 1994; Money 1995). Particular problems are associated with the re-introduction of *Sphagnum* species on bare peat soils. Since the establishment of *Sphagnum* species in many situations holds the key for successful bog restoration environmental requirements and methods for its artificial introduction are currently investigated by a number of research teams (Poschlod 1994, 1995; Grosvernier *et al.* 1995; Money 1995; Rochefort *et al.* 1995).

REWETTING

The successful re-establishment of suitable hydrological conditions for bog restoration in many cases simply involves the blocking of drains. Whether this measure is sufficient depends partly on the hydraulic conductivity of the underlying strata (Pfadenhauer 1987; Schouwenaars 1995). For example, downward losses may be excessive if a sandy aquifer underlies the peat; in such situations, the water storage capacity of the (black) peat layer is crucial and a thickness at least 0.5 m of well humified peat is essential (Blankenburg 1994). In addition, a layer of "top-spit" (= incompletely humified surface layer or "acrotelm"; German *Weisstorf* or *Bunkerde*) of at least 30 cm thickness should overlay the black peat for additional water storage and to reduce evaporation. A functional top layer is mandatory because it will reduce water table fluctuations – a key requirement for successful colonization of bog species and one of the main problems to overcome in surface milled areas (Money 1995). Thus, in most cut-over bogs water storage capacity on the surface is strongly reduced and has to be compensated for artificially. Schouwenaars (1995) recommends the construction of "bunds", i.e. embankments allowing a shallow (*c.* 20 cm) inundation in winter or spring which will stabilise the water table and in addition prevent the growth of grasses and shrubs. A reduction of water table fluctuations can also be achieved by the creation or enlargement of open water bodies within the bog from where a constant water infiltration occurs. In situations where water is lost laterally, for example in bog remnants situated adjacent to cut-over bogs, it may be necessary to create a hydrological buffer zone, i.e. a zone surrounding the bog where the water table is kept relatively high (Schouwenaars 1995). In a situation where only a thin layer of black peat was left after peat extraction, flooding during winter (10–30 cm) favours the development of *Eriophorum vaginatum* and *Sphagnum cuspidatum* and leds to a decrease of *Molinia coerulea* (Nick 1993).

RE-ESTABLISHMENT OF BOG VEGETATION

Given favourable hydraulic conditions, a spontaneous re-establishment of a typical bog vegetation after peat extraction by block cutting has been observed in numerous cases including the formation of new peat (Poschlod 1994). In areas where peat is extracted by surface milling, however, the raise of the water table will often not be sufficient to stimulate the recolonization by bog vegetation because of high fluctuations of the water table (Beets 1993; Blankenburg 1993). Thus, removal and storage of the top spit prior to milling and its reapplication after the completion of peat extraction are likely to improve restoration results. Besides stabilising water table fluctuations this top spit also has the potential to provide an inoculum for certain important bog species. Poschlod (1995) showed that some *Sphagnum* species maintain diaspore banks persistent for several decades as long as the top spit is stored wet. Field and laboratory experiments showed that only the surface layer (0–10 cm) of a peat profile contained enough viable material to be of practical use as a source of diaspores (Campeau & Rochefort 1996).

When *Sphagnum* species were artificially re-introduced best results were achieved with fragments such as stems or branches but not with whole plants (Money 1995). The artificial introduction of vegetative parts of *Sphagnum magellanicum* on bare peat substratum, however, often fails due to the lack of nutrients. Rochefort *et al.* (1995) found that mineral fertiliser helped *Sphagnum* species to spread more rapidly. An alternative method is the creation of rafts floating in small flooded peat pits and ditches on which vegetative parts of *Sphagnum* plants are placed (Money 1995). This will help to overcome problems with water table fluctuations since *Sphagnum* plants in a raft are less vulnerable to drought as those growing on a solid peat surface. In setting priorities for bog restoration, *Sphagnum fallax* should be favoured as a pioneer to stimulate a rapid colonization and recovery of *Sphagnum* lawns on which other species, more characteristic of the ombrotrophic bog environment, can re-establish (Grosvernier *et al.* 1997).

TIME SCALES OF RECOVERY AND SUCCESSIONAL PATTERNS

Initial colonization of rewetted bare peat soils is often characterized by the establishment of *Eriophorum vaginatum, E. angustifolium, Erica tetralix* and *Rhynchospora alba* (Poschlod 1994; Mawby 1995; Joosten 1995). *Molinia coerulea* and *Calluna vulgaris* can also spread rapidly but are controlled by a long lasting high water table. Alternatively, Grosvernier *et al.* (1995) documented initial colonization of bare peat soils by *Polytrichum alpestre* which facilitates the subsequent establishment of *Sphagna* by providing a canopy structure with a favourable microclimate and prevents the formation of a hardened crust of peat. *Sphagnum cuspidatum* and *S. recurvum* are able to expand rapidly at the onset of succession possibly due to increased drought tolerance. The response of the late successional species *Sphagnum magellanicum* and *S. papillosum* to increases in the water table are slow (Joosten 1995; Mawby 1995; Money 1995). A slow response was also documented for *Andromeda polifolia* and *Drosera* spp. (Joosten 1995; Mawby 1995). While the re-establishment of bog vegetation – even under favourable hydrological conditions – may take at least several decades, a restoration of bogs as self-regulating landscapes after severe anthropogenic damage (including peat cutting) is impossible within a human time perspective, because the necessary colonization of typical bog species and renewed accumulation of peat require many centuries (Joosten 1995).

Riverine ecosystems

Ecosystems of undeveloped rivers are based on interactions between the main channel and adjacent low-velocity habitats during periods of overbank flooding. Spatial and temporal heterogeneity are created by erosion and deposition as the channel migrates back and forth across the floodplain. Thus, riverine ecosystems play a key role in the maintenance of regional biodiversity providing a rich variety of habitats for many rare and endangered plant and animal species (Naiman *et al.* 1989; Gallandat *et al.* 1993; Lachavanne 1993). In addition, riverine ecosystems carry out important hydrological functions, such as flood-peak reduction, ground water recharge and water quality improvement (Henry & Amoros 1995; Large & Petts 1996).

In industrialized countries most rivers have been confined to single channels with high flood velocities and extremely low levels of habitat diversity. The nationwide loss of riparian habitat in the USA is estimated to *c.* two thirds of the original area (Swift 1984). In Germany the situation is even more severe: only 10% of all creeks and rivers are still in a natural state (Eggers *et al.* 1991). This loss of habitat diversity is usually followed by drastic decline in animal and plant diversity. Compared to 300 species and 5000 individuals per m^2 in a natural creek shaded by alders Voser (1995) found only 50 animal species at a density of *c.* 1000 individuals per m^2 in an artificially channelled treeless creek. Channelization of the Kissemee River in Florida resulted in the loss of 14'000 ha of floodplain habitat and led to a severe damage of biological communities on all levels (vegetation, invertebrate, fish, wading bird, and waterfowl); for example, floodplain utilization by wintering waterfowl declined by 92% (Koebel 1995). Apart from the impact on biodiversity the disappearance of riverine wetlands by river embankment has also led to a loss of hydrological functions. Thus, in the United States several projects to restore large river systems are currently under way – not primarily motivated by goals of nature conservation but aiming at reducing flood hazards and the restoration of water quality (review in US Dept. of the Interior 1991).

Efforts to restore riverine wetlands are complicated by the hydrologic and sediment regimes that have been changed in most rivers, which make it impossible to return wetlands to their natural condition without massive removal of dams and altering of channelization (Henry & Amoros 1995). Restoration of river systems must also be balanced against other human interests such as safety against flooding and the requirements of agriculture. In addition, financial needs may be prohibitive, since expansions of riverine wetlands in industrialised countries will in most cases require the purchase of privately owned land (Zollinger 1995; Stalder 1995). Concepts of restoration for many individual rivers and documentation on historic conditions of the river system are often unpublished but will be available from hydroelectric power companies involved on request.

When restoring riverine systems the following components should be considered:

1. Meander restoration: to break up the monotony of straight embankments the river should be allowed to meander in a winding course particularly where the flowing speed is slow; this will increase the variety of flow conditions and habitats. Meanders can be recreated by letting the river find its new course on its own by lowering or removing embankments. Alternatively, a new river bed can be constructed in a meandering way by technical means (Schlüter 1992). Spontaneous meandering will take time but will reflect the natural dynamics of the river; artificial meanders should take the original (pre-embankment) course of the river into account (perhaps available from old maps or old aerial photographs; Eggers *et al.* 1991). After restoration sections of the old (embanked) course that are no longer serving as river bed should be preserved as still water bodies.
2. River bed design: Recreation of meanders involves a reduction of the slope leading to reduction of the speed of a river. In turn this will reduce the danger of river bed erosion (Schlüter 1992). Thus, weirs, sills or ramps may not be necessary now but when installed should not hinder the up- and downstream migration of animals. Where blockage of migratory fish is of concern fish passageways should be incorporated (Schnick *et al.* 1982). A re-elevation of river bed bottoms will in most cases occur naturally through the reduction of the flowing speed which will lead to increased deposition of sediments. In the cross-section a new river bed should not be designed too large in order to allow the river to leave its bed at times of high floods. Generally, the more variable the design the better, e.g. symmetrical sections should alter with asymmetrical ones. For example, stream bank protection structures made of a wide range of stone sizes create more diverse habitat than do those made of uniform concrete blocks (Wolf 1977; Gore & Shields 1995).
3. Vegetation: Restored areas can be left to natural succession, particularly when the river is allowed to choose its course. If planting is considered at all it should be restricted to small clumps triggering initial colonization. Natural zonation should also be taken into account (Schlüter 1992; for the rivers Glatt and Reuss: Klötzli 1991).
4. Habitat elements: the need of resting places which provide protection for fish from high current velocities or predation should also be considered in restoration projects. The

reduction or loss of cover in river ecosystems may reduce fish populations by up to 80% (Wesche 1985). Overhanging vegetation, undercut banks, submerged vegetation, submerged objects (e.g. logs, roots, boulders, and cobble), floating debris, and turbulence in the water are examples of habitat elements that provide refuge for fish (Gore & Shields 1995).

5. River margin management: Where no large-scale restoration measures are possible the protection, rehabilitation and restoration of river margins is a valuable alternative. The decision to include river margins in river management is based on the four key functions: water quality, nature conservation, instream habitat enhancement, and recreation (Large & Petts 1996). In temperate situations it has been clearly demonstrated that riparian buffer zones with permanent vegetation can significantly reduce the concentration of nutrients (80–90% for phosphates and nitrogen) in surface water and groundwater entering streams (Howard-Williams *et al.* 1986; Cooke & Cooper 1988; Knauer 1990; Fustec *et al.* 1991; Schreiber 1994). Specifically, nutrients originating from adjacent intensively managed arable land that are transported by surface runoff or ground water will effectively be filtered by buffer strips (Kickuth 1970; Haycock & Burt 1990, 1991). The width of such buffer strips to achieve the function of water quality control varies widely: a range of 10–80 m on both sides of the river channel appears to be appropriate in most situations (Phillips 1989; Schreiber 1994; Large & Petts 1996).

The greatest effort to restore a riverine system motivated by ecosystem conservation is made in the Kissimee River restoration project in central Florida (Restoration Ecology 1995 (3), multiple authors). The historic Kissimee River meandered blindly for *c.* 166 km within a 1.5 to 3 km-wide floodplain. Between 1962 and 1971, a 90 km wide canal was dredged through the river/floodplain system leading to a drainage of 2/3 of the floodplain wetlands and an accompanied loss of biodiversity and ecosystem functions. To undo these impacts 35 km of the canal will be backfilled and some of the water control structures eliminated. Original river channel eliminated by the excavation of the canal will be re-excavated and connected to existing remnant river channels. The planned 15-year restoration project will result in a return to a more natural condition of about 70 km of river channel and 11'000 ha of wetland. An evaluation programme considering hydrological, biological and ecological attributes will measure the success of the restoration efforts (Koebel 1995).

A meander restoration project is also considered for the Danube River in Germany (Kern 1992; Gore & Shields 1995). Detailed plans for restoring two meanders include the provision of gently sloping rock drop structures that are to divert base flows into the old channel yet allow high flows to use the present (straightened) channel. Natural floodplain habitats are to be restored, and the purchase of a 100-meter strip of land (the predicted maximum meander belt width) along concave banks is to allow unrestricted bank erosion in order to restore natural channel cross-section and bed morphology.

The revitalisation of the Ise River and its tributaries in northern Germany involves no restoration of meanders but concentrates mainly on reducing the nutrient loads in the river water by converting adjacent arable fields into hay meadows and pastures of low productivity (Reuther *et al.* 1993). To reduce water temperatures by shading shrubs and trees are planted along both sides of the river on a length of *c.* 18 km. Additionally, any obstacles that impede the migration of fish and other organisms will be removed from the course of the river.

The restoration of a section of the Glatt River near Zurich, Switzerland, allowed the river to leave its straight channel. Gravelly islands, new meandering branches and flood plains were created on a total area of 8 ha (Voser & Kobe 1995). Within 15 years a vigorous flood plain forest developed and a number of endangered insects, amphibians, reptiles (e.g. *Natrix natrix, Lacerta agilis*), bird species (e.g. *Charadrius dubius, Picus viridis*), and bats now sustain viable populations in the area.

TIME SCALES OF RECOVERY AND SUCCESSIONAL PATTERNS

Compared to other systems riverine habitats can regain their functions in relatively short time, often within a few years (Voser 1995). According to Gore & Milner (1990) recovery times may vary between 10 days and 25 years, depending on the channel condition (entirely destroyed, only reach destroyed, species abundance reduced in reach), source of colonists (none, hyporheic, upstream and downstream), and recovery patterns (presence or absence of organic substrates). Assuming that water quality problems have been mitigated and that habitat quality has been enhanced in a disturbed area, the rate of recovery is dependent upon the availability and location of a source of colonizing organisms. Recolonization of reclaimed and rehabilitated river channels appears to follow a deterministic pattern of colonists (Gore & Shields 1995): upon development of a biofilm, periphyton, especially diatoms, colonize new substrate rapidly (in some cases in less than ten days) followed by invertebrate grazers and collectors that are able to use periphyton and accumulating organic particulates. Finally, predatory invertebrates arrive. Forage fish arrive after sufficient numbers of invertebrate invade to form a food source, and will finally top carnivores. Establishment of a more natural fish community structure may take several years to match pre-disturbance conditions.

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