11 Fen Management and Research Perspectives: An Overview

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11.1 Introduction

A fen has vegetation that is actively forming peat and is fed by ground- or surface water (Joosten and Clarke 2002). In Europe a "fen meadow" is a ground- or surface water-fed mown grassland that does not form peat, since it was formed after modest drainage of a fen or it developed on a predominantly moist soil (Grootjans and Van Diggelen 1995). Therefore, fens and fen meadows are considered to be different ecosystems by most European authors. Others do not make a distinction between fens and fen meadows because the species composition of both ecosystems may overlap considerably (Wheeler et al. 1995). In North America, fens dominated by tussock-forming sedges are referred to as "sedge meadows", which are often grazed. Since there is not yet scientific agreement on whether sedge meadows are fens or fen meadows, we will refer to them as fens in this chapter.

Fens are the most diverse temperate community types (Bedford and Godwin 2003). The biodiversity of fens is threatened by agricultural and urban development, fragmentation, hydrologic changes, shifts in grazing and mowing, structural changes in vegetation, and reproductive problems related to isolated populations. Species are declining in numbers in fens in Europe and North America. Estimates of species lost, for example, from the United Kingdom, are that 95–98 % of species-rich fens present before 1940 have been lost (García 1992). Similar losses are reported for France (Muller et al. 1998) and the Netherlands (Jansen et al. 2000).

The fragmentation of fens occurs via agricultural and urban development; and as these encroach, the wetlands become smaller and more isolated because of hydrological modification (Figs. 11.1, 11.2). In particular, any alteration in surface water flow from streams and rivers creates a barrier to seed dispersal (Galatowitsch and van der Valk 1996; Middleton 1999). Species loss

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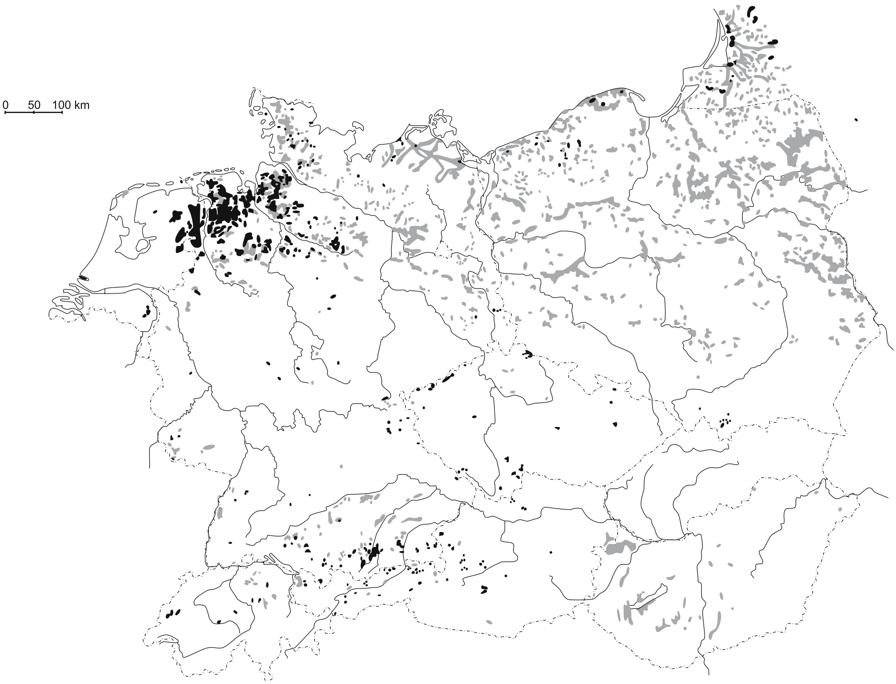


Fig. 11.1 Mire distribution in Central Europe under natural conditions (Succow 1988; reproduced by permission of Fischer Verlag, Jena). Only peatlands exceeding 300 ha have been included. *Black* 3D bogs, *gray* 3D fens

occurs because of the lack of habitat (Bedford and Godwin 2003), but also because of reproductive problems associated with small population sizes (Ellstrand and Elam 1993; Fischer and Stöcklin 1997). While these problems might be alleviated by the reconnection of dispersal corridors (Taylor et al. 1993), higher amounts of dispersal between these disconnected wetlands could contribute to more opportunities for an invasion of exotic species in North America, where invasive species in wetlands are a major problem (Galatowitsch et al. 1999).

Water quality and sedimentation patterns can be altered by agricultural and urban development; and these changes contribute to shifts in species composition in fens. Sedimentation is on the increase in fens adjacent to cities in the Midwestern United States; and this encourages invasive species (*Phalaris arundinacea*, *Typha* x *glauca*) at the expense of *Carex stricta* (Werner and Zedler 2002). Urban water runoff increases the amounts of N and P in the surface water, and subsequently, the biomass of *T.* x *glauca* increases. *C. stricta* does not increase in biomass, so that fens with urban runoff become dominated by *T.* x *glauca* (Woo and Zedler 2002). The nutrient dynamics in fens changed with land use changes. Atmospheric nitrogen

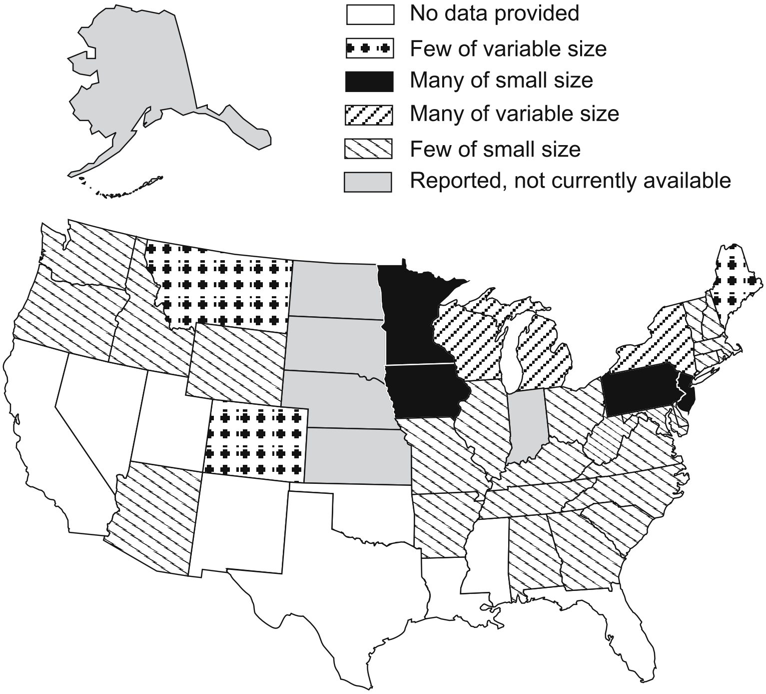


Fig. 11.2 Distribution of North American fens (Bedford and Godwin 2003; reproduced by permission of Society of Wetland Scientists)

deposition, in particular, increased 3- to 10-fold during the past decades in these wetlands (Olde Venterink et al. 2001, 2002). Eutrophication causes increases in the plant biomass of certain species and contributes to the encroachment of large herbaceous and woody species (Bobbink and Roelofs 1995).

11.2 Hydrological Systems of Fens

Fens with similar vegetation types exist in very different landscapes and soil conditions. Small sedge vegetation with many Red List species (*Caricion davallianae*, i.e., species of alkaline fens), for instance, occur in coastal dune slacks on sandy soils (Ranwell 1959), in brook valleys of the European lowlands on peat soils (Succow and Joosten 2001), but also in calcareous spring mires in alpine regions of Europe (Ellenberg 1986).

Hydrological systems can create similar habitats within a wide range of landscapes. Groundwater fed fens are "flow through" systems, in which nutri-

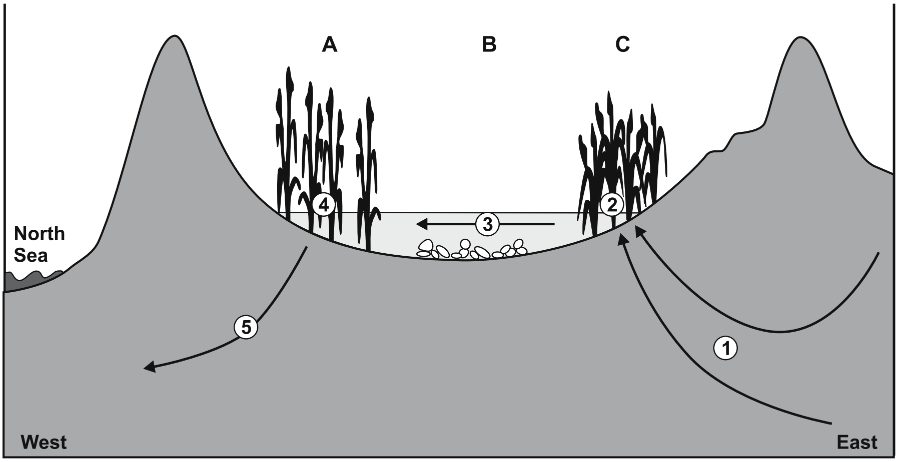


Fig. 11.3 Schematic presentation of a through-flow dune slack on the Dutch Wadden Sea island. *1* Incoming calcium and iron-rich groundwater, *2* exfiltration of groundwater in a vegetation zone with sedges, *3* surface flow and precipitation of iron and $CaCO_3$ in a zone where nutrient-poor pioneer vegetation persists, *4* infiltration of iron- and calcium-poor surface water in a *Phragmites australis* stand, *5* sulfate reduction during infiltration (after Adema 2002; reproduced by permission of Opulus Press)

ents become less available for plant growth due to geochemical changes along the flow paths of the groundwater (Fig. 11.3).

11.2.1 Large and Small Hydrological Systems

Much research has been done on the hydrological conditions of fen systems (Komer 1994; Wassen et al. 1996; Van Diggelen 1998; Jansen et al. 2000; Schot et al. 2004). Some authors present very detailed hydrological research in species-rich fens on mineral soils, a case in which most of the groundwater comes from a very small catchment area (Jansen et al. 2000), while others worked in very large mire systems, like the Biebrza in Poland (Wassen et al. 1996) and the Ob River in Russia (Loos and Schipper 2003; Fig. 11.4). Sometimes the surface area of a fen is rather small, but the catchment area supplying the fen with groundwater is very large. This is often the case with fens and spring mires that are supplied with artesian water, originating from large aquifers in which the water flow has been blocked due to shifting of geological strata (Van Andel and Arondson 2005). Fens are very vulnerable to erosion by groundwater flow. The digging of drainage ditches may result in the down-cutting of the ditch channel, which effectively lowers the local water table within the fen. The peat may begin to decompose and erode into the ditch. After the ditch is established, the degradation process may be very difficult to

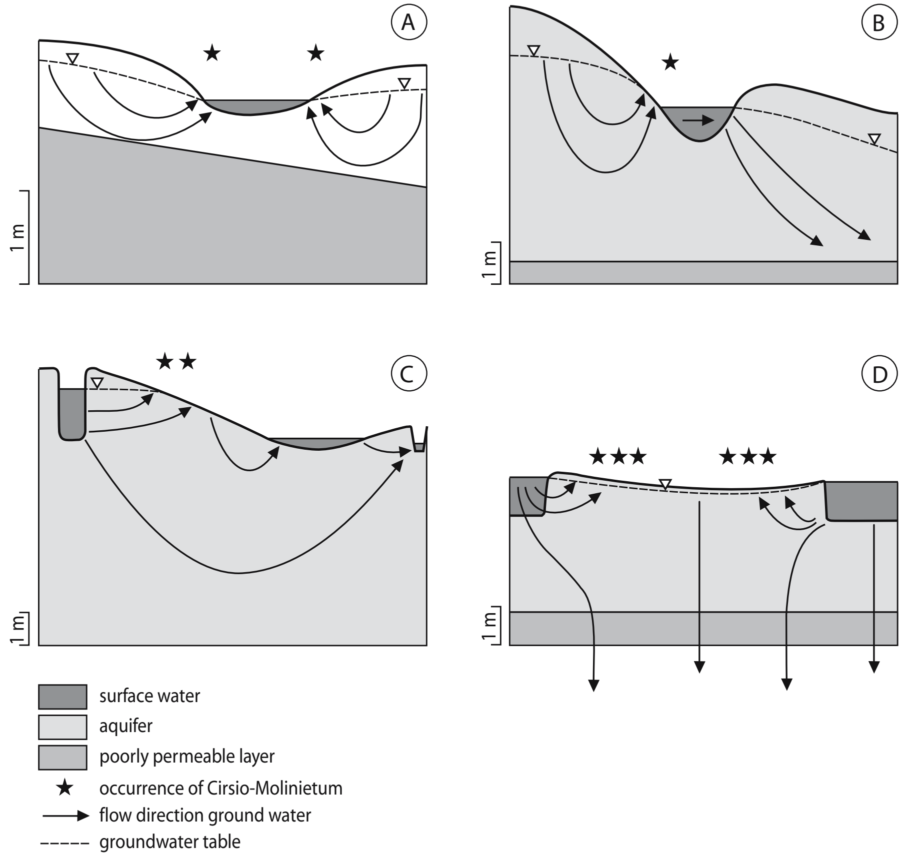


Fig. 11.4 The fen meadow community *Cirsio–Molinietum* (indicated by stars) occurs in the Netherlands under various hydrological conditions. **A** Heathland pool situated above shallow loam layer with groundwater supply from local sand ridges, **B** local (small) hydrological system with through flow of groundwater, **C** artificial hydrological system with lateral discharge of calcareous surface water from large canals, **D** surface water system with discharge of calcareous surface water from ditches (modified after Jansen et al. 2000; reproduced by permission of Opulus Press)

contain, and may result in the total destruction of the mire (Wolejko et al. 1994). Due to rapid peat mineralization, these systems become very eutrophic and tall herbs (such as *Urtica dioica*), shrubs, and trees move in quickly (Wolejko et al. 1994).

11.2.2 Natural Fens Can Be Very Stable

Fens are mires that receive water from the mineral surroundings and hence respond to changes in regional hydrology. Large fens, however, also raise their water level and that of their surroundings autogenously, leading to the expansion of fens on progressively higher positions in the landscape. Such rising water levels may also result in the gradual paludification of hitherto dry upland depressions and may even reverse water flow from groundwater recharge to groundwater discharge conditions (Succow and Joosten 2001).

Due to their thick and rather flexible peat layer, such fens became increasingly more resilient during the course of their long-term development and remained in a more or less stable situation for several centuries, despite considerable climatological and hydrologic changes. They only changed character when certain hydrological thresholds were crossed, the exact magnitude of these thresholds being dependent on the stage of mire development. Schult (2002) carried out detailed palynological research in the Trebel area and found that between 8000 and 7000 years BP, a terrestrialization mire developed in a calcareous lake, with species such as *Menyanthes trifoliata*, *Cladium mariscus*, and *Carex rostrata*. At this stage, small amounts of $CaCO_3$ (travertine) were deposited. Between 7000 BP and 5000 BP, a groundwater-fed fen developed on top of this young mire, with predominantly moss species, such as *Drepanocladus* spp, *Homalothecium nitens*, and *Meesia triquetra*. *Menyan-*

	1	2	4	5	6	7	8	9	12
Carex paniculata	+	+	+	+		+	4		+
Eleocharis quinqueflora					3		2b		
Primula farinosa		+	2b	2b	1	3		1	
Carex davalliana		1	3	5	5	4		3	
Drepanocladus cossonii		4	3			2b		2a	
Homalothecium nitens	+	2a	1	+		+		+	1
Valeriana simplicifolia			+	+	2a	1		+	
Alnus incana	3								3

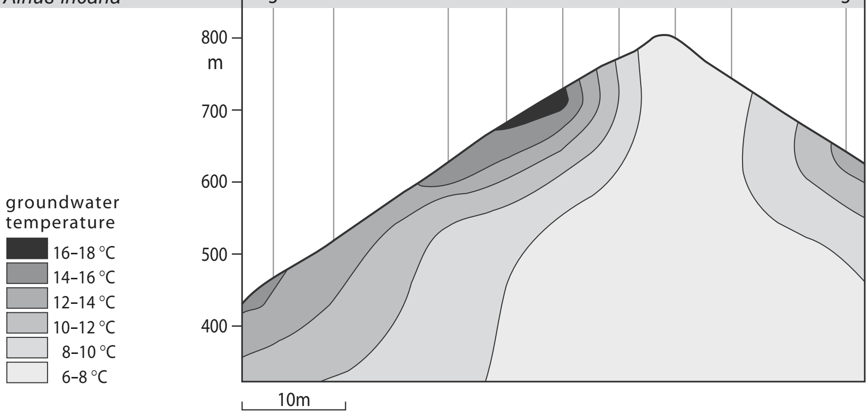


Fig. 11.5 Temperature profile and vegetation composition across a small Slovak spring mire near the High Tatra Mountains. The species coverage was estimated using a Braun–Blanquet cover/abundance scale (Grootjans, unpublished data)

thes trifoliata remained present throughout this period and small sedges such as *C. diandra* and *C. limosa* also occurred. Chalk deposition occurred until ca. 3000 BP and then stopped. This developmental sequence was remarkably stable; and similar results have been found in Western Germany (Schwaar 1980) and Poland (Wassen and Joosten 1996).

Nowadays such vegetation types are very rare and chalk deposition is almost no longer found in the North German lowlands. However, remnants of these vegetation types can still be found in the mountain areas of Bavaria (Germany) and Slovakia. Here we find relatively small calcareous spring mires, where travertine is still being formed in pools with *Drepanocladus* spp, *Menyanthes trifoliata*, and *Chara* spp. This travertine formation is a rather recent phenomenon in some wetlands and mainly occurs in oligotrophic (calcareous) pools where supersaturated groundwater discharges (Grootjans et al. 2005). During summer the surface water of these pools is warmed (Fig. 11.5). Outgassing occurs and $CaCO_3$ precipitates. Especially moss species and some water plants, such as *Chara* spp, can accelerate this travertine formation by using the high concentration of CO_2 in the pools as a carbon source. $CaCO_3$ precipitates on the leaves and is later added to the soil (Van Breemen and Buurman 2002). The added travertine prevents acidification and associated eutrophication in the soil and sustains a very high biodiversity in such fens. However, such small groundwater-fed systems are vulnerable to hydrological changes in the landscape and also to climatic change. These small groundwater-fed fens can easily shift from peat-forming systems toward eroding stages within a short period of time (Wolejko et al. 1994).

11.2.3 Hydrochemical Processes Stabilizing the Biodiversity of Fens

Formerly, hydrological systems that provided natural fens with a large supply of base-rich groundwater were able to stabilize nutrient poor fen vegetation for many centuries, without any management by man. However, during the past few centuries, almost all fens have been slightly drained and changed into low-productive meadows and pastures that cannot be maintained without management (Kotowski 2002). When large hydrological changes in the surrounding landscape occur, the through-flow of anaerobic groundwater should be re-established to restore the high biodiversity of fen species. A relatively slow groundwater flow in fen systems is essential for maintaining a high biodiversity. Slow-flowing groundwater prevents erosion and stimulates the precipitation of iron and $CaCO_3$ in the root zone, thus stabilizing nutrient cycling at a low level (Lamers et al. 2002; Olde Venterink et al. 2001, 2002). The study of historical descriptions and well preserved remnants of fen systems may prove very helpful in understanding why some systems can easily maintain a high biodiversity and why other cannot perform this function.

11.3 Eutrophication in Fens

Eutrophication is considered one of the main factors contributing to the loss of biodiversity in fens in Western Europe, because increased nutrients are related to increased productivity and the competitive exclusion of shorter species. Species-rich communities of low productivity are eventually transformed into communities of high productivity dominated by a few tall sedges, grasses or shrubs (Grime 1979). This increased dominance of tall herbaceous and woody species occurred both in Europe and North America, not only due to eutrophication, but also because of changes in rural land usage related to grazing (Middleton 2002a) and drainage (Wheeler et al. 1995; Grootjans et al. 1996).

11.3.1 Change in Management

After 1945, agriculture shifted away from small farms and moved toward larger mechanized ones that no longer utilized fens for pastures (Middleton 2002a) or hay-mowing (García 1992). After that, the short diverse vegetation shifted toward communities dominated by woody and tall herbaceous species. These changes in vegetation structure are directly attributable to the cessation of cattle grazing and mowing. Especially after moderate to heavy grazing, shrubs and trees grow and shade the rarer fen species (Middleton 2002a, b).

Strategies for dealing with these changes in community structure and related losses in biodiversity have taken different routes in Europe and North America. Managers in North America control the dominance of woody and tall species with fire, whereas European managers rely on mechanical mowing and hay removal (Schrauzer et al.1996). Fire temporarily increases the species richness, flowering and seed set of fen species in North America (Middleton 2002b). Because eutrophication is a contributing factor in the growth of these large fen species, fire may also be useful because it removes nutrients such as nitrogen from the system. Similarly, hay removal also removes nutrients from these systems, which eventually may lead to phosphorus limitation, and is therefore of value in maintaining the biodiversity of fens (Verhoeven et al. 1996; Tallowin and Smith 2001).

11.3.2 Change in Nutrient Budgets

The increase in productivity in fens is the result of increased availabilities of potentially growth-limiting nutrients such as nitrogen (N), phosphorus (P), and potassium (K; Verhoeven et al. 1996; Van Duren and Pegtel 2000). To

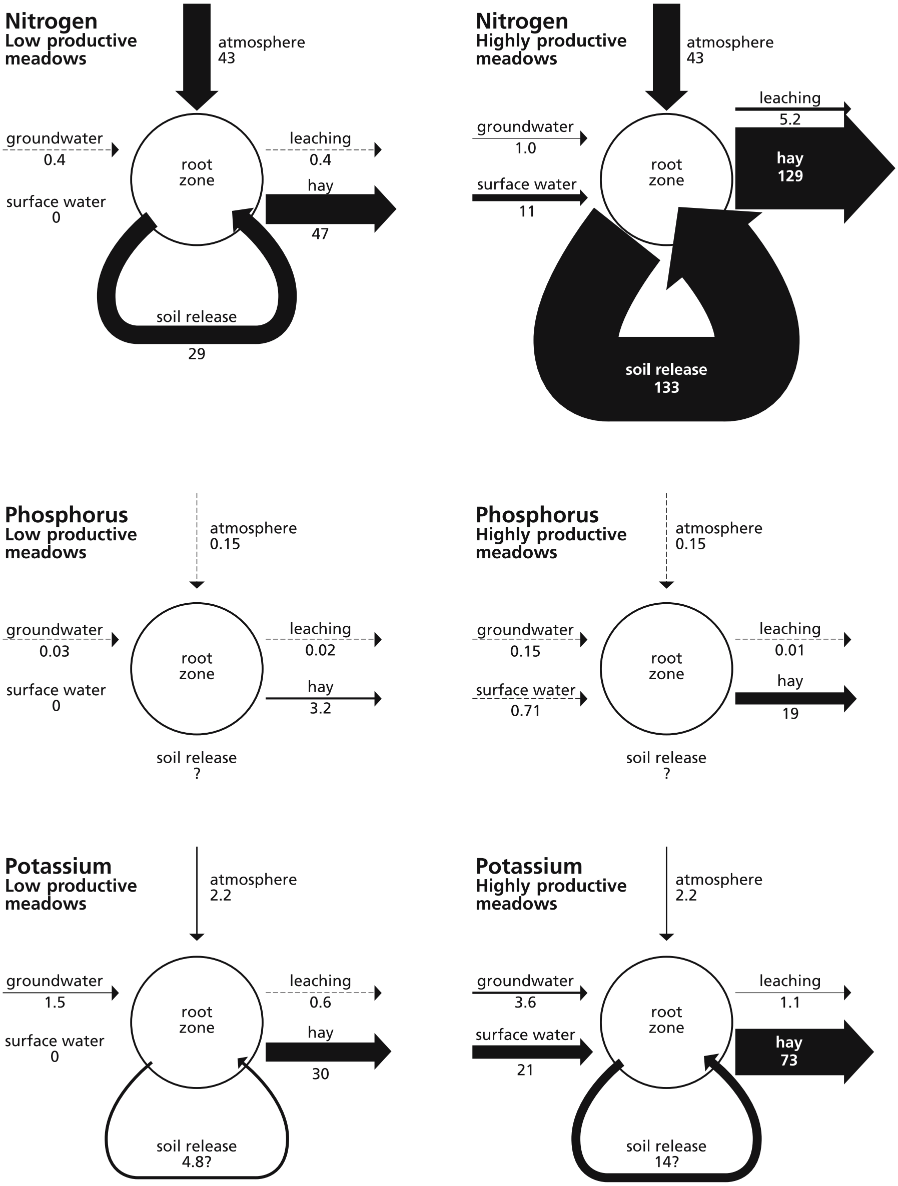


Fig. 11.6 Annual N, P, and K fluxes into, out of, and within the root zone of low-productive species-rich meadows and highly productive species-poor meadows. All fluxes are in kg N, P, or K per hectare per year (Olde Venterink et al. 2002; reproduced by permission of the Ecological Society of America)

understand the causes of nutrient enrichment and evaluate the effects of management, one should have insight into annual nutrient budgets, i.e., the rates of nutrient fluxes into and from external sources as well as net release rates of nutrients in the soil (Koerselman et al. 1990).

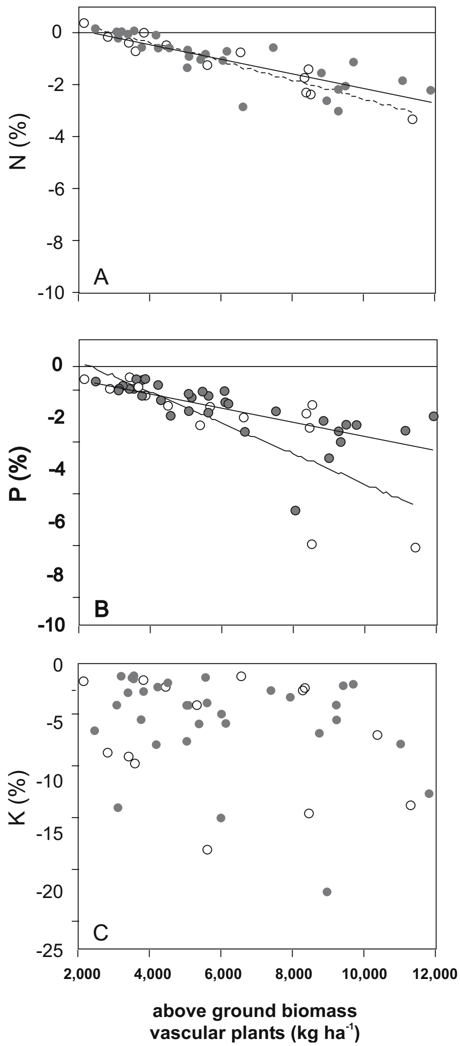
Olde Venterink et al. (2001, 2002) quantified N, P, and K budgets along productivity gradients in fens to evaluate the importance of atmospheric deposition, flooding, groundwater flow, leaching, and soil release rates. Differences in N availability along productivity gradients were caused by release rates in the soil (mineralization; Fig. 11.6). Besides N release in the soil, atmospheric N deposition made up large proportions of the annual N budgets. P and K availabilities along the productivity gradients were mainly influenced by soil processes, as indicated by soil extractable nutrient. The input and output fluxes of nutrients by groundwater flow were very low, while the input by flooding seemed only important for K (Fig. 11.6). However, it should be noted that the input of nutrients adsorbed to sediment was not included in the flooding assessment, which could be a very important input source of P in fens (Mitsch et al. 1979). The dominant role of soil processes in setting nutrient availabilities along the productivity gradients demonstrated that human alterations of site conditions, such as drainage, have been a major cause of nutrient enrichment for N, P, and K (Olde Venterink et al. 2002). Furthermore, considering that atmospheric N deposition has increased from 5–20 kg N ha^{-1} $year^{-1}$ (Erisman and Draaijers 1995) to an average 43 kg N ha^{-1} $year^{-1}$ in this part of Europe (Fig. 11.6), increased atmospheric N deposition also has been a major source of nutrient enrichment.

The long-term effects of annual hay-harvesting on nutrient limitation were also assessed by Olde Venterink et al. (2002). In fens with low productivity, N output by hay-harvesting accounted for N input from atmospheric deposition, whereas there was a net output of P and K (Fig. 11.7; Olde Ventervink et al. 2002). At highly productive sites, hay-harvesting resulted in net output of N, P, and K (Koerselman et al. 1990). When the annual nutrient output was expressed as percentages of the total soil pools, it was found that net output of K (1–20% of soil K pool) was mostly much larger than that of P (generally 0.5–3.0%) or N (0–3%). Hence, in the long-term, hay-harvesting seems to induce K limitation, particularly when they are drained for a long time (Van Duren et al. 1981; Van Duren and Van Andel 1997).

11.3.3 Internal Eutrophication

More intensive agricultural use not only leads to lowered water tables but also to increased concentrations of NO_3^- and SO_4^{2-} in ground- and surface water. Effects of water table alterations receive much attention, but there is confusion about its consequences for nutrient kinetics. Desiccation often leads to eutrophication in peatlands (Grootjans et al. 1996). However, this is not always

Fig. 11.7 Net annual output of N, P, and K by hay-harvesting as a percentage of the total N, P, and K pools in the soil (0–10 cm), in relation to the above-ground biomass of vascular plants in fens (*open circles*) and meadows (*filled circles*) (Olde Venterink et al. 2002; reproduced by permission of the Ecological Society of America)



true. Fens that are limited by P and K, in particular, usually respond to drainage with a decreased productivity. This had already been noticed by Russian researchers in the early 1930s during the communist period, when extensive mires were reclaimed for agriculture. They found a marked decrease in groundwater-fed fens after drainage (Ozushko 1934, cited in P'yavchenko 1976). Ozushko warned: "that money invested in drainage, is wasted unless the drained area is then used for the cultivation of agricultural crops". At that time, Russian agro-scientists already knew that P and K were limiting in many drained mires and not N.

Restoring desiccated fens is not simply a matter of increasing water levels. Recent research shows that rewetting with sulfate-rich surface water, in particular, may lead to increased eutrophication in fens. High SO_4^{2-} availability can induce PO_4^{3-} mobilization in wetland soils. This may occur when the former discharge of iron-rich groundwater is not restored. In such a case, the groundwater is replaced by sulfate-rich surface water. When the water stagnates, all oxygen is consumed and sulfate reduction is stimulated when the outside temperatures are sufficiently high. Sulfate reduction produces sulfides, which in the presence of iron form FeS_x (pyrite). PO_4^{3-} bound to iron hydroxides is then released to the pore water or the stagnating water body (Smolders et al. 1995; Lamers et al. 1998).

Although the active manipulation of the internal mobilization of nutrients has been practiced since medieval times by farmers trying to improve their crop yield by regular flooding with surface water, present biogeochemical research in peatlands often overlooks the alterations in nutrient mobilization rates due to such anthropogenic water quality changes. Flooding with surface water during the summer, in particular, can be very harmful for species-rich fens and may result in the rapid growth of tall grasses, such as *Phalaris arundinacea* or *Glyceria maxima*. Periodic local desiccation of the sediment during the summer, which is a natural phenomenon in fens, is important in sulfate-rich wetlands because this determines the ability of the soil to retain phosphates. The oxidation of FeS_x again increases the reactive Fe^{3+} concentration, stimulating P-binding in the top layer (Lucassen et al. 2004a).

The role of increased NO_3^- levels in the groundwater due to intensive agricultural fertilization is complex. Nitrate may increase the redox potential but also leads to a higher binding capacity of P due to FeS oxidation. A high input of nitrate inhibits SO_4^{2-} reduction because it is a more favorable electron acceptor and also the oxidation of FeS_x will increase the sulfate levels. So, high nitrate levels in the groundwater may counteract the negative effects of high sulfate concentrations. On the one hand, NO_3^- leaching into the groundwater increases the risk of eutrophication of groundwater discharge areas by mobilizing SO_4^{2-}, in FeS_x-containing soils or aquifers. On the other hand, high concentrations of NO_3^- may prevent P eutrophication (Lucassen et al. 2004b). As a rule of thumb, to avoid eutrophication due to internal release of nutrient in fens, one should not increase surface water tables above the potential groundwater table.

11.4 Seed Bank and Seed Dispersal

An increase in species richness over time following restoration depends on the environmental site conditions and on seed availability, germination, and survival. Fine-tuning of the environmental conditions for restoration is already a difficult task, but very often seed availability and also seed dispersal are restricting restoration prospects considerably (Bakker et al. 1996).

When degeneration processes have been taken place for a long time, seed banks might have been depleted and ways to introduction of locally extinct species have to be considered. Sometimes lack of seed availability can be improved by stimulating natural dispersal processes, such as flooding (Middleton 1999).

11.4.1 Seed Banks

Wet grassland species have a limited ability to regenerate by seed, because these species usually have only transient seed banks (Bekker et al. 1997, 2000) and seeds of rare and endangered species also have a comparatively low persistence (Fischer and Stöcklin 1999). In addition, seed dispersal distances are usually low (Bullock et al. 2002; Soons 2004) and thus it can be questioned whether seed persistence and dispersal significantly contribute to the restoration of species-rich wet grassland communities.

Jensen (1998) analyzed seed banks and the local seed rain of abandoned wet grasslands and found that the (anemochorous) seed rain of the investigated sites was indeed dominated by species of the aboveground vegetation, implying that long-distance seed dispersal by wind is a rather rare event in this habitat type. In contrast, the seed bank of late successional stages, in which most typical wet grassland species had disappeared in the vegetation, always contained some elements of the typical wet grassland flora. These results at least partly contradict the hypotheses of transient seed banks of most grassland species. Jensen's research group, therefore, carried out seed burial experiments to examine directly the seed mortality and longevity of approximately 45 species of the regional fen flora (Schütz 1997, 1998, 1999; Jensen 2004a, b; Brändel 2004a, b). These studies revealed that the mortality of artificially buried seeds of most species was low, that only a few species exhibited transient seed banks, and that the seeds of almost half of the investigated species were able to persist in the soil for at least five years (i.e., these seed banks were long-term and persistent). To explain such differences in seed longevity estimates, Jensen (2004b) compared the seed-longevity estimates for a set of 230 herbaceous wetland species from Northern Germany using different methods of seed bank analysis (e.g., burial experiments vs classic soil seed bank sampling) and various (indirect) indicators for seed bank classification (presence/absence in vegetation and seed bank, depth distribution in the soil; see Thompson et al. 1997). This analysis showed that burial experiments generally resulted in the highest longevity estimates. The criterion stated as "presence in the vegetation, but absence in the seed bank" is the criterion used by Thompson et al. (1997) to classify a species as "transient". However, many species classified as transients in this way had unrealistically low longevity estimates. From these results, it can be concluded that many typical wet grassland species are able to persist at least some years as viable seeds in the soil. We may conclude that much more experimental evidence is needed to develop realistic criteria for estimating seed longevity of soil seed banks.

11.4.2 Seed Dispersal

Water is an important dispersal vector and a number of studies have quantified hydrochorous seed dispersal in wetlands, such as clear streams in an alpine zone (Bill et al. 1999), forested temperate wetlands (Middleton 2000), boreal rivers (Andersson and Nilsson 2002), and a small lowland river (Boedeltje et al. 2003). All these studies revealed that running water might disperse seeds over quite long distances. Vogt et al. (2004) analyzed the hydrochorous seed dispersal in a fen in Northern Germany (Eider Valley) by means of seed traps. This two-year study revealed very large temporal and spatial variation in both quantity (seed number) and quality (species composition) of the hydrochorous seed transport. Overall, seeds of almost 200 different taxa were found in the traps, which is over 60 % of the regional species pool. Seed transport was dominated by (common) species which were abundant along the river shores, but rare and regionally endangered species were also present, albeit only in low quantities. Wind dispersal may also be important in these wetlands and human effects on landscapes, including habitat fragmentation and eutrophication, may affect the colonization capacity of species (Soons et al. 2004).

11.5 Fen Restoration: An Example From Hungary

11.5.1 Introduction

Restoration provides a way to offset some habitat loss for the rare species of fens and other wetland habitats (Middleton 1999; Grootjans et al. 2002; Lamers et al. 2002; Wheeler et al. 2002), particularly because more than 50 % of the world's wetlands have been lost (Munro 1988). Few specific estimates of loss for fens have been made in North America, but in some regions of the United States fen loss exceeds 40 % (Bedford and Godwin 2003). In Europe, less than 1 % of the original mire area has remained in most countries (Joosten and Clarke 2002), although some countries have some floodplain wetlands that are nearly natural (e.g., Turkey, Romania, France; Wenger et al. 1990). In eastern European countries, some 10–50 % of the former mire area still remains. In some Scandinavian countries, European Russia, and Romania, the mires are the best preserved (>50 % of the mires remaining). On a world scale, mire losses are highest in Europe due to its high population pressure on nature and the climatic suitability for agriculture and forestry. In Hungary, in particular, wetland losses are large. It is estimated that 97 % of the natural wetlands have been drained and converted to agricultural fields (Lájer 1998); and many of the remaining wetlands are damaged. While the restoration of fens may be a way to offset these worldwide losses of fens and related biodiversity loss, few examples exist of successful fen restoration, so this example from Hungary is of interest to our discussion.

11.5.2 Destruction and Restoration of a Fen System in Hungary

Fen restoration is less commonly attempted than some other types of restoration, possibly because of the difficulties of reestablishing the hydrology associated with groundwater. One attempt to restore fens has been made in the Hanság, a wetland that was once the largest fen system in the Carpathian basin of Hungary (55 000 ha; Szenkendi 1938). Before the nineteenth century, several rivers from the Alps flowed into the Danube valley, disappeared into the huge floating fen system, and re-appeared at the outflow. Local people used the biological resources of the fen for fishing, hunting, collecting eggs of water birds, hay-making, and reed-harvesting. However, human settlements and agricultural fields near the fen were continuously threatened by inundation, so attempts were made to drain the Hanság, beginning in the nineteenth century. Despite drainage and disturbance, the Hanság retained large patches of natural vegetation in the wet meadows and forests; and the vegetation of aquatic communities survived in drainage channels. Because of its natural value, the Hanság was designated a protected natural area in 1976 and in 1994 as the Fertő–Hanság National Park. It is within this park that the restoration project was conducted.

To restore the wetlands, managers of the Fertő–Hanság National Park reflooded 400 ha of poor quality grassland, with the support of the Hungarian and Dutch governments. To re-engineer the restoration sites, dikes were built around the three separate wetlands. Gravity transported water through sluices from the river Rábca and the channel Kismetszés (Fig. 11.8). The first and second units were flooded in spring 2001 and the third unit in autumn 2001. The water-level has stayed constant at 0–100 cm above the soil surface. While a floating fen was not restored by this procedure, a wetland with an open-water habitat was created for water-birds and fen plants. Invasive plant species such as *Solidago gigantea* were reduced in the restored wetland.

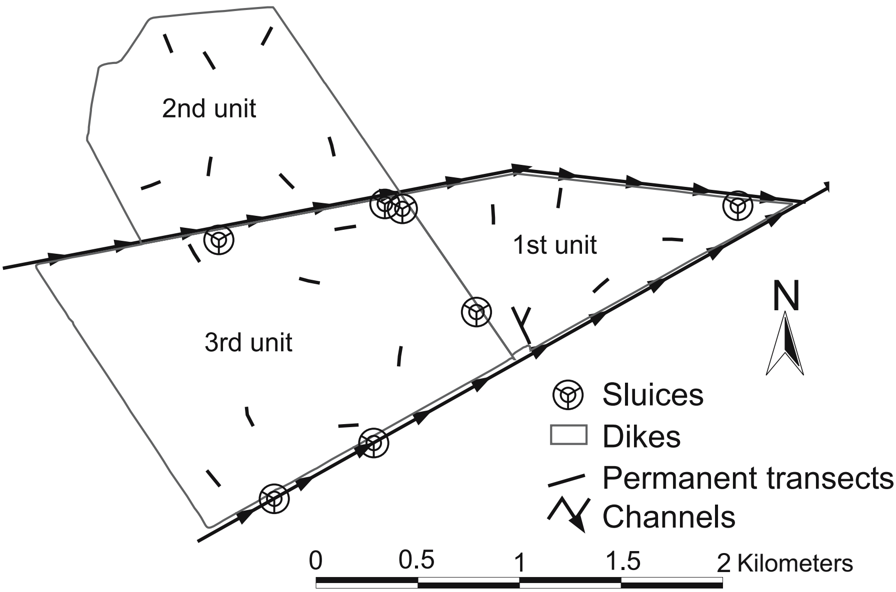


Fig. 11.8 Map of the restoration area, showing the location of permanent transects, in the Hanság fens of the Fertő–Hanság National Park, Hungary (Margóczi, unpublished data).

11.5.3 Monitoring and Evaluation of the Created Wetland

Monitoring of the restoration site was conducted in two ways, including field sampling and remote sensing. Field surveys of the vegetation were conducted by recording percent cover within 5 × 5 m phytosociological relevés along 21 permanent transects, 100 m long each, during 2001–2003. To create maps of vegetation change over time, fine resolution aerial orthophotos were compared every third year following the hydrologic restoration and a GIS database was developed. A rough vegetation map was made before the flooding began. The main vegetation types before flooding included fen meadows with large sedges (57 % *Carex riparia*, *C. acutiformis*), wet meadows (28 % *Alopecurus pratensis*, *Festuca arundinacea*), and reeds (5 % *Phragmites australis*, *Glyceria maxima*; Takács and Margóczi 2002). Important shifts in vegetation type were apparent during the first three years of flooding (Fig. 11.9). During the second year of flooding, upland vegetation disappeared; the area of the *Typha*, aquatic plants, and open water increased, while the area dominated by sedges decreased.

Natural wetland communities have redeveloped in the Ferto–Hanság National Park and are an attractive breeding and feeding area for birds. However, the hydrology is considerably different from the original fen, so the natural plant communities undoubtedly differ from those communities present

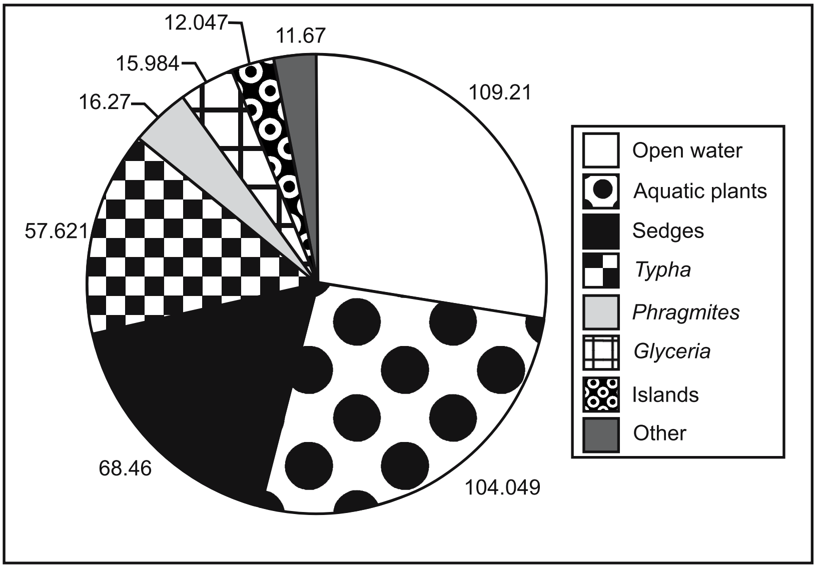


Fig. 11.9 Area (ha) of the main vegetation types in the flooded units (Units 1–3) in the Hanság fens, Hungary, in 2003. Area sizes were computed using raster statistics from the GIS database (Margóczi, unpublished data)

before the drainage of the ancient fen. While fen vegetation is not expected to re-develop, the Hanság National Park will continue to monitor the wetland restoration. Among the challenges that the managers of this national park face is that the restoration site is surrounded by agricultural lands, so that a considerable amount of water management and soil conservation in the regional landscape may be necessary to save the natural values of these wetlands.

11.6 Concluding Remarks

In times without much human influence, fen development was remarkably stable and vegetation types existed for centuries until they were replaced by meadows and alder wood. Nowadays fens cannot be maintained without management such as mowing or grazing. Nevertheless biochemical mechanisms exist that can retard a rapid succession to shrubs and forest. Such mechanisms are governed by subtle hydrological systems that keep the fen in a good condition.

A prerequisite for successful nature management is the creation of suitable habitat conditions (nutrient status, hydrology). There will be no adequate fen restoration without a proper assessment of the hydrological functioning of the fen system. It is very important that stagnation of surface water in the summer is prevented in restoration projects dealing with desiccated fens, in particular when the groundwater or surface water is rich in sulfate. The best option, which is not always available, is to stimulate the discharge of unpolluted (sulfate- and nitrate-poor) groundwater in such a way that this groundwater can flow through the system, leaving iron behind to contain the phosphates in the top soil.

Management measures are needed to counterbalance, for instance, the increased annual nitrogen input from atmospheric deposition. This may be accomplished by annual hay-harvesting, but it should be realized that hay-harvesting may induce a shift in the type of nutrient limitation, which also may affect biodiversity (Olde Venterink et al. 2003).

From studies on soil seed banks and seed dispersal, we may conclude that, until now, both seed persistence in the soil and dispersal distances of wet grassland species have been rather underestimated. Both types of seed sources can be manipulated by nature management, both in conservation and restoration projects. For successful restoration, both temporal (time since alteration of the habitat) and spatial (degree of fragmentation/isolation) aspects have to be taken into account.

We may expect that, in the near future, large areas on low-lying peat soils in large part of Europe and North America can no longer be maintained as intensely used agricultural areas. This offers good opportunities for the development of eutrophic marshes, as in our example of the Hungarian wetland restoration. The reflooding of former agricultural areas usually results in very eutrophic soil conditions (Richert et al. 2000), in particular when the surface water is rich in sulfates and the soil has been iron-depleted due to long-term drainage. Under such conditions persistent marsh species, such as *Glyceria maxima*, *Typha* spp, and *Phalaris arundinacea* can expand rapidly, but these species do not usually form peat, although they can rapidly form highly productive marsh vegetation, suitable to sustain large populations of waterfowl. To restore the peat-forming function of the mire, the vegetation should consist of marsh species with less easily degradable tissue (*Phragmites australis*, tall *Carex* spp; Richert et al. 2000). Restoring all of the fen qualities and functions will not be possible in many cases where restoring the hydrological system is not yet an option. Setting clear targets which can meet the opportunities presented, combined with a realistic strategy for future possibilities, may prevent much disappointment. A well explained modest result sometimes does more good for the public support than a scientifically sound project that lacks community or administrative support.

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