

Restoring the Vegetation of Mined Peatlands in the Southern Rocky Mountains of Colorado, U.S.A.

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Abstract

South Park is a high-elevation, semi-arid, treeless intermountain basin in central Colorado. A few extreme rich fens occur on the western margin and in the center of South Park where regional and local groundwater flow systems discharge to the ground surface. Over the past 40 years there has been extensive peat mining in these fens, but restoration methods have yet to be developed and successfully applied. The first part of this study compared the naturally reestablished vegetation on six mined peatlands with six pristine sites, while the second part of the study tested different revegetation techniques in 27 plots with varying depths to the water table. The six mined sites had only 30 plant species as compared with 122 species in the unmined sites; 43% of the species in the mined sites were not present in the undisturbed fens. Even after 40 years the sedges and willows that dominate the undisturbed sites were largely absent on the mined sites. The revegetation experiments seeded eight species, transplanted *Carex aquatilis* (water sedge) seedlings, transplanted rhizomes from six species, and transplanted four species of willow cuttings. Of the eight species seeded, only *Triglochin maritima* (arrowgrass) germinated and established seedlings. *C. aquatilis* seedlings, rhizome transplants of *C. aquatilis*, *Kobre-*

sia simpliciuscula (elk sedge), and *Juncus arcticus* (arctic rush), and willow cuttings all had differing patterns of survival with respect to the annual maximum height of the water table. These results indicate that the dominant species can be successfully reintroduced to mined surfaces with the appropriate hydrologic conditions, but human intervention will be necessary to rapidly re-establish these species. The slow rate of peat accumulation means that restoration of the mined fens will require hundreds, if not thousands, of years.

Key words: Colorado, South Park, Rocky Mountains, fen, restoration, peat mining.

Introduction

Peat mining is a widespread industry in boreal regions of the Holarctic, particularly in Russia, Finland, and Ireland where peat is used as fuel (Moore & Bellamy 1974; Mitsch & Gosselink 1993). In North America peat is mined primarily for use as a soil supplement; Canadian *Sphagnum* peat is well known and widely used in the gardening and horticulture industries. Peat is also mined in the United States, especially in Michigan and Minnesota, and to a much more limited extent in Colorado (Carpenter & Farmer 1981; Stevens & Dorfer 1989).

The relatively dry continental climate in Colorado means that peatlands are found only in high-elevation sites where a constant supply of groundwater maintains saturated soils. Peatlands occupy only about 0.1% of Colorado's land area, and they are classified as transitional rich, rich, and extreme rich fens (Cooper & Andrus 1994; Cooper 1996). The dominant species are usually *Carex* (sedges) and *Salix* (willows), with little *Sphagnum* cover (Cooper 1990; Cooper & Andrus 1994).

Although Colorado peatlands are mined primarily for horticultural material, they have also been mined to reclaim land for livestock grazing and create fishing ponds. In contrast to areas such as Finland (Heikkilä & Lindholm 1995), Britain (Anderson et al. 1995), and northern Europe (Zeitz 1995), few Colorado peatlands have been converted to agriculture or forestry. Peat mining in Colorado is particularly destructive, as in most cases the entire peat body is removed with a dredge. The loss of this organic layer alters both the hydrologic condition and the substrate available for recolonization.

Because most Colorado peatlands are relatively small, most mined surfaces in Colorado have an adjacent area with relatively undisturbed peatland vegetation. The seeds from these undisturbed areas can be widely dispersed by surface runoff during spring

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snowmelt. Field observations indicate that many mined areas can remain relatively unvegetated for decades, or be colonized by pioneer species that are not commonly found in undisturbed peatlands. Many mined areas show little recolonization by the dominant fen species, even after 40 years.

These observations are generally consistent with studies of the recolonization of mined surfaces in Finland. Salonen (1992, 1994) and Salonen and Setälä (1992) reported that seed source, nutrients, and shade limit the natural recolonization of mined surfaces in Finland. However, many mined surfaces may remain largely unvegetated for decades, even when a seed source is present (Salonen & Setälä 1992; Salonen 1994). The rhizomatous species of *Carex* that dominate Holarctic fens and meadows rarely produce new genets (Gorham & Somers 1973; Callaghan 1976; Ratliff 1983; Jónsdóttir & Callaghan 1988; Jonsson et al. 1996). Seedlings of *Carex* are apparently established only during a relatively brief period of ecosystem initiation, with little additional recruitment as the community closes. Clonal genets may be very old (Eriksson 1993; Jonsson et al. 1996).

The majority of peat mining in Colorado has occurred in South Park, a large, cold, high-elevation intermountain basin located 120 km southwest of Denver. This is the only location in the southern Rocky Mountain region where extreme rich fens occur, and these fens are the southernmost limit for a large number of boreal plant taxa in North America. Thus the fens in South Park are floristically unique and of great conservation importance (Cooper 1991a, 1991b, 1996; Cooper & Sanderson 1997). Over the last 40 years mining has greatly altered many of these extreme rich fens, but revegetation and restoration have generally not been attempted.

Qualitative observations suggested that natural recolonization was not satisfactory. Therefore, this study was designed to more systematically evaluate the recovery of mined fens and test techniques for restoring the dominant plant species in mined areas. The purpose of this research was to determine which fen species are naturally recolonizing mined surfaces and to test four different revegetation techniques in mined plots. The four revegetation techniques that we tested were seeding, transplanting greenhouse-grown seedlings, transplanting rhizomes from adjacent sites, and transplanting willow stem cuttings. Because depth to the water table is a dominant control on wetland vegetation (Mitsch & Gosselink 1993), the revegetation techniques were tested on replicated plots located along a water table gradient. By analyzing the success of each revegetation technique along this gradient, we were able to evaluate the relative sensitivities of each species and each technique to the height of the water table. This information is necessary for the management, protection, and resto-

ration of extreme rich fens in Colorado and possibly elsewhere in the central and northern Rocky Mountains.

Study Area

The study was conducted in the northwestern part of South Park, which is a relatively level, semi-arid, cold, high-elevation steppe. Elevations range from 2,900 to 3,100 m, while the peaks in the surrounding mountain ranges reach over 4,200 m. Average annual precipitation is only 250 mm, and potential evapotranspiration is approximately 865 mm (Spahr 1981). Approximately 72% of the annual precipitation occurs during mid- to late-summer. Winters are generally dry, with less than 10 mm of precipitation falling in most months. Periods of several weeks without precipitation are common year-round. The limited winter precipitation and frequent winds result in little or no snow cover for extended periods.

The Mosquito Range forms the western boundary of South Park, and these mountains are the source of most of the surface and ground water in the fens. Annual runoff is dominated by the seasonal snowmelt peak in early June. The extensive areas of limestone and dolomite in the Mosquito Range produce surface and ground water that is highly calcareous. Ca^{2+} concentrations can exceed 50–100 mg/kg and the pH is generally greater than 7.6 (Cooper 1996). The high pH and calcium concentrations create extreme rich fen conditions (Sjörs 1950, 1961; Slack et al. 1980; Vitt & Chee 1990).

South Park's extreme rich fens occur in a north-south belt approximately 50 km in length within 10 km of the base of the Mosquito Range. They range in size from less than 1 ha to several square kilometers (Cooper & Sanderson 1997). The peat is from 1 m to more than 3 m thick, and peat basal dates range from approximately 8,000 to nearly 12,000 years BP, with mean rates of peat accumulation ranging from 8 to 30 cm/1,000 years in South Park peatlands (D. Cooper, unpublished data).

Within the fens there appear to be three distinct units defined by a combination of hydrologic regime, topography, and vegetation composition. These units are: (1) peat hummocks and hollows, with the hummocks dominated by the *Trichophorum pumilum*/*Kobresia simpliciuscula* community and the hollows by the *Eriophorum angustifolium*/*Juncus alpino-articulatus* community; (2) springs dominated by either the *Triglochin maritima*/*Carex aquatilis* or *T. maritima*/*Carex simulata* communities; and (3) water tracks dominated by either the *T. maritima*/*Triglochin palustris* or *Carex microglochin*/*Eleocharis quinqueflora* communities (Cooper 1996). Peat mining has occurred in all of these units and typically leaves no more than a few centimeters of organic substrate on top of Pleistocene outwash sands and gravel. After mining,

the sites can be quite wet due to continuing groundwater discharge, or, more commonly, the removal of the peat can cause the sites to be relatively dry. Growing conditions in the latter sites are particularly difficult because the lack of snow cover and large temperature fluctuations cause severe freeze-thaw and frost heave.

The revegetation experiments tested techniques to establish *Carex* species, *Kobresia simpliciuscula*, *Salix* species, and *Juncus arcticus* (arctic rush). These species were selected because many of the dominant plant species in the water track and hollow communities (*T. maritima*, *T. palustris*, *J. alpino-articulatus*, and *E. quinqueflora*) do establish from seed on bare wet sediment and were observed to be recolonizing the wetter parts of mined areas.

Materials and Methods

Floristic Comparison of Mined and Unmined Fens

The first part of this study assessed the floristic composition of 12 extreme rich fens during at least three visits per site over 2 years. Six of the fen sites were pristine, four sites had been largely mined (>80% of the peat body removed), and two sites had been partially mined (<50% removed). The six mined sites were selected because the adjacent unmined portions had similar site characteristics and plant communities to the six unmined sites, suggesting that the mined sites would have had similar plant communities as the unmined sites. No revegetation had been attempted at any of the mined sites. Mining had occurred from 7–40 years prior to our floristic inventory, and the size of both the mined sites and the unmined controls ranged from less than 1 to greater than 20 ha. Plant nomenclature follows Weber and Wittmann (1996).

Regeneration Experiments

Study Site and Water Table Measurements. The regeneration experiments were conducted in a 0.5 ha site in High Creek fen that had been mined in 1985. The initial field reconnaissance indicated a strong hydrologic gradient from north to south within the site. At the northern edge the vegetation indicated much drier conditions, while at the southern edge there were two small ponds with about 20 cm of standing water.

To better characterize the hydrologic conditions, a grid of 12 ground water monitoring wells was established in June 1992. One well was placed at each corner of the site, six wells were installed within the site, and one well was placed in each of the two ponds at the southern edge. The relative elevations of the monitoring wells were surveyed with an engineer's level and stadia rod. The depth to the water table was measured

six times during 1992, and less regularly during the summers of 1993 and 1994. The mean annual high water table was used to characterize the water table at each well and the hydrologic gradient. The representativeness of the hydrologic conditions during the study period was assessed by comparing the summer precipitation for 1992–1994 to the long-term averages. Precipitation data were obtained from Antero Reservoir, which is approximately 15 km south of the site used for the regeneration experiments.

Five equally spaced transects were laid out from north to south along the water table gradient. Along each transect five 5 × 3 m plots were laid out ("terrestrial plots"), and two additional plots were located in the area with permanent standing water ("aquatic plots"). Fifteen 0.5 × 0.5 m subplots were then established in each of these 27 plots. The subplots were cleared of any existing vegetation in early June 1992 and were immediately planted with one planting treatment per subplot. The survival of the plantings was periodically monitored through September 1994.

Seed Germination. Seeds of eight species—*Carex aquatilis* (water sedge), *C. simulata* (sedge), *C. utriculata* (beaked sedge), *K. simpliciuscula* (elk sedge), *E. angustifolium* (cottongrass), *Iris missouriensis* (iris), *T. maritima* (arrowgrass), and *Salix candida* (hoary willow)—were planted in June 1992. All of the seeds planted were collected in the field, but there were differences in the timing and treatment of the seeds. The monocot seeds were collected from culms that had grown in 1991 and overwintered, because this provided the desired cold stratification. For *S. candida* we used ripe, filled 1992 seed. Filled *C. aquatilis* perigynia were analyzed using tetrazolium to determine if embryos were viable, and 99% viability was found (Barry 1994).

For each species, one subplot was systematically selected in each of the 25 terrestrial plots and 20 seeds of the appropriate species were planted in each subplot, yielding a total of 200 planted subplots. Each subplot was watered immediately after planting to ensure a moist seedbed. Germination and survival surveys were conducted weekly through August 1992, and again early in the following summer.

Seedling Transplants. *C. aquatilis* seeds were collected from the site in late fall 1991 and early spring 1992, germinated in a glasshouse in April 1992, and transferred to 15-cm-long plastic tubes. After 2 months, 10 vigorous seedlings were transplanted into one randomly chosen subplot in each of the 27 study plots.

Rhizome Transplants. Rhizomes from six species were collected from vigorous populations in High Creek fen during June 1992. Each rhizome was at least 5 cm in

length and supported at least two live shoots. For each of three species (*C. aquatilis*, *K. simpliciuscula*, and *J. articus*) 10 rhizomes were transplanted into one subplot in each of the 27 plots (total of 81 subplots). Similarly, 10 rhizomes of *C. simulata*, *C. utriculata*, and *E. angustifolium* were respectively transplanted into one subplot within each of 10 plots that encompassed the water table gradient. Fewer of these species were collected and transplanted because they are relatively uncommon at High Creek fen.

Salix Stem Cuttings. *Salix* stems of largely woody tissue were harvested from four species with pruning shears in late June. The harvested stems were approximately 30 cm in length, and about half of the leaves were removed to reduce transpiration. Approximately two-thirds of the stem was placed below the soil surface. Twenty stems of *S. candida*, and 10 stems each of *Salix brachycarpa* (barren ground willow) and *Salix monticola* (mountain willow) were transplanted into separate subplots within each plot. The rare *Salix myrtilifolia* (myrtle-leaf willow) was transplanted into one subplot in each of four plots.

Statistical Analysis. The effect of water table depth on seedling, rhizome transplant, and willow stem transplant survival was tested using logistic regression analysis (Procedure GENMOD, SAS Institute 1992). A plant was considered alive if it had at least one living shoot and live leaves on that shoot. We focused our analysis on the survival data from late August 1992, early June 1993, late August 1993, and late August 1994, because this allowed us to separate winter and summer mortality. We tested logistic models with first-, second-, and third-order polynomials, and selected a model according to the chi-square goodness-of-fit and model deviance (Trexler & Travis 1993).

Results

Floristic Comparison of Mined and Unmined Fens

Only 30 plant species were found in the six mined fens as compared with 122 species in the six unmined fens. Thirteen of the 30 species found on mined surfaces were not found in the unmined fens, and these 13 species included both native and exotic species. Most of the native plants were found in the wetter part of the mined fens, and this was particularly true for *T. palustris*, *J. alpino-articulatus*, and *E. quinqueflora*. *Juncus bufonius* (toad rush) was another common colonizer in the wetter mined areas, but this was not present in the unmined sites.

Deschampsia cespitosa (tufted hairgrass) and *Argentina anserina* (silverweed) were the most common native col-

onizers in the seasonally dry mined sites. These two species are widespread in undisturbed fens but have low cover. *Agrostis scabra* (ticklegrass), *Potentilla norvegica* (Norway cinquefoil), and *Polygonum arenastrum* (knotweed) were the most common non-native plants found on the mined surfaces but did not occur in the unmined fens.

Although the mined surfaces varied in age from 7 to 40 years, there was no apparent trend in floristic composition or plant density with time since mining. In addition, there appeared to be little difference in the species composition between the edges and the middle of mined sites, and this suggests that seed source was not a limiting factor in recolonization. A few plants that appeared to be seedlings of *C. aquatilis* were found on the margin of one small pond in a mined surface; however, seedlings of other *Carex*, *Kobresia*, or *Salix* species were not found.

Regeneration Experiments

Hydrologic Conditions. The water level data showed that the water table was generally deeper and more variable in the northern portion of the 0.5-ha site used for the regeneration experiments. In spring and early summer the water table was approximately 20 cm below the soil surface in the northern portion of the study site, while the southern portion was either saturated or had standing water. Over the course of the summer, water levels dropped to as much as 50 cm below the ground surface in the northern portion, while there was relatively little change at the southern edge. There was about 20 cm of standing water in the two ponds throughout the study.

May through August precipitation during 1992, 1993, and 1994 was near average for all months except July. In 1992, July precipitation was 23 mm or 50% of normal, while in 1993 and 1994 July precipitation was less than 25% of the long-term average. These data suggest that surface soil moisture would have been somewhat drier than normal in July, but the wetter sites were not as affected because they are dominated by regional-scale groundwater inputs.

Seed Germination Experiment. Of the eight species planted in early June 1992, only two germinated. *T. maritima* was the most successful as it germinated in 59% of the subplots. Germination was not observed in the ponds and the driest plots. *S. candida* germinated in six plots, but all the seedlings died within 1 month. The low survival may have been due in part to the below average precipitation recorded in July 1992.

Seedling Transplants. Of the 270 *C. aquatilis* seedlings planted, 95% survived the first summer (Table 1). Sev-

Table 1. Number of seedlings, rhizome transplants, and willow stem cuttings planted in June 1992 and number (*n*) surviving from 1992 to 1994.

		Number Surviving			% Survival	
		1992 (Aug.)	1993 (June)	1993 (Aug.)	1994 (Aug.)	
Seedlings						
<i>Carex aquatilis</i>	270	257	181	170	136	50
Rhizomes						
<i>Carex utriculata</i>	100	92	86	81	65	65
<i>Carex aquatilis</i>	270	247	215	186	137	51
<i>Carex simulata</i>	100	89	69	59	42	42
<i>Kobresia simpliciuscula</i>	270	217	118	81	70	26
<i>Eriophorum angustifolium</i>	100	90	37	32	13	13
<i>Juncus arcticus</i>	270	200	137	74	19	7
Cuttings						
<i>Salix myrtillofolia</i>	40	34	22	14	13	33
<i>Salix monticola</i>	270	245	130	108	70	26
<i>Salix candida</i>	540	267	108	96	71	13
<i>Salix brachycarpa</i>	270	137	47	45	33	12

^aPercent survival to late August 1994 is shown in the last column.

enty-six seedlings died during the first winter and 11 died during the second summer. After three summers 136 seedlings (50%) were alive, and many had spread to form clones with 10 or more shoots. Seedling survival was highest in plots where the mean maximum water table was within 10 cm of the surface or where shallow standing water was present (Fig. 1a).

Logistic models of seedling survival along the water table gradient had similar shapes for each date when survival was assessed. The model for late August 1994 is $p = 1 / \{1 + \exp[-(0.897 + 0.070x - 0.007x^2 - 0.0002x^3)]\}$, where x is the mean maximum depth to water table. All three model parameters are statistically significant (WT1 $\chi^2 = 9.41$, $df = 1$, $p < 0.01$; WT2 $\chi^2 = 29.40$, $df = 1$, $p < 0.001$; WT3 $\chi^2 = 7.38$, $df = 1$, $p < 0.01$).

Rhizome Transplants. There was considerable variation in rhizome survival rates among the five species tested. The highest survival rates were for *C. utriculata* (65%) and *C. aquatilis* (51%), while relatively few *K. simpliciuscula* (26%), *E. angustifolium* (13%) and *J. arcticus* (7%) survived (Table 1). There were substantial differences among species with respect to the timing of mortality and the pattern of survival along the water table gradient.

As in the case of the seedling experiment, *C. aquatilis* had the highest survival rate in the plots with either shallow standing water or a water table near the soil surface (Fig. 1b). In the drier sites, the rhizome transplants of *C. aquatilis* had slightly higher survival rates than the seedlings (Fig. 1a & 1b). Logistic analysis confirmed a statistically significant increase in August 1994 survival with increasing height of the water table; the best regression was $p = 1 / \{1 + \exp[-(0.2996 + 0.053x)]\}$,

and again the mean maximum depth to water table is significant ($\chi^2 = 24.34$, $df = 1$, $p < 0.001$).

For all three *Carex* species the greatest percent mortality occurred in the second year (between August

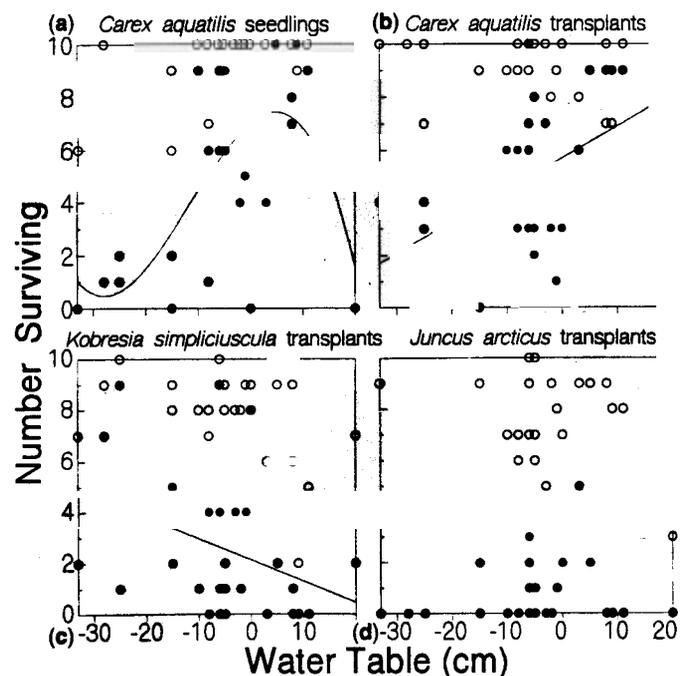


Figure 1. Survival of (a) *Carex aquatilis* seedlings, and (b) *Carex aquatilis*, (c) *Kobresia simpliciuscula*, and (d) *Juncus arcticus* transplants in the first year (August 1992, open circles) and third year (August 1994, filled circles) as a function of mean maximum water table. Each circle represents one subplot. Regression lines are shown for seedling and rhizome survival through August 1994. See text for regression statistics.

1993 and August 1994) (Table 1). Logistic models were not developed for *C. utriculata* and *C. simulata* because the number of surviving rhizome transplants was too small. However, the highest survival of *C. utriculata* and *C. simulata* was in the wetter plots.

In contrast to the three *Carex* species, the greatest mortality of the *K. simpliciuscula* rhizomes was between August 1992 and June 1993 (Table 1). The survival of *K. simpliciuscula* was also lowest in the wetter sites, with few plants surviving through August 1994 at sites where standing water was present (Fig. 1c). Logistic analysis indicated a statistically significant increase in transplant survival with a decrease in the mean maximum height of the water table. The model for *K. simpliciuscula* survival in late August 1994 is $p = 1 / \{1 + \exp [-(1.355 - 0.044x)]\}$ and the model parameter is significant ($\chi^2 = 14.54$, $df = 1$, $p < 0.001$).

The *J. arcticus* rhizomes had the lowest survival rate and the most consistent mortality over time (Table 1). Too few transplants survived through August of 1994 to model, but the highest survival rates were in plots with a maximum water table ranging from 15 cm below the soil surface to 5 cm above the soil surface (Fig. 1d).

The *E. angustifolium* rhizome transplants also had a very low survival rate. Survival tended to be higher in the drier plots, but the sample size was too small to model. Like *K. simpliciuscula*, most of the mortality occurred over the winter between August 1992 and June 1993 (Table 1).

Willow Stem Cuttings. Stem cuttings of *S. myrtilifolia* and *S. monticola* had moderate survival rates of 33% and 25%, respectively (Table 1). Survival rates of *S. candida* and *S. brachycarpa* were only 13% and 12%, respectively. For all four species the greatest percent mortality was between August 1992 and June 1993 (Table 1).

Survival of *S. candida* and *S. brachycarpa* was highest in plots with a water table near the soil surface or where shallow standing water occurred seasonally (Fig. 2a & 2b). In contrast, the highest survival rates of *S. monticola* were in those plots with a mean maximum water table height of 0–30 cm below the soil surface (Fig. 2c). *S. myrtilifolia* cuttings had the highest survival rate but too few stems were planted to accurately determine the effect of water table depth on survival. The high mortality rates of *S. candida* and *S. brachycarpa* cuttings were apparently due to the slow development of an adventitious root system.

Discussion

We found little similarity in the species composition between mined and unmined fens. Most of the species found in the mined sites were exotic and native plants that were uncommon or not found in the unmined sites.

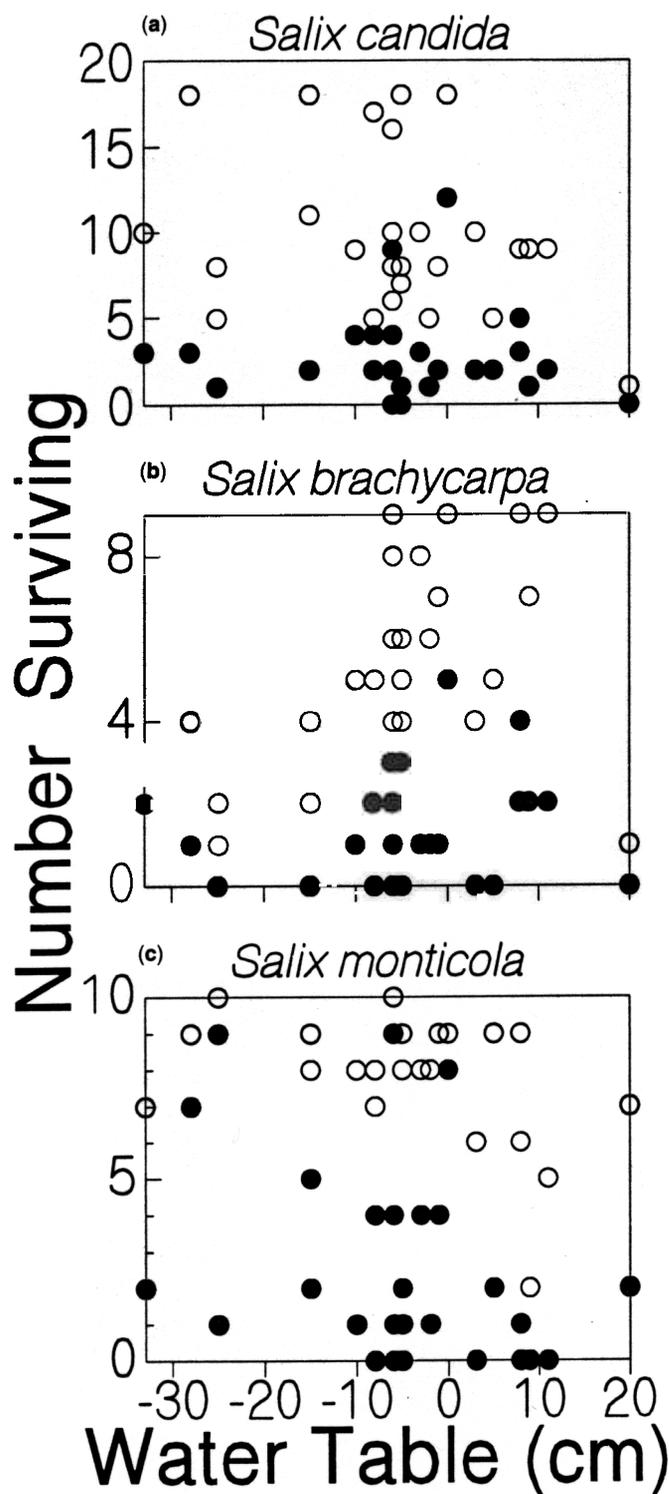


Figure 2. Survival of (a) *Salix candida*, (b) *S. monticola*, and (c) *S. brachycarpa* stem cutting transplants in the first year (August 1992, open circles) and third year (August 1994, filled circles) as a function of mean maximum water table. Each circle represents one subplot.

This suggests that colonization of mined sites by seed rain from adjacent unmined sites is generally not successful in re-establishing the natural vegetation. The wettest portions of the mined sites had the highest cover of native plants, as *Triglochin palustre* and *E. quinqueflora* had become well established in these areas. The only naturally established *C. aquatilis* seedlings were located on the margin of a pond. This suggests that permanently saturated soils or shallow standing water may provide sites that are less prone to drought and frost heaving, thereby facilitating germination and seedling establishment. However, only a small fraction of South Park's mined surfaces are permanently saturated.

The seeding trials confirmed that sedges and willows do not readily germinate in the field, even where the hydrologic conditions seem appropriate. In contrast, the dominant plant species in the water track and hollow communities (*T. maritima*, *T. palustre*, *J. alpino-articulatus*, and *E. quinqueflora*) have established from seed in mined sites where springs occur and in water tracks.

The survival of transplanted seedlings, rhizomes and stem cuttings varied by species. Survival for all species varied with location along the water table gradient. However, some important species such as *J. arcticus*, *E. angustifolium*, *S. candida* and *S. brachycarpa* had poor survival over the entire gradient tested.

The overall survival of *C. aquatilis* seedling and rhizome transplants was similar (50% and 51%, respectively), but there were slight differences in the water table conditions where each survived. Seedling survival was highest in sites with a water table near the soil surface, while rhizome transplants had slightly better survival rates in those plots with either deeper standing water or a deeper water table (Fig. 1a & 1b). The broader tolerance of *C. aquatilis* rhizome transplants suggests that their rhizomes may be a promising technique for revegetating mined areas.

Rhizome transplants of *K. simpliciuscula* were most successful in sites where the seasonal water table maximum ranged from 10–30 cm below the soil surface. This species is of particular importance because it forms the peat hummocks that support many of the rare plants found within South Park's extreme rich fens (Cooper 1996).

The pattern and timing of survival among the different species suggest that different processes may limit the recolonization success of each species. Species that dominate water tracks and hollows may be limited more by seed source and tend to have greater mortality during the summer. In contrast, the species found on hummocks and in drier sites are limited more by the lack of safe sites for seed germination and seedling establishment, and tend to have greater winter mortality.

Although this study focused on the natural and experimental revegetation of mined sites, wetland resto-

ration efforts must also consider the extent to which the basic hydrologic regime has been modified and the scale of the disturbance. With respect to the hydrologic regime, the fens in South Park are relatively unaffected by the water diversions, groundwater pumping, drainage ditches, or other hydrologic changes that hinder fen restoration in Europe (Van Diggelen et al. 1994; Grootjans & Van Diggelen 1995; Okruszko 1995) and Colorado (Cooper et al. 1998). Because the hydrologic systems within the study area are relatively intact, it is at least theoretically possible to restore the fen vegetation and reinitiate the accumulation of peat.

However, a key limitation to the restoration of mined peatlands in South Park is the loss of the peat layer. In South Park it appears that groundwater discharge occurs primarily at discrete springs and mining the peat layer leaves large areas with relatively dry surface soils. We hypothesize that after the retreat of the glaciers, the peatlands first began to form with establishment of *C. aquatilis* and other fen dominants at groundwater discharge locations. Over time these sites developed into saturated peat mounds, and these mounds slowly spread across the valley bottom to coalesce with other peat mounds.

The basal ages indicate that the South Park peatlands are the product of thousands of years of peat accumulation. Mining effectively resets the system, and creates greater heterogeneity in the hydrologic regime. These differences then affect the suitability of sites for plant growth and the frequency and severity of other physical processes, such as frost heave.

In theory, the sedges that dominate most Colorado peatlands can spread clonally to recolonize the wetter mined sites. As organic matter accumulates and helps sustain saturated conditions, these clones can slowly spread to reoccupy more of the mined areas. The problem is that peat mining removes the vegetation and accumulated peat from large areas that cannot be rapidly colonized by clones from adjacent unmined sites. These patches necessitate the establishment of new genets that represent independent colonization events (Harper 1977).

Unfortunately, the floristic assessment indicated that there is little or no natural colonization of mined peatlands by the dominant species of *Carex*, *Kobresia*, and *Salix*. At least in the short term, natural invasion cannot be relied upon for revegetation. Hence the dominant peat-forming species such as *C. aquatilis*, *C. simulata*, and *C. utriculata* will have to be established from rhizome transplants or greenhouse-grown seedlings, and these will have the greatest survival in the wetter sites. Over time they could spread to form sedge mats and increase the capacity of the soil to retain water. In the drier areas it might be possible to establish *K. simpliciuscula* from rhizome transplants or greenhouse-grown seedlings. Additional field experiments and observa-

tions from undisturbed sites are needed to identify the optimal conditions for the survival of each transplanted species.

While these techniques may lead to the reestablishment of a plant cover that bears some similarities to undisturbed sites, the formation of a peatland is much slower. Peat accumulation rates in other montane fens in Colorado average approximately 20 cm/1,000 years (D. Cooper, unpublished data), indicating that the formation of peat bodies 1 m thick will require thousands of years. Since South Park supports the only known extreme rich fens in the southern Rocky Mountain region, and many fens contain unique plant communities (Cooper & Sanderson 1997), the protection and restoration of these peatlands should be a high conservation priority.

A final issue is that southern Rocky Mountain fens have formed in several different landforms, including kettle basins, mountain front toe slopes, and sites of regional ground water discharge such as High Creek fen. The restoration potential of fens in each landform type may differ because of the varying hydrologic regimes and the way in which peat bodies are formed, i.e., filling ponds from the edge to the center versus vertical accumulation at sites of ground water discharge.

Acknowledgments

This research was supported by the Colorado Field Office of The Nature Conservancy. We thank Dr. Alan Carpenter for his help throughout this study. Patsy Barry and Heather Crosby assisted with the field work, and Dr. David Merritt assisted with statistical analyses. We thank two anonymous reviewers and Dr. W. A. Niering for comments that substantially helped this paper.

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