Wetlands Ecology and Management **11:** 65–83, 2003.

Hydrological processes in abandoned and restored peatlands: An overview of management approaches

J.S. Price[1], A.L. Heathwaite[2] & A.J. Baird[2]
[1]*Department of Geography, and Wetlands Research Centre, University of Waterloo, Waterloo, ON, N2L 3G1, Canada; E-mail: jsprice@watserv1.uwaterloo.ca;* [2]*Sheffield Wetlands Research Centre (SWeRC), and Department of Geography, University of Sheffield, Sheffield, S10 2TN, U.K.*

Received 6 February 2001; accepted in revised form 29 June 2002

Abstract

Mined peatlands do not readily recover their hydrological function, mainly because the dominant peat-forming plant genus, *Sphagnum,* cannot easily reestablish on the degraded surface peat found on cutover sites. Drainage and removal of the acrotelm can result in surface subsidence of up to 3.7 cm y^{-1} m^{-1} of peat shortly after drainage (compression), and long-term rates up to 0.3 cm y^{-1} m^{-1} (compression and oxidation). This can decrease the hydraulic conductivity by over 75%, and decrease the water retention capacity and specific yield. In old abandoned systems, drainage ditches may continue to facilitate a significant seasonal water loss. Colonization of abandoned sites by trees may increase the evapotranspirative losses by as much as 25%, and interception losses can be as high as 32% of rainfall. Without natural or planned occlusion of ditches, some peatlands become drier over time. Blocking ditches may largely restore water balance components, although the hydrological regime requires years to stabilise sufficiently for *Sphagnum* recolonization, especially where residual peat is well decomposed, having inadequate water storage capacity. Consequently, winter precipitation (Europe) and spring snowmelt (North America) are critical recharge periods. Over the long term, consolidation of the peat due to drainage and methane production (where drainage systems are blocked and soils reflooded) decreases hydraulic conductivity, thereby reducing lateral seepage losses. This may actually assist in *Sphagnum* recolonization. A regenerated cover of *Sphagnum* increases soil wetness and reduces water tension (increases pore-water pressure) in the substrate, thus ameliorating its own environment. However, natural recolonization and recovery of many hydrological and ecological processes may not occur, or may require many decades. Water management and selective plant reintroduction can accelerate this. Water management options such as blocking ditches, constructing bunds, reconfiguring the surface and managing microclimate have met with varying degrees of success. No standard management prescription can be made because each site presents unique challenges.

Introduction and background

The annual volume of peat extraction is a small fraction of net global peat accumulation, which has led industrial producers to claim that peat extraction is a sustainable resource (e.g. CSPMA, 2000). Globally, peatlands are an important carbon store (Gorham, 1991), and exploitation can increase the release of carbon dioxide by 400% (Waddington and Price, 2000). On a regional and local scale, peat cutting profoundly alters hydrological and ecological functions; on the time scale of resource extraction and use, cutting cannot be considered sustainable (Joosten, 1995). Many of the changes are irreversible, such as changes to the soil structure (Okruszko, 1995; Gottlich et al., 1993). It is still uncertain if the hydrological, ecological and carbon storage functions of peatlands can be restored. Restoration implies the development, through management, of functions *very similar* to those of the undisturbed peatland. It is important to note that it is unlikely a peatland can ever be restored *sensu stricto* except perhaps in the long term. Over the short-term, ‘rehabilitation’ of broadly similar functions is a more realistic objective. Nevertheless, the term ‘restoration’ is used loosely by many to include both restoration and rehabilitation. In keeping with this convention, we use the term in this looser sense throughout this paper. This paper reviews the hydrological processes associated with peat cutting and abandonment, primarily on bogs, and demonstrates the role and need for appropriate water management in peatland restoration.

Peatlands are the result of complex interactions of biotic and abiotic processes that typically occur over thousands of years. In managing the hydrology and ecology of degraded peatlands, it is important to recognise and understand the constraints imposed by climate, landscape and continued human impact. Globally, most peatlands occur between latitudes of 45 and 65°N (there being relatively little land area in this zone in the southern latitudes). There are some important occurrences of peatlands in Australasia; these are discussed at length by Whinam et al. (this issue). This pattern of peatland distribution is related to latitudinal and meridianal gradients that control peat accumulation through (1) plant productivity and decomposition as influenced by photosynthetically active radiation and temperature; and (2) moisture availability, which is a function of atmospheric water supply and evapotranspiration. When these conditions are favourable, and occur within a suitable landscape (e.g. slope, substrate permeability), peatlands may begin to develop. Patterns of water flow and storage gradually change as peat accumulates. The physical and chemical properties of the accumulating peat regulate, in part, the quantity and quality of water flowing into and out of the peatland, and therefore have a profound effect on ecological processes within the peatland (e.g. redox and fertility gradients – cf. Wheeler, 1999; Ingram, 1983).

Bogs are peatlands that have reached a stage of development such that the properties of their peat, together with their geometry and position within the landscape, limit water and solute inputs to those solely derived from direct precipitation (Ingram, 1983; Ivanov, 1981). Under these circumstances, *Sphagnum* mosses usually form the predominant surface cover (Clymo and Hayward, 1989), although some bogs, e.g. some blanket bogs, may function naturally without much *Sphagnum* (Baird et al., 1997; Tallis and Meade, 1997). The *Sphagnum* eventually decomposes into the peat that has become the object of most peat extraction operations. When bog restoration is planned, it is important to recognise that climate, depth of peat accumulation, and peatland geometry have changed substantially since the initial peat formation. Under today's (typically warmer) climate, the hydrological conditions may be sufficiently different (dry) to preclude initiation of peat forming processes: this clearly has implications for restoration (Heathwaite, 1993). The success of any restoration programme relies on a good understanding of natural bog functions to establish suitable restoration goals. For example, there is a distinction, as noted above, between bog restoration in the strictest sense, and creating and managing a new wetland that provides some of the functions and features that it had prior to its exploitation.

Undisturbed boos are characterised by a two-layered soil structure (Ingram, 1978; Ivanov, 1981; Ingram, 1992), with an upper layer, or acrotelm, composed of living, dead and poorly decomposed mosses typically 0 to 50 cm thick. The acrotelm, by usual definition (but see Wheeler, 1999, p. 154), encompasses the full range of water table fluctuations. Its hydraulic structure changes rapidly with depth below the surface. Hydraulic conductivity (K) typically decreases by up to 4 orders of magnitude over a distance of 50 cm (Boelter, 1965) and sometimes more (e.g. Hoag and Price, 1995); and specific yield (Sy) from 0.5 to 0.1 over the same range (Price, 1992). By contrast, the lower layer of more decomposed peat, or the catotelm (Ingram, 1978), usually shows less variation in hydraulic structure with depth (but see Chason and Siegel, 1986). The more decomposed peat of the catotelm has smaller pores than peat in the acrotelm: consequently K is lower because the flowpath is more tortuous and frictional effects on flow are much greater. Undisturbed bogs rely on this two-layered structure to regulate the storage and discharge of water. When the water table is high, flow is through and over the relatively permeable upper layers of the acrotelm, so bogs readily shed 'excess' water (Bay, 1969). The high Sy in the near-surface layers allows this efflux with a relatively small drop in water table (see below for a description of Sy). However, extended periods of drainage coupled with evapotranspiration losses cause the water table to drop, so that flow occurs through layers of lower permeability (Boelter, 1965). Consequently, water is retained longer, and runoff may cease (Bay, 1969). Thus non-vascular plants (esp. *Sphagnum* spp.) receive adequate aeration for growth, yet have access to water (by capillary suction) in all but the driest conditions.

Peat extraction typically removes the acrotelm to expose the more decomposed and lower-K peat of the catotelm. This profoundly affects the water storage-runoff relationships (Schouwenaars and Vink, 1992), the nature and magnitude of evaporation losses (Price, 1996; Spieksma et al., 1997) and, eventually, soil processes including carbon dynamics (Waddington and Warner, 2001). Consequently, following extraction the hydrological conditions of the cutover peat are unfavourable for the re-establishment of *Sphaguum* mosses, the dominant peat forming plants. Thus, sites do not readily regenerate to functional bog ecosystems (Famous et al., 1991), although other bog species (mostly vascular plants) may re-establish (Lavoie and Rochefort, 1996; Spieksma et al., 1997). With the combination of partly operative drainage ditches and the absence of the acrotelm to stabilise water levels, the peat continues to degrade, exacerbating water table instability and the general unsuitability of the site for revegetation with *Sphagnum.* The hydrological problem is related to the inability of the non-vascular *Sphagnum* to generate the necessary capillary pressure to withdraw water from the cutover peat. In a natural setting, *Sphagnum* mosses are adapted to growing on dead, but essentially undecomposed versions of themselves. The high storativity (mostly Sy), imparted by these loosely packed poorly or undecomposed mosses, maintains a high and stable water table, so that a relatively small capillary rise is necessary to ensure an adequate moisture supply to the growing part of the plant. The *Sphagnum* carpet can only generate relatively weak capillary pressures within their intracellular spaces (hyaline cells) and inter-cellular pores (between branches and leaves of adjacent plants). Hayward and Clymo (1982) found that drainage of hyaline cells in *Sphagnum* occurred when pore-water pressure was below about –100 mb. The corollary of this is that *Sphagnum* plants are unable to withdraw moisture from a substrate where the pore-water pressure is lower that –100 mb. In a natural setting, this is not normally a problem.

In addition to water availability limitations, bare peat is sometimes subject to disturbances caused by needle-ice formation. This creates an unstable surface and can directly damage (kill) vascular plants through root exposure and breakage (Groeneveld, 2002). Nurse plants, especially the moss *Polytrichum strictum* curtail the damage done by frost (Groeneveld, 2002).

Hydrological landscapes of cutover peatlands

1. Block cut peatlands

Small-scale peat cutting to provide fuel to offset shortages of firewood has taken place in Europe since at least 2000 BP (Gottlich et al., 1993). Evidence of peat cutting is still visible in the raised bogs and blanket bogs of Britain and Ireland. Despite the basic tools and limited technology, there is evidence to 0suggest that historical peat cutting, albeit in non-bog peats, was extensive enough to alter the regional hydrology in some areas. The Mediaeval peat cuttings of the Norfolk Broads in the east of England, for example, would, at their peak, have rivaled in size many modern day commercial operations (Lambert et al., 1960).

Early horticultural peat extraction was typically by block-cutting (Gottlich et al., 1993), leaving a landscape of alternating baulks and trenches. Tertiary drainage occurs along trenches, into secondary drains dug well into the peat, eventually connecting to higher order channels (primary drains) either through, or adjacent to the peatlands. Generally, manual block-cutting is no longer used commercially but numerous abandoned block-cut peatlands remain in eastern North America (Lavoie et al., this issue), UK, and continental Europe (Wheeler and Shaw, 1995). Certain peat extraction operations, including non-horticultural peat production employ mechanical block-cutting but this does not leave the characteristic baulk-trench landscape typical of hand-dug systems.

2. Mechanised peat cutting

By the mid-1970s mechanised cutting became the dominant method of peat harvesting in Europe and North America. This typically occurs on a much larger scale than block-cutting, exacerbating the hydrological and ecological constraints on revegetation by peatland plants (Money, 1995). To ensure drainage and an appropriate bearing capacity for large vehicles, ditches are deep (Mulqueen, 1989), and the surface cambered to enhance surface runoff. Ditches for horticultural peat extraction are typically 0.7 to 1 m deep, and spaced 30 m apart. Surface peat (acrotelm) is typically removed and the skag or stripping spoil (Heathwaite et al., 1993) discarded. Milled peat harvesting (Gottlich et al., 1993) is the most common method of extraction, where the surface is milled (to facilitate drying) and peat fragments vacuumed from the surface (vacuum harvesting), or removed mechanically.

Table 1. The relationship between specific yield and *Sphagnum* peat of varying humification at different depths derived from lysimeter experiments in the Engbertsdijksvenen raised bog, The Netherlands (after Schouwenaars, 1993).

Peat type	Degree of humification on Von Post scale	Depth (cm)	Specific yield*
Young living *Sphagnum*	–	0–15	0.23–0.34
Slightly humified *Sphagnum* peat	H2–3	10–30	0.11–0.17
Moderately humified *Sphagnum* peat	H3–4	0–40	0.11–0.13
Strongly humified *Sphagnum* peat	H6–7	0–35	0.14–0.33^{+} 0.05–0.10$^{\#}$

* Schouwenaars (1993) terms this the 'water storage coefficient' which he obtained by regressing changes in water storage (per unit area) against changes in water table depth. The water storage coefficient (dimensionless) is the gradient of the regression line. In essence this is equivalent to the specific yield.

$^{\#}$ Range for samples with many living roots.

$^{+}$ Range for samples without roots.

The size (distance to seed-bank) and hydrological disruption (e.g. ditch depth and surface cambering) make these sites much less favourable for self-regeneration or restoration (Money, 1995; Lavoie and Rochefort, 1996).

Hydrological changes

The intentional outcome of drainage for peat extraction is lowering of the water table. The combination of drainage and peat cutting produces a complex and sometimes contradictory hydrological response. Observed drainage effects include increases in runoff, increases in peak flows and increases in baseflow relative to natural conditions (Burt et al., 1990; Conway and Millar, 1960; Mikulski and Lesmak, 1975; Nicholson et al., 1989; Robinson, 1985), although these changes do not necessarily occur together or on all sites. Indeed, some studies (Burke, 1972; Baden and Eggelsmann, 1968) report a decrease in peak flow, for example, because of greater available storage capacity in drained soils between storms. Occlusion of tertiary drains occurs shortly after abandonment, and partial collapse of secondary ditches may follow after a few decades, but primary drains may remain effective for longer (Van Seters and Price, 2001). After rewetting cutover peatlands may exhibit a flashy hydrograph, and over the longer term may not be effective regulators of streamflow (Spieksma, 1999).

An unintentional effect of peatland drainage includes peat subsidence caused by shrinkage and oxidation of peat above the water table, and compression below (for a general discussion see Heathwaite et al., 1993). Schothorst (1977, 1982) estimated that 65% of long-term subsidence is caused by shrinkage above the water table (85% of this due to oxidation), and 35% by compression below the water table. Van Seters and Price (2002) found subsidence in an uncut remnant of a drained bog to be approximately 80 cm over 57 years, causing Sy to decline from 0.14 to 0.07, water table fluctuations to be 67% greater, and mean saturated K to decline from 4.1×10^{-5} to 1.3×10^{-5} cm s^{-1}. The decrease in hydraulic conductivity reduces the vertical and lateral movement of water in the peat deposit and may eventually assist in raising the water table by reducing groundwater seepage losses (Van Seters and Price, 2002). Furthermore, greater (vertical) capillary flows may occur as a consequence of the new hydraulic geometry (Price and Whitehead, 2002), whereby smaller pores in the unsaturated zone exert a stronger pull on the residual water, compared to uncompressed peat at a similar water content (Schlotzhauer and Price, 1999). Rothwell et al. (1996) concluded that potentially negative effects of drainage (i.e. on peat oxidation, ecological response, etc.) over a period of several years can be buffered by subsidence, because of the enhanced soil water retention. It is important to recognise, however, that such countering effects are not necessarily favourable to the ecology of a bog because capillary water flow to non-vascular mosses may become more difficult.

In addition to the effects of drainage, removal of the acrotelm significantly alters the water storage capacity of the peatland, notably through changes to Sy. In particular, specific yield changes because of 1) removal of the acrotelm where pores are large, and 2) because of peat degradation by oxidation and compression. Specific yield (Sy) is the ratio of the volume of water yielded by gravity drainage to the volume of the block of soil. In acrotelm peat with many large pores, specific yield is high because a large part of the pore space can drain by gravity. In peat with small pores (like catotelm peat, especially when degraded by harvesting) less water can be drained by gravity, thus its water retention capacity is higher. Thus, where Sy is low only a small amount of water needs to be drained from a soil to cause a large lowering of the water table. Price (1996) showed that removal of the acrotelm changed Sy from approximately 0.6 to 0.2; the latter value representing the Sy of peat in the catotelm that was exposed after the surface was cleared. Sy then declined to 0.04 to 0.06 within 5 years due to subsidence and oxidation. Joosten (1992) similarly recorded a decrease in the storage capacity of damaged peatlands relative to intact ones. Schouwenaars (1988 and 1993) also found that groundwater fluctuation in damaged bogs depends on the Sy of the upper peat layers and in particular, the pore size distribution, as affected by decomposition (Table 1). Figure 1 demonstrates the relationship between changes in water volume and changes in water table position for different peat layers in the Engbertdijksvenem raised bog in the Netherlands (Schouwenaars, 1988a). The consequence of this decrease in Sy is enhanced water table fluctuation (Price, 1996; Price, 1997; Money, 1995), a reduction in the proportion of time the water table is at the peat surface (Lindsay, 1988), and increased water retention in the unsaturated zone (Price, 1997). A summer water table of 40 cm below the ground surface is commonly accepted as a critical level for the growth of raised bog plant communities (Verry, 1988). Heathwaite (1994) found that the average summer water table position on Thorne Moors, a cutover raised bog in eastern England, was 70 cm below the winter water level. As a consequence, species such as *Betula* spp. may become dominant because they are better adapted to a widely fluctuating water table (Heathwaite, 1995). Schouwenaars (1990) found that, for Dutch bogs, deeper rooting species such as *Molinia* became dominant where the water table fluctuated widely. Presence of *Betula* and *Molinia* may be detrimental to *Sphagnum* colonization because of their water consumption and interception effects (see below).

When the water table drops sufficiently below the surface in damaged bogs (perhaps $>$ 50 cm below the ground level depending on peat type), groundwater ceases to contribute to atmospheric water losses, i.e. the water table no longer responds to evaporative losses, and soil moisture is instead depleted. Water in the soil withdraws into the smallest pore spaces, and is held at a correspondingly lower pore-water pressure. This pore-water pressure may be significantly lower than the threshold pore-water pressure of –100 mb at which *Sphagnum* cells drain (Hayward and Clymo, 1982). For example, Price (1997) noted pore-water pressure measured 2 cm below the surface in cutover peat at a mechanically block-cut site (Lac St-Jean, Québec) dropped to –355 mb during summer (1995). At an old abandoned manually block-cut site (Cacouna, Québec), a pore-water pressure of –170 mb was recorded in a trench in 1999 (Price and Whitehead, 2001), while in 1997 pore-water pressures of less than –600 mb were recorded on baulks (Price, unpublished data). Pore-water pressures of –156 mb were recorded on an old abandoned vacuumed site in 1999 (Price, unpublished data). Baird (unpublished data) recorded relatively modest falls in pore-water pressure at depths $>$ 30 cm in a drought summer (1989) in a drained fen peat in the Somerset Levels, southwest England (for details of the peat type see Baird, 1997). The lowest pore-water pressures on two sites at this location were –80 and –160 mb respectively, with pore-water pressures generally much higher than these values. Although pore-water pressures nearer the surface were not measured, it is likely they were much lower than –80 and –160 mb and therefore consistent with those of Price and Whitehead (2001). Price and Whitehead (2001) further conclude that even short periods of pore-water pressure $<$ –100 mb may limit or preclude *Sphagnum* establishment.

The changes to evapotranspiration caused by drainage and cutting are more difficult to characterise. Price (unpublished data) studied evaporation from a natural bog, an actively vacuum harvested site, and an abandoned cutover site with blocked ditches, all on the St. Marguerite peatland near Lac St. Jean, Québec. Net radiant energy (Q^*) was similar on each site, but

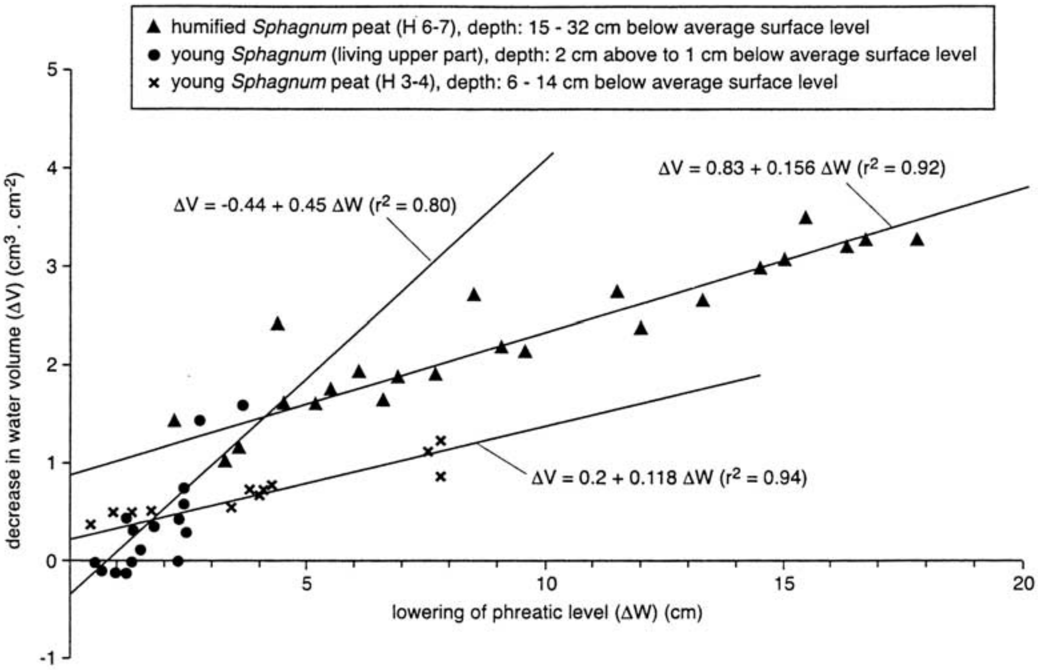


Figure 1. Relationship between changes in water volume and changes in water table position for different peat layers in a *Sphagnum*-dominated raised bog (after Schouwenaars, 1988a).

Table 2. Evaporation from undrained and drained peatlands, as affected by vegetation community (after Heathwaite, 1995).

Vegetation cover	Drainage status	Winter evapotranspiration (mm y^{-1})	Summer evapotranspiration (mm y^{-1})	Annual evapotranspiration (mm y^{-1})
Sphagnum	Undrained	140	410	550
Calluna	Slightly drained	125	395	520
Grassland	Drained	90	410	500
Arable	Drained	75	395	470
woodland	Drained	125	465	590

differences in ground heat flux (Qg) caused by large differences in soil thermal diffusivity (a function of moisture content) and ground temperature (cutover bogs froze deeply because there was little vegetation to trap snow), resulted in less energy available for convective fluxes at cutover sites (Figure 2). However, following snowmelt the saturated frozen peat of the abandoned reflooded site occasionally resulted in downward sensible heat flux, which boosted the evaporative loss (i.e. Qe flux) there. The drained vacuum harvested site, with lower water table, and dryer peat, had the lowest evaporation loss. Similarly, bogs converted to agricultural use have lower evapotranspiration losses, see Table 2 (after Heathwaite, 1995). In natural bogs lower evapotranspiration is also associated with a lower water table (Romanov, 1968).

Although it is unlikely that drainage of peatlands imposes significant water restrictions on transpiring

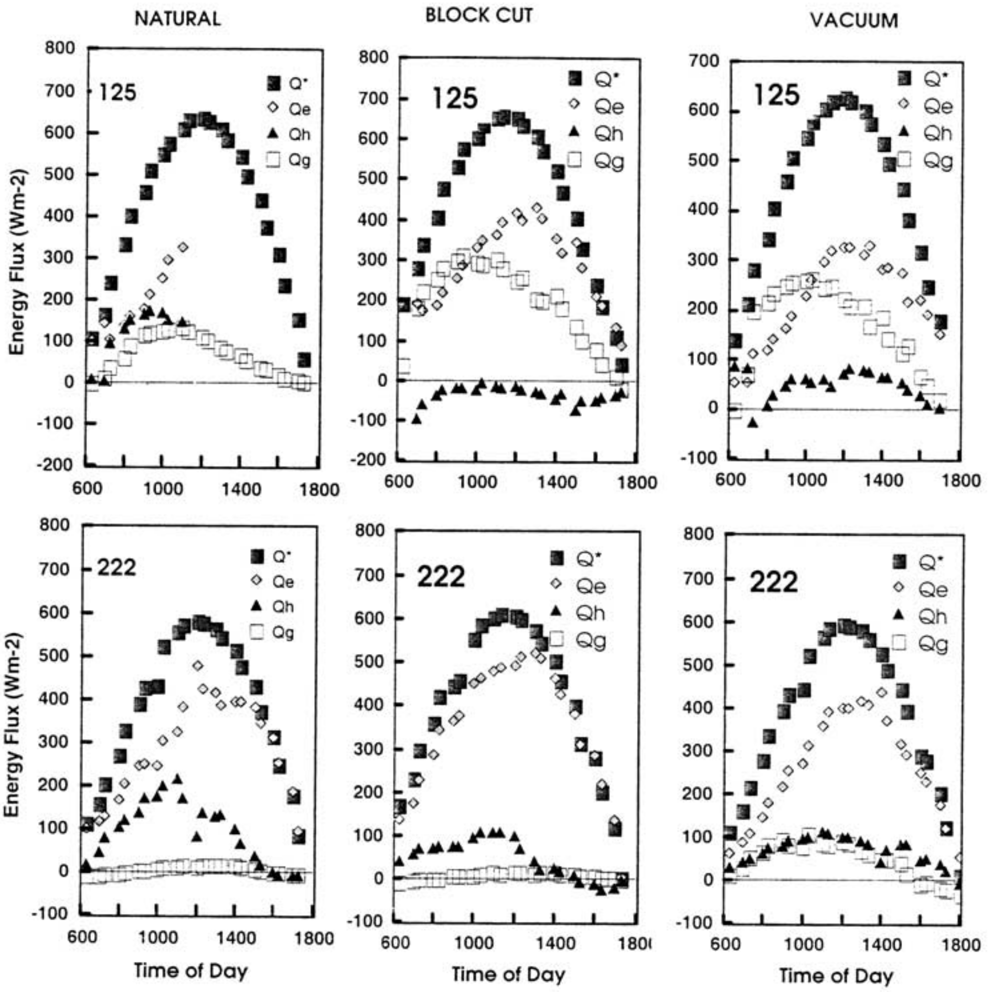


Figure 2. Energy flux density at a natural, machine block-cut (level surface) and vacuum harvested site on the Ste. Marguerite peatland near Lac St. Jean, Québec, on Julian days 125 (May 5) and 222 (Aug. 10) in 1994. Q* is net radiation, Qg is ground heat flux, Qh is sensible heat flux, and Qe is latent heat flux. Unpublished data from Price, 1994 (see Price, 1996 for methods).

vegetation, recent work by Birdsall (2001) found that the rate of evapotranspiration from species characteristic of UK cutover bogs (e.g. *Betula* and *Calluna)* was significantly lower at lower water table positions. Birdsall (2001) suggests that despite this water table control, evapotranspiration losses from invasive species such as *Betula* and *Calluna* are generally higher than from natural bog surfaces (see Table 2). Van Seters and Price (2001) estimated that invasive trees (*Picea mariana*) increased evapotranspiration losses by up to 25%. In addition to transpiration losses, unpublished data (D.E. Reid and A.J. Baird) from the Somerset Levels in southwest England indicate that interception losses under 20 year old *Betula* frequently (more than 30% of observations) exceed 30% (based on approximately weekly measurement intervals), and that water tables in soils under *Betula* fall considerably below (>20 cm) those under immediately adjacent *Molinia* grassland. The latter is especially significant, since it has been shown that *Molinia* enhances transpirative losses from cutover peatlands undergoing restoration (Schouwenaars, 1988). A similar figure (32%) was reported under *Picea mariana* by Van Seters and Price (2001) in an abandoned block-cut bog (Québec).

Seepage losses are reduced by the lower K, but the because of the low Sy, water table variability is greater (i.e. specific yield effects more than outweigh the effects of reduced K, because typically more water is lost via evapotranspiration than seepage). Therefore, during dry periods the water table depth in damaged bogs can be much greater than in undisturbed bogs (Price, 1996) even if ditches are blocked and the water balance essentially restored. Furthermore, the large water table drawdown can greatly enhance seasonal subsidence, which itself can accommodate about half the seasonal storage change (Price and Schlotzhauer, 1999). The high water retention capacity of residual cutover peat (Schlotzhauer and Price, 1999) is due to the smaller pore-size distribution (Okruszko, 1995; Shouwenaars, 1988). When the water table drops below 60 cm, evapotranspiration can draw on the soil moisture reserve without replenishment from the water table (Price, 1997). Unlike in natural bogs, therefore, changes in soil moisture content can comprise a significant component of storage change (Van Seters and Price, 2001). The consequence of this is lower pore-water pressure, and a reduced flow of water to mosses that may lead to their desiccation, and intolerable conditions for *Sphagnum* (Sagot and Rochefort, 1996).

Abandonment and natural recolonization

The gradual accumulation of organic material in a natural raised bog as it develops is accompanied by a rise in the water table, such that the system becomes hydrologically isolated from the surrounding upland. Lowering the surface by removal of the acrotelm and peat cutting is accompanied by a concurrent drop in the water table (in addition to drainage effects). This may alter the regional hydrogeological flow patterns that are sometimes delicately balanced between recharge and discharge (Siegel et al., 1995), and/or the relationship with undisturbed adjacent peatland (Heathwaite, 1994, 1995). Furthermore, where the bog surface is lowered by peat cutting, it is possible for ingress of enriched water from adjoining land to occur, (e.g. at Cumwhitton moss, Cumbria, UK (Ratcliffe, 1977). Attempts to restore functions to bogs where the hydrological system is not intact or has been substantially altered by peat cutting may be doomed to failure because hydrogeological conditions may be more typical of fens. For example, in the UK, Lindsay (1988) and Smart (1983) found evidence that in some sites fen vegetation recolonised abandoned flooded hollows where peat cutting had removed the lower peat layers and exposed the underlying clay. Under such circumstances, natural or planned restoration to a bog ecosystem may be inappropriate. Whilst absolute ombrotrophic chemical conditions are not a prerequisite for the persistence of many typical ombrotrophic plants (Wheeler, 1988), the degree of base-enrichment, and particularly nutrient enrichment, under which these species can grow is probably strictly limited. The depth of ombrotrophic peat needed to prevent recolonisation by fen vegetation is not known: Wheeler (1988) suggests it may be in the region of 50 cm. Retaining a peat layer of this thickness is extremely difficult where the mineral substrate and subsequent peat layers are undulating. In some circumstances, however, it may be possible to develop bog plant communities from an initial fen community. Giller and Wheeler (1988) identified acidifying (oxyphilous) nuclei in the rich-fen vegetation communities of the flooded nineteenth century peat pits (turf ponds) in the Norfolk Broads, eastern England. Here *Sphagnum* nuclei have developed on vertically mobile vegetation rafts suspended over base-rich water (see also Wheeler, 1999 and van Wirdum, 1991).

Following abandonment, the degraded peat cannot establish the necessary ‘self-regulation’ characteristic of intact bogs; thus low and variable water tables, low soil moisture contents and pore-water pressures persist for significant periods during summer (e.g. Price, 1997). Enhanced microbial activity may also hamper the development of ombrotrophic *Sphagnum* plant communities. The surface remains hostile for the establishment of *Sphagnum* propagules, and even where *Sphagnum* regrowth occurs, it remains vulnerable to water table lowering for many years.

As noted earlier, the hydrological conditions of abandoned vacuum-harvested peatlands are less conducive to natural regeneration than manually block-cut peatlands (Money, 1995). The reasons for this involve a complex array of interrelated biotic and abiotic factors that are not well understood. There have been few (if any) systematic studies designed to address this. The proximity of two abandoned cutover peatlands near Riviere-du-Loup, Québec, provide an opportunity to elucidate some of the processes. One was manually block-cut (Cacouna bog – abandoned c., 1970) and the other vacuum harvested (Bois-des-Bel – abandoned, 1976). Peat cutting covered 133 ha at Cacouna, and 12 ha at Bois-des-Bel. These bogs have undergone substantial degradation of the residual peat. This includes a 50 to 100% increase in bulk density, loss of storativity (Sy near the surface decreased from about 0.5 to 0.06±0.03), a decrease in hydraulic conductivity average of upper 1.5 m of peat (from 8.2×10^{-5} to 2.2×10^{-6} cm s^{-1} at Cacouna, and 5.7×10^{-6} cm s^{-1} at Bois-des-Bel), and a comparable change in water retention characteristics (Van Seters and Price, 2001; Price and Whitehead, 2001; and von Waldow and Price, unpublished data for Bois-des-Bel). The hydraulic structure of the peat is not greatly different between sites, in spite of the heavy machines that were used at the vacuum harvested site. The main difference is the topographic structure, from which has arisen a significantly different course of ecological recovery. Up to 15% of the block-cut bog had *Sphagnum* dominated surfaces in 1998 (Girard, 2000), whereas the vacuum harvested site was devoid of *Sphagnum* (Lavoie and Rochefort, unpublished data for Bois-des-Bel) and had very little vegetation cover at all. Water tables, pore-water pressures and soil moisture were more spatially variable on the block-cut site. At the vacuum harvested site the hydrological variables were generally less extreme (i.e. than in baulks of the block-cut site), but typically worse (e.g. drier, lower pore-water pressure) than in the trenches of the block-cut site. It is suggested that it is this spatial variability at the block-cut sites that enables some (albeit few) areas to serve as loci for *Sphagnum* regeneration. The presence of *Sphagnum* cushions on cutover peat surfaces raises the volumetric moisture content and pore-water pressure in the substrate. In addition, where the water table is near the surface, Sy in these *Sphagnum* regenerated areas increases to values similar to natural bogs, thus assisting in stabilizing the water table (Price and Whitehead, 2001). Over time *Sphagnum* accumulation in trenches raises the water table and can generate 100% cover within the trenches. Much longer is required to accumulate sufficient organic material to cover baulks and roads, which can represent >30% of the peatland area. Price and Whitehead (2001) noted that only areas of the block-cut bog where pore-water pressures were consistently above −100 mb and volumetric water content above 50% were associated with *Sphagnum* regeneration. There were no such areas meeting these criteria in the vacuum-harvested boa. Where the surface is fairly regular, such as in vacuum-harvested bogs, the seasonal water deficit, especially prominent in continental areas, creates unsuitable conditions everywhere for *Sphagnum* recolonization. It is in the block-cut peatland, where the surface is highly variable, that some suitable locations are found. There are also some highly unsuitable areas (e.g. baulks) that are unlikely to ever become loci for *Sphagnum* colonization. However, lateral expansion onto baulks from 'infilled' trenches has been observed near St. Henri, Québec (Price, pers. obs., Oct., 2000). The role of baulks is not clearly understood. Price and Whitehead (2001) did not observe a groundwater mound beneath baulks, as one might expect beneath a raised topographic feature. Interception and transpiration losses by recolonised trees, or by trees that pre-date abandonment (large trees were not always removed) apparently promoted the water table drawdown beneath them. Nevertheless, there is some shading from the baulk and overhanging ericaceae that may have an important microclimatic effect.

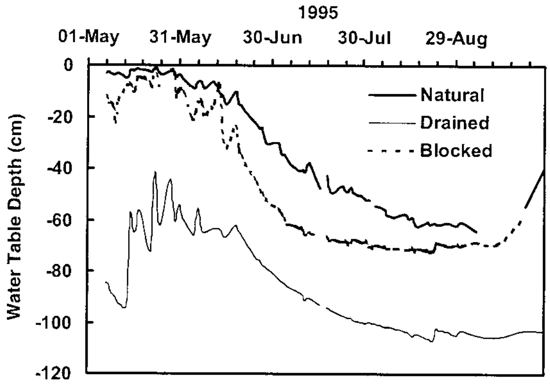


Figure 3. Water table depth at a natural (undisturbed), drained (cutover), and drained (cutover) site with ditches blocked; all on the Ste. Marguerite peatland, near Lac St. Jean, Québec (modified from Price, 1997).

Water management for restoration

In most cases, the fundamental problem in peatland restoration is a lack of water caused by drainage, compounded by the instability of water storage during the growing season caused by rainfall and evaporation from the low Sy peat. Various approaches have been used to minimise the duration of unsuitably low water table conditions, including blocking ditches (Eggelsmann, 1988), using bunds or polders to retain water (Eggelsmann and Blankenburg, 1993), providing open water reservoirs to increase passive lateral seepage (Beets, 1992; LaRose et al., 1997; Schlotzhauer and Price, 1999) or pumped lateral seepage (Price, 1998), altering the surface microtopography (Ferland and Rochefort, 1997; Quinty and Rochefort, 1996; Price et al., 2002), and using shading devices (Rochefort and Bastien, 1998), straw mulches (Quinty and Rochefort, 1996; Price, 1997) or companion species (Marcoux, 2000; Ferland and Rochefort, 1997). A thorough review of many of these methods is given in Wheeler and Shaw (1995). Below is an evaluation of the main approaches – 1) blocking ditches, 2) using bunds or terraces to retain water, 3) establishing hydrological buffer zones, 4) surface reconfiguration, and 5) reducing evaporation losses.

1. Blocking ditches

The efficacy of drainage is related to the depth of ditching, distance between ditches, and the hydraulic conductivity of the peat (Boelter, 1972; Armstrong, 2000). The drawdown is greatest near the ditch, and often diminishes quickly with distance. For example, Prevost et al. (1997) found that drainage was most effective within 15 m of ditches in a drained tree-covered bog near Riviere-du-Loup, Québec. Boelter (1972) found ditches were effective at up to 50 m in fibric peat in Minnesota, but ineffective beyond 5 m in more decomposed peat where the hydraulic conductivity was lower. Similar restrictions on drain impact exist for other peat types. For example Heathwaite (1991) found ditch drawdown was minimal in the decomposed fen peats of West Sedgemoor in the Somerset Levels, in southwest England. Blocking ditches is typically the first and most obvious management prescription. Numerous studies have shown it markedly raises the water table (e.g. Figure 3).

The degree of re-wetting that is possible by blocking or damming ditches can be illustrated with a simple unpublished water table model (developed from that presented in Baird et al., 1998), based on the Boussinesq equation (see Freeze and Cherry, 1979). Details of a similar model can also be found in Gilman (1994, pp. 67–71) (see also Armstrong, 2000). In applying the model it is assumed (i) that flow to a ditch is largely horizontal (i.e. vertical flows are ignored), and (ii) that water table dynamics can be accurately described using the concept of specific yield. The peat is assumed to be 1 m thick and to lie above a relatively impermeable substratum (e.g. clay). Peat K and Sy are assumed to be 2×10^{-4} cm s^{-1} and 0.15 respectively. The modelled ditch extends to the base of the peat. Two scenarios are considered. In the first, the ditch water level is maintained at a height 20 cm above the ditch base. In the second the ditch is assumed to be dammed/blocked at its outlet, so that water levels in it are maintained 70 cm above the ditch base, i.e. water can flow over the dam but cannot leak through it.

Water table dynamics were modelled for a period of 15 days from a start condition in which the water table was everywhere 10 cm below the ground surface. Dunng the 15-day period there were no net exchanges of water with the atmosphere. Although rather artificial, the simulations are useful in showing the magnitude of effects imposed by drainage and the effects of ditch damming. The spatial pattern of drawdown is shown in Figure 4a, where the more dramatic drawdown in the vicinity of the open ditch can be seen. Despite the clear effect of the open ditch, the drawdown is relatively small at modest distances ($>$ 10 m) from it. The time rate of drawdown is also much greater in the drained ditch (Figure 4b), with water tables dropping by over 27 cm within 2.5 m of the ditch and more than 20 cm 5 m from the ditch. In contrast, the blocked ditch simulation reveals a much more modest water table drop, with levels falling less than 6 cm for all areas $>$ 5 m from the ditch. Thus, it would appear that the ditch has little drainage effect on water tables except in its immediate vicinity. However, it is important to note that even small changes in water tables can have significant ecological effects. For example, water table drawdown could allow tree colonisation and subsequent drying due to interception losses (see ‘Hydrological Changes’ above).

2. Bunds and terracing

In peatlands where there is a notable surface slope, and in rather dry sites, the use of bunds and terraces offer several important advantages over blocking of ditches. While ditches blocked at appropriate intervals raise the water table and help reduce runoff during dry and moderately wet conditions, they are ineffective in preventing water losses during rain or snowmelt events that produce surface runoff. Bunds constructed along contour lines can reduce most surface runoff, and when prepared in conjunction with site grading can produce level terraces that promote higher and more evenly distributed water table and moisture conditions (Figure 5). During dryer conditions, when the water table drops below the surface, the bunds have no appreciable effect on water retention, although water table differences between terraces persist throughout the summer. The retention of water during the wet season has a profound effect on the minimum summer water table (Figure 6).

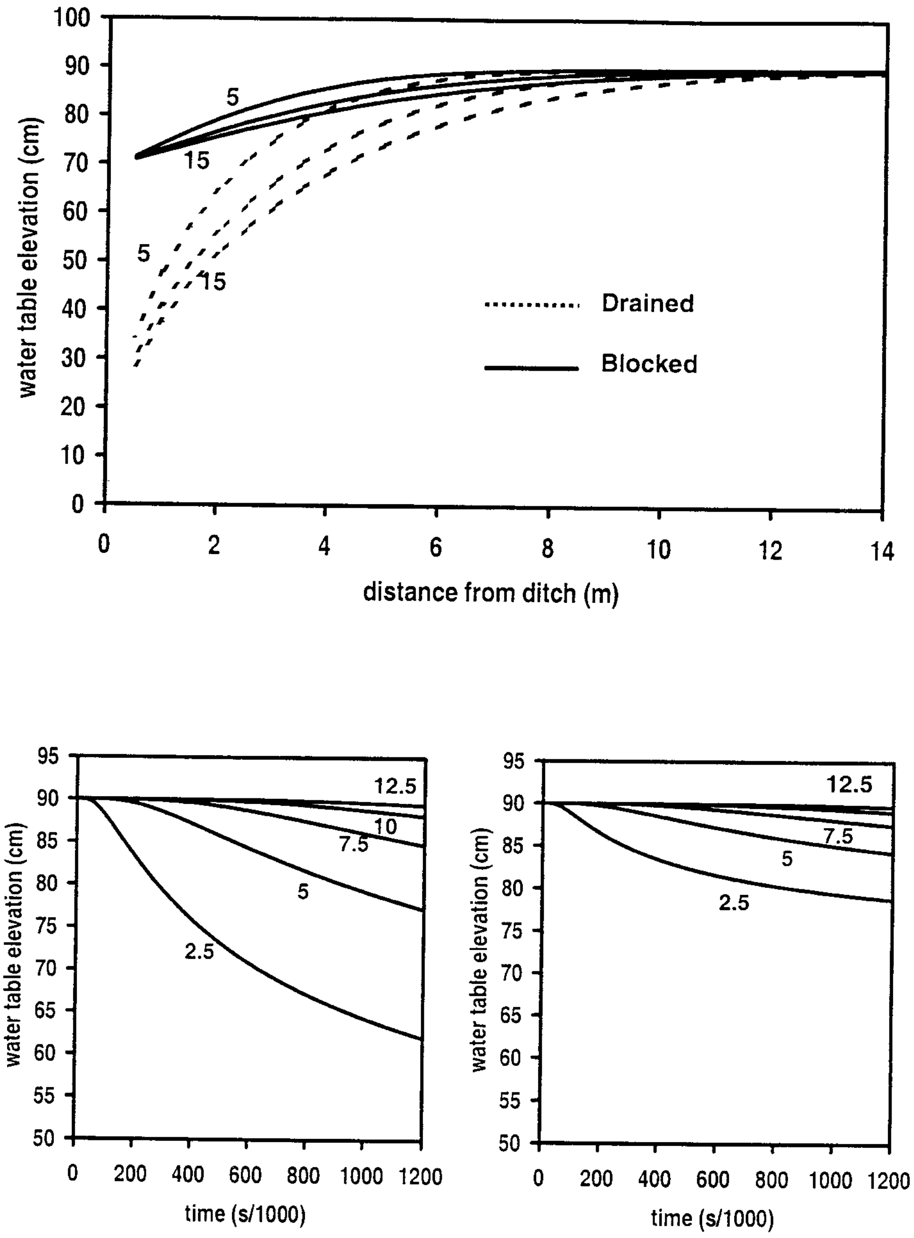
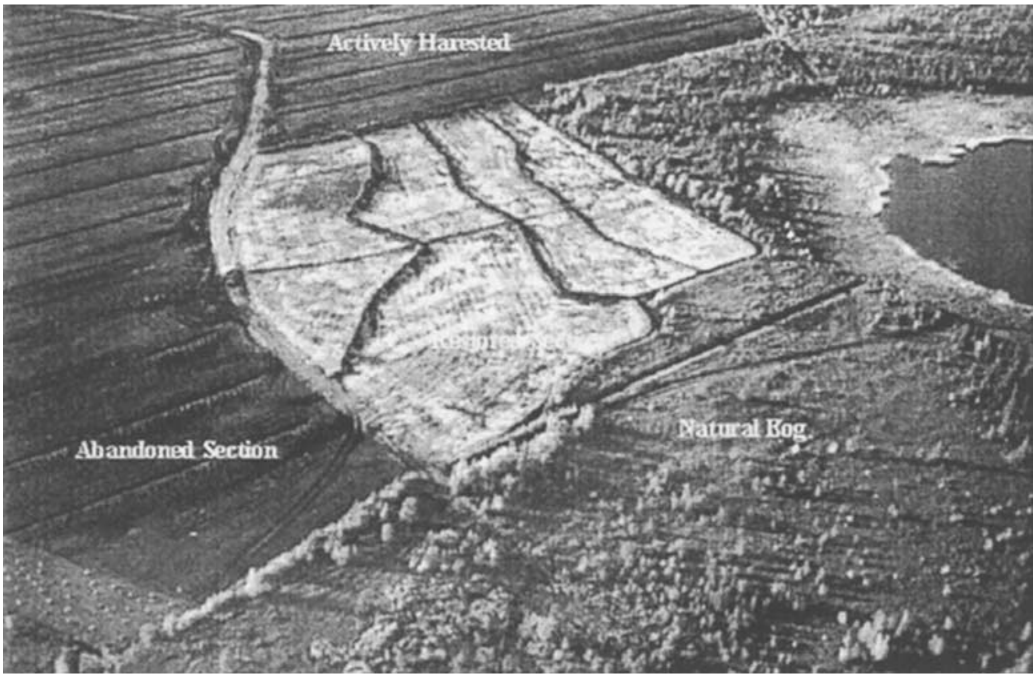


Figure 4. a) Temporal patterns of drainage for positions 2.5, 5, 7.5, 10 and 12.5 m from the ditch edge for the unblocked (left) and blocked (right) ditch, b) Spatial patterns of drawdown away from the unblocked (above) and blocked (below) ditch after 5, 10, and 15 days of drainage.

a)



b)

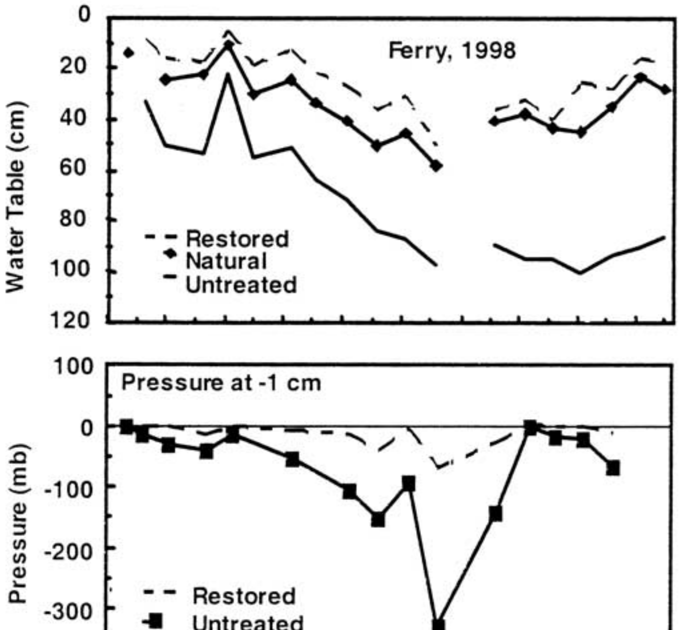


Figure 5. a) Aerial view of bunded peatland at Inkerman Ferry, New Brunswick (top), b) average water table (centre) and c) tension (lower).

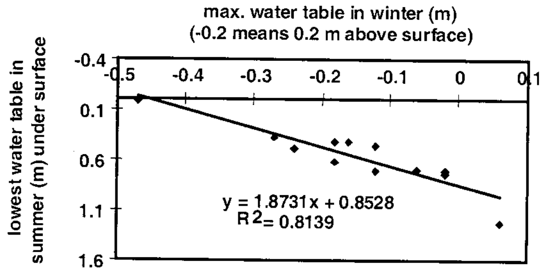


Figure 6. The maximum water table in winter is plotted against the lowest water table recorded the following summer, at Leegmoor, Germany.

The efficient water retention at bunded peatlands may create problems with 'excess' water at certain times of the year. Extensive periods of inundation are favorable to *S. cuspidatum* – a water loving species that thrives in pools, and to *S. angustifolium* (Chirino et Rochefort, unpublished data) and *S. fallax*. However, these conditions are generally unfavourable to species adapted to dryer conditions like *S. fuscum* or *S. capillifolium* (Chirino and Rochefort, unpublished data; Campeau, Rochefort and Price, unpublished data). Erosion and sedimentation problems can also be problematic when inundation occurs. At Leegmoor, Germany, *Eriophorium vaginatum* was established to stabilise the surface against such problems, and once established, the water level in the polders was increased to create conditions favourable to *Sphagnum* (Eggelsmann and Blankenburg, 1993).

3. Buffer zones

Eggelsmann (1988) argues that regeneration of damaged bogs is possible only if the complete hydrological system of the bog is preserved – this includes land surrounding the bog, even where it has been converted to agricultural use, for example. Significant problems arise when restoration is attempted alongside active peat cutting, because of ongoing restrictions with regard to drainage. In order to reconcile such a situation at Thorne Moors in eastern England, a double-ditch pumping system was used (Heathwaite, 1994). The data shown in Figure 7 demonstrate the principles of the double-ditch approach and also show some of its failings. Two factors limited the success of the strategy. First, there was an insufficient reservoir of water available to maintain adequate water levels in the outer (donor) ditch. Secondly, many cross-drains from the original peat workings, now infilled with *Sphagnum,* caused lines of weakness (higher K) in the peat bund separating the inner (receptor) ditch from the outer (donor) ditch, increasing water flow from inner to outer ditch in response to the altered hydraulic gradient caused by pumping. Since this initial pumped ditch design was tested at Thorne Moors it has been shown that it is possible, with careful water management, to raise and maintain water tables in restricted sections of the peatland with judicious use of ditch blocking, bunding and regular pumping. This approach was aided because the width of available hydrological buffer was larger than originally planned. Furthermore, the initial double-ditch design was tested when the average annual rainfall was considerably below the long-term average (Heathwaite, 1995) in the early 1990s; since then rainfall input has been closer to or above the average. The work by Heathwaite (1994, 1995) demonstrates that on a small scale water pumped from a reservoir within a peatland may provide continuously wet conditions to a restoration site. However, there is generally an inadequate supply of ombrotrophic water to nourish larger areas and it was concluded that larger buffer strips are necessary (Heathwaite, 1994).

4. Surface reconfiguration

At sites where mechanical peat cutting has left extensive tracts of relatively uniform cutover peat, the hydrological conditions are uniformly poor, as described previously. Small changes in surface elevation or texture can increase the variability of near surface microclimate. For example, bulldozer tracks and plough furrows in cutover peat near Lac St. Jean, Québec, became more suitable sites for *Sphagnum* recolonization than the untreated surface because of important microclimatic differences (Price et al., 1998). However, the corresponding positive relief elements became much less favourable for *Sphagnum* (Price et al., 1998). Shallow basins bulldozed into the peat at this location did provide enhanced site wetness (Price et al., 2002). A positive hydrological and ecological response was also recorded near Rivière-du-Loup, Québec, to a reversal of the domed cross-sectional profile of a vacuum harvested peat field (Bugnon et al., 1997).

5. Companion species and mulch for microclimate and evapotranspiration control

The issue of the role of companion species has caused some controversy in peatland management. Under dry

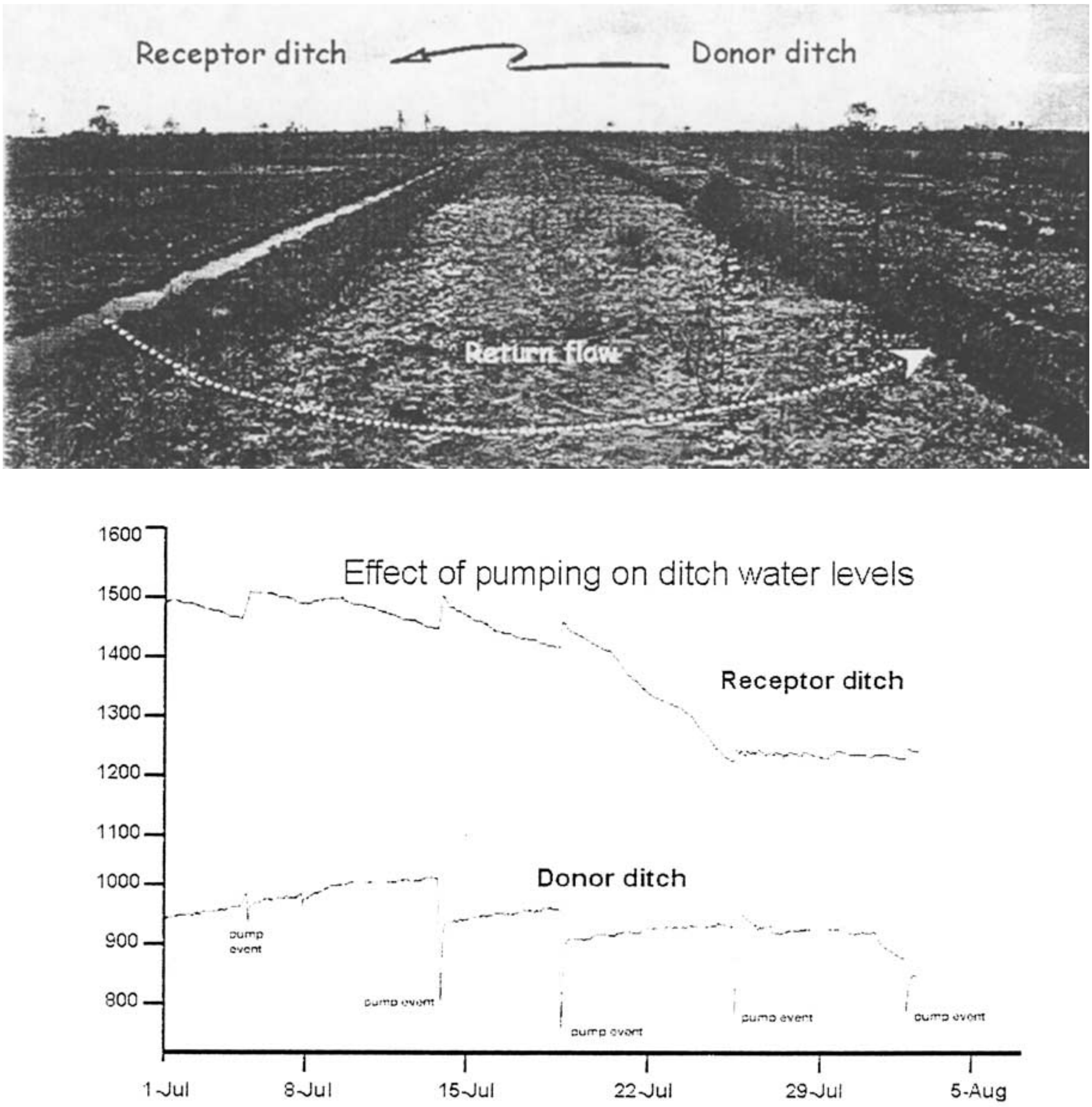


Figure 7. Water is pumped from the donor ditch located in the cutover and actively drained section of peatland (right), to the buffer zone of the restoration site (left). As water is pumped from donor to receptor ditch, the hydraulic gradient between them increased, increasing the return groundwater flow away from the restoration site.

soil conditions, vascular plants can enhance evaporative losses from a site (Lafleur, 1990; Spieksma et al., 1997), but the losses are from deeper in the peat profile, thus may not necessarily be incompatible with reestablishment of *Sphagnum* mosses, which access water directly from the surface of the cutover peat in addition to direct precipitation. In contrast, other studies (e.g. Marcoux, 2000) showed that shelter and shading by *Eriophorum* spp. can increase relative humidity near the surface, and decrease the evaporation loss compared to a bare peat site, thus promoting higher volumetric soil moisture contents (Figure 8). The role of shade and shelter was also observed at the abandoned block-cut bog at Cacouna, where *Sphag-*

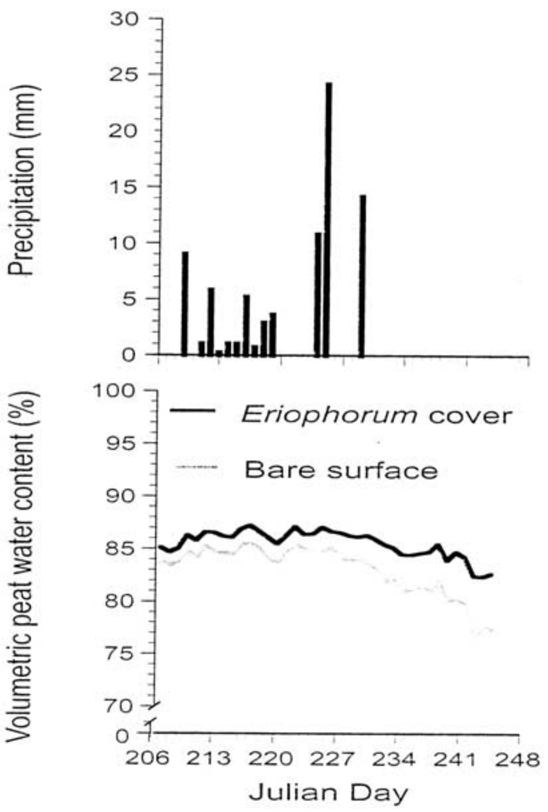


Figure 8. Precipitation (mm) and volumetric peat water content (%) 0–12 cm beneath an *Eriophorum vaginatum* cover, and a bare surface in Saint-Henri mined peatland, Québec (after Marcoux, 2000).

Table 3. Average daily net radiation (Q*), ground (Qg) and latent heat (Qe) flux (W m^{-2}), and evaporation (E, mm) from a bare peat surface, and a similar surface covered with straw mulch (2250 kg ha^{-1}). Measurements are from Lac St. Jean peatland, Québec, between June 3 and Oct 13, 1995. Revised from Price et al. (1998).

	Q*	Qg	Qe	E
Bare peat	128.2	16.4	88.3	3.1
Mulch covered	112.0	2.2	74.2	2.6

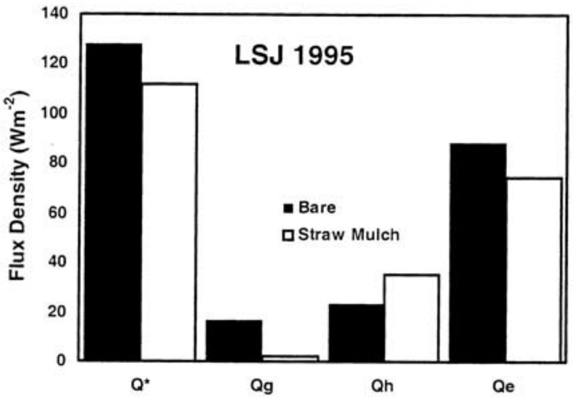


Figure 9. Energy flux density over a bare and straw mulch covered cutover peat at Lac. St. Jean, Québec, 1995. Q* is net radiation, Qg is ground heat flux, Qh is sensible heat flux, and Qe is the latent heat flux. Metholological details are given by Price et al. (1998).

num hummocks recolonise under low-lying spruce boughs and ericaceae even without good contact with the bare cutover peat (Price, personal observations from Cacouna Station bog). These hummocks quickly become desiccated when ericaceae or the supporting tree is removed. This suggests that some *Sphagnum* cushions can survive and grow, relying only on direct precipitation and water stored within the moss matrix, as long as shading reduces the evaporative loss.

While the use of companion species has been shown to improve conditions significantly for *Sphagnum* recolonization (Boudreau and Rochefort, 1998; Grosvemier et al., 1995; Marcoux, 2000), artificial shading with straw mulch can perform this role without the transpiration loss associated with vascular plants (Table 3). Price et al. (1998) noted that straw mulch could intercept about 2 mm of rain, but that the overall benefit of mulch on soil moisture and porewater pressure was substantial (Figure 9). This involved a reduction in net radiation, probably due to albedo effects; a substantial decrease in Qg, resulting in a cooler peat surface; and an increase in Qh, presumably because the relatively dry straw surface is more readily heated. These processes, coupled with sheltering and shading against wind resulted in higher relative humidity within the mulch, and thus lower Qe from the peat below.

Conclusion

Peat mining cannot be considered sustainable within the time frame of resource extraction and use. While the volume of peat mined annually is a small fraction of the global annual production on natural systems, local impacts are profound. Due consideration of the hydrological properties and processes of the impacted system can do much to soften the duration of impacts, so should become part of the planning process in the early stages. For example, the depth of cutting, duration of operations, and method of extraction have a long-lasting effect on the quality of the substrate, and in some cases the hydrogeological relations of the peatland within the broader landscape. Consequently, the ability to provide a suitable base for reestablishment of the original hydrological and ecological functions can, to a certain extent, be controlled. In conjunction with appropriate water management after abandonment, many of the original peatland functions can be recreated.

Drainage of the peat and removal of the acrotelm cause irreversible changes in the hydraulic properties of the residual peat that are unfavourable to the re-establishment of *Sphagnum* mosses. The lower hydraulic conductivity and higher water retention capacity may limit the water availability to non-vascular plants. At the same time, however, these same properties will restrict the seepage losses to the substrate, provided that a sufficiently thick layer of peat is left. Alternatively, deeper cutting may induce mineratrophic discharge from the regional flow system, which is detrimental to bog flora. Only by understanding the hydrogeological relations of the bog within the landscape, the hydraulic properties of the peat, and how they change when mining occurs, can effective management be implemented.

The change in the water storage properties of peat, along with the drainage ditch network, significantly affect the amount and nature of water storage on the site. Blocking the ditches can significantly restore the summer water budget of a drained cutover peatland, but the water storage processes of the residual peat do not replicate that of the acrotelm. Consequently, the summer water deficit that occurs in most seasons frequently results in pore-water pressures far below –100 mb, which field and laboratory studies suggest is necessary to ensure an adequate supply of water to non-vascular *Sphagnum* mosses. In many cases, therefore, it is necessary to provide additional water during the growing season, by retaining more winter rainfall or snowmelt water. This requires a more aggressive approach to restoration than simply blocking ditches. Creating terraces or polders surrounded by bunds, is one such method that can retain sufficient water to avoid excessive drying in summer. Shallow basins bulldozed into the peat have a similar effect. A less aggressive approach, which can be used in combination with the above methods, or sometimes alone, is the use of companion species (e.g. *Eriophorum*) or a straw mulch, whose shading and shelter reduce evaporation losses or otherwise protect *Sphagnum* diaspores on the surface. Detailed descriptions of the North American approach to restoration are given by Quinty and Rochefort, 1997: and Rochefort et al., this issue, while a description of the European and Australasian approaches are noted in Vasander et al. (this issue), and Whinam et al. (this issue), respectively.

Observations of old abandoned peatlands indicate that manually block-cut peatlands with a topography of alternating baulks and trenches more readily recover than vacuum harvested sites. Microtopographic minima may serve as loci for *Sphagnum* recolonization, which once established, progressively ameliorates the adjacent environment, and thus facilitates its own propagation. Vacuum harvesting leaves a landscape more suited to shedding surface water (cambered surfaces), with very little microtopography. Consequently, conditions are uniformly poor across the site, and loci for *Sphagnum* recolonization are generally absent. Furthermore, milled peat left on the surfaces of vacuum harvested sites dries quickly, and impedes capillary water flow to the surface, and to any *Sphagnum* propagules that may have arrived there. It should be noted that the huge areal extent of most vacuum harvesting operations decreases the likelihood of propagules arriving. While vacuum harvesting has proven to be an effective method for peat extraction, it has exacerbated the difficulty of restoration. Until such time as less invasive extraction techniques become common, managers must make the best of a bad situation. Our observations on block-cut peatlands, and with experiments on microtopography suggest that increasing the variety of micro-habitats is an important option for restoration. This must be done in conjunction with a peatland-scale effort to retain more water on-site, as described above.

References

Armstrong, A.C. 2000. DITCH: A model to simulate field conditions in response to ditch levels managed for environmental aims. Agric. Ecosyst. & Envir. 77(3): 179–192.

Baden, W. and Eggelsmann, R. 1968. The hydrologic budget of the highbogs in the Atlantic regions. Proceedings of the 3rd International Peat Congress, pp. 206–211, Québec, Canada.

Baird, A.J. 1997. Field estimation of macropore functioning and surface hydraulic conductivity in a fen peat. Hydrol. Proc. 11: 287–295.

Baird, A.J., Beckwith, C.W. and Heathwaite, A.L. 1997. Water movement in undamaged blanket peats. *In:* Tallis, J.H., Meade, R. and Hulme, P.D. (eds.), Blanket Mire Degradation: Causes, Consequences and Challenges, pp. 128–139, Macaulay Land Use Research Institute, Aberdeen.

Baird, A.J., Mason, T. and Horn, D.P. 1998. Validation of a Boussinesq model of beach ground water behaviour. Marine Geol. 148: 55–69.

Bay, R.R. 1969. Runoff from small peatland watersheds. J. Hydrol. 9: 90–102.

Beets, C.P. 1992. The relation between the area of open water in bog remnants and storage capacity with resulting guidelines for bog restoration. *In:* Bragg, O.M., Hulme, P.D., Ingram, H.A.P. and Robertson, R.A. (eds.), Peatland Ecosystems and Man: An Impact Assessment, pp. 133–140, International Peat Society/Department of Biological Sciences, University of Dundee, Dundee, UK.

Birdsall, K.R. 2001. Evapotranspiration from a Damaged Raised Mire: Water Table and Vegetation Control. Unpublished PhD thesis, University of Sheffield, UK.

Boelter, D.H. 1965. Hydraulic conductivity of peats. Soil Sci. 100: 227–231.

Boelter, D.H. 1972. Water table drawdown around an open ditch in organic soils. J. Hydrol. 15: 329–340.

Boudreau, S. and Rochefort, L.L. 1998. Restoration of post-mined peatlands: Effect of vascular pioneer species on *Sphagnum* establishment. *In:* Malterer, T.J. Johnson, K. and Stewart, J. (eds.), Proc. International Peat Society, International Peat Symposium, Peatland Restoration and Reclamation: Techniques and Regulatory Considerations, pp. 39–43, Duluth, Minnesota.

Bugnon, J.-L., Rochefort, L. and Price, J.S. 1997. Field experiment of *Sphagnum* reintroduction on a dry abandoned peatland in eastern Canada. Wetlands 17: 513–517.

Burke, W. 1972. Aspects of the hydrology of blanket peat in Ireland. *In:* Unknown (eds.), Hydrology of Marsh Ridden Areas, pp. 171–183, Proceedings of the Minsk Symposium, UNESCO-IAHS, Paris.

Burt, T.P., Heathwaite A.L. and Labadz, J.C. 1990. Runoff from peat covered catchments. *In:* Anderson, M.G. and Burt, T.P. (eds.), Process Studies in Hillslope Hydrology, pp. 463–501, John Wiley and Sons, Chichester.

Chason, D.B. and Siegel, D.I. 1986. Hydraulic conductivity and related physical properties of peat, Lost River Peatland, Northern Minnesota. Soil Sci. 142: 91–99.

Clymo, R.S. and Hayward, P.M. 1989. The Ecology of *Sphagnum. In:* Heal, O.W. Perkins, D.F and Brown, W.M. (eds.), Bryophyte Ecology, pp. 229–288, Chapman and Hall, London.

Conway, V.M. and Millar, A. 1960. The hydrology of some small peat-covered catchments in the N. Pennines. J. Inst. Water Engin. 14: 415–424.

CSPMA, 2002. Canadian Peat Harvesting and the Environment, http://www.peatmoss.com/pmme3.html, Canadian Sphagnum Peat Moss Association.

Eggelsmann, R.R.F. 1988. Rewetting for protection and renaturation/regeneration of peatland after or without peat winning. Proceedings of the Eighth International Peat Congress, Leningrad, 2, pp. 251–260.

Eggelsmann, R. and Blankenburg, J. 1993. Moor und wasser – Leegmoorprojekt, Naturschutz Landschaftpfl. Niedersachs., Heft 29: 19–48.

Famous. N.C., Spencer, M.S, and Nilsson, H. 1991. Revegetation patterns in harvested peatlands in central and eastern North America. *In:* Grubich, D.N. and Malterer, T.J. (eds.), Peat and Peatlands: The Resource and Its Utilization, pp. 48–66, Proceedings of the International Peat Symposium, Duluth.

Ferland, C. and Rochefort, L. 1997. Restoration techniques for *Sphagnum* dominated peatlands. Can. J. Botany 75: 1110–1118.

Freeze, R.A. and Cherry, J.A. 1979. Groundwater, Prentice Hall, Englewood Cliffs, New Jersey. p. 604.

Giller, K.E. and Wheeler, B.D. 1988. Acidification and succession in a floodplain mire in the Norfolk Broadland, UK. J. Ecol. 76: 849–866.

Gilman. K., 1994. Hydrology and Wetland Conservation. John Wiley and Sons, Chichester, p. 101.

Girard, M. 2000. The Natural Regeneration of Highly Disturbed Wetlands: Mined Peatlands of Southern Québec. M.Sc. Thesis. Départment Geography, Université Laval, Sainte-Foy, Québec, Canada.

Gorham, E. 1991. Northern peatlands: Role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. 1: 182–195.

Groeneveld, E. 2002. Le polytrichum strictum comme stabilisateur de substrat et plant compagne pour les sphaignes dans la restauration des tourbières exploitées par aspiratuer. M.Sc. Thesis, Départment de Phytologie, Université Laval, Sainte-Foy, Québec, Canada.

Gottlich, Kh., Richard, K.-H., Kuntz, H., Eggelsmann, R., Gunther, J., Eichelsdorfer, D. and Briemle, G. 1993. Mire Utilization. *In:* Heathwaite, A.L. and Gotlich, Kh. (eds.), Mires: Processes, Exploitation and Conservation, pp. 417–484, John Wiley and Sons, Chichester.

Grosvernier, Ph., Matthey, Y. and Buttler, A. 1995. Microclimate and physical properties of peat: New clues to the understanding of bog restoration processes. *In:* Wheeler, B.D. Shaw, S.C., Fojt, W. and Robertson, R.A. (eds.), Restoration of Temperate Wetlands, pp. 435–450, John Wiley and Sons, Chichester.

Hayward, P.M. and Clymo, R.S. 1982. Profiles of water content and pore size in *Sphagnum* peat and their relation to peat bog ecology. Proceedings of the Royal Society of London B215: 299–325.

Heathwaite, A.L. 1991. Solute transfer from drained fen peat. Water, Air & Soil Poll. 55: 379–395.

Heathwaite, A.L. 1993. Disappearing peat: regenerating peat? The impact of climate change on British peatlands. Geogr. J. 15: 203–208.

Heathwaite, A.L. 1994. Hydrological management of a cutover peatland. Hydrol. Proc. 8: 245–262.

Heathwaite, A.L. 1995. The impact of disturbance on mire hydrology. *In:* Hughes, J.M.R. and Heathwaite, A.L. (eds.), Hydrology and Hydrochemistry of British Wetlands, pp. 401–417, Wiley, Chichester.

Heathwaite, A.L., Eggelsmann, R. and Gottlich, Kh. 1993. Ecohydrology, mire drainage and mire conservation. *In:* Heathwaite, A.L. and Gotlich, Kh. (eds.), Mires: Processes, Exploitation and Conservation, pp. 417–484, John Wiley and Sons, Chichester.

Hoag, R.S. and Price, J.S. 1995. A field scale natural gradient solute transport experiment in peat at a Newfoundland blanket bog. J. Hydrol. 172: 171–184.

Ingram, H.A.P. 1978. Soil layers in mires: Function and terminology. J. Soil Sci. 29: 224–227.

Ingram, H.A.P. 1983. Hydrology. *In:* Gore, A.J.P. (ed.), Ecosystems of the World 4A: Moores: Swamp, Bog, Fen and Moor, pp. 67–224, Elsevier Scientific, Amsterdam.

Ingram, H.A.P. 1992. Introduction to the ecohydrology of mires in the context of cultural perturbation. *In:* Bragg, O.M., Hulme, P.D., Ingram, H.A.P. and Roberstson, R.A. (eds), Peatland Ecosystems and Man: An Impact Assessment, pp. 67–93, British Ecological Society/International Peat Society, Dundee.

Ivanov, K.E. 1981. Water Movement in Mirelands. Thomson, A. and Ingram. H.A.P. (Tr.) Academic Press, London, p. 276.

Joosten, J.H.J. 1992. Bog regeneration in The Netherlands: A review. *In:* Bragg, O.M., Hulme, P.D., Ingram, H.A.P. and Roberstson, R.A. (eds), Peatland Ecosystems and Man: An Impact Assessment, pp. 367–373, British Ecological Society/International Peat Society, Dundee.

Joosten, J.H.J. 1995. Time to regenerate: Long-term perspectives of raised bog regeneration with special emphasis on paleoecological studies. *In:* Wheeler, B.D. Shaw, S.C. Fojt, W. and

Robertson R.A. (eds.), Restoration of Temperate Wetlands, pp. 379–404. John Wiley & Sons, Chichester.

Lafleur, P.M. 1990. Evapotranspiration from sedge-dominated wetland surfaces. Aq. Botany 37: 341–353.

Lambert, J.M., Jennings, J.N., Smith, C.T., Green, C. and Hutchinson, J.N. 1960. The Making of the Broads: A Reconsideration of Their Origin in the Light of New Evidence, Royal Geographical Society Research Series No. 3, Royal Geographical Society, London.

LaRose, S., Price, J.S. and Rochefort, L. 1997. Rewetting of a cutover bog. Hydrological Assessment. Wetlands 17: 416–423.

Lavoie, C., Grosvernier, P., Girard, M. and Marcoux, K. this issue. Spontaneous revegetation of mined peatlands: an useful restoration tool? Wetl. Ecol. & Manag.

Lavoie, C. and Rochefort, L. 1996. The natural vegetation of a harvested peatland in southern Québec: A spatial and dendroecological analysis. Ecoscience 3: 101–111.

Lindsay, R. 1988. The raised mires of Britain: a conservation perspective. *In:* Fojt, W. and Meade, R. (eds), Cutover Lowland Raised Mires, pp. 15–21. Nature Conservancy Council publication No. 24, Peterborough.

Marcoux, K. 2000. Les invasions biologiques: aide ou frein à la restauration des tourbières (Biological invasions: Do they Help or Impede Peatland Restoration?). M.Sc. Thesis, Université Laval, Québec.

Mikulski, Z. and Lesniak, E. 1975. Hydrological research on a peatbog in the upper Suprasl basin. *In:* Hydrology of Marsh Ridden Areas, Proceedings of the Minsk Symposium, pp. 55–67, The UNESCO Press, Paris.

Money, R.P. 1995. Re-establishment of a *Sphagnum* dominated flora on cut-over lowland raised mires. *In:* Wheeler, B.D. Shaw, S.C. Fojt, W. and Robertson, R.A. (eds.), Restoration of Temperate Wetlands, pp. 405–422, John Wiley & Sons, Chichester.

Mulqueen, J. 1989. Hydrology and drainage of peatland. Envir. Geol. & Water Sci. 9: 15–22.

Nicholson, I.A., Robertson, R.A. and Robinson, M. 1989. The effects of drainage on the hydrology of a peat bog. Int. Peat J. 3: 59–83.

Okruszko, H. 1995. Influence of hydrological differentiation of fens on their transformation after dehydration and on possibilities for restoration. *In:* Wheeler, B.D. Shaw, S.C. Fojt, W. and Robertson, R.A. (eds.), Restoration of Temperate Wetlands, pp. 113–120, John Wiley & Sons, Chichester.

Prevost, M., Belleau, P. and Plamondon, A.P. 1997. Substrate conditions in a treed peatland: Responses to drainage. Ecoscience 4: 543–544.

Price, J.S. 1992. Blanket bog in Newfoundland: 2. Hydrological processes. J. Hydrol. 135: 103–119.

Price, J.S. 1997. Soil moisture, water tension, and water table relationships in a managed cutover bog. J. Hydrol. 202: 21–32.

Price, J.S. 1996. Hydrology and microclimate of a partly restored cutover bog, Québec. Hydrol. Proc. 10: 1263–1272.

Price, J.S. 1998. Methods for restoration of a cutover peatland, Québec, Canada. *In:* Malterer, T.J., Johnson, K. and Stewart, J. (eds.), Proc. International Peat Society, International Peat Symposium, Peatland Restoration and Reclamation: Techniques and Regulatory Considerations, pp. 149–154, Duluth, Minnesota.

Price, J.S., Rochefort, L. and Campeau, S. 2002. On the use of shallow basins to restore cutover peatlands: Hydrology. Restoration Ecology 10: 259–266.

Price, J.S., Rochefort, L. and Quinty, F. 1998. Energy and moisture considerations on cutover peatlands: Surface microtopography, mulch cover and *Sphagnum* regeneration. Ecol. Engin. 10: 293–312.

Price, J.S. and Schlotzhauer, S.M. 1999. Importance of shrinkage and compression in determining water storage changes in peat: The case of a mined peatland. Hydrol. Proc. 13: 2591–2601.

Price, J.S. and Whitehead, G. 2002. The influence of past and present hydrological conditions on *Sphagnum* recolonization and succession in a bloc-cut bog, Québec. Hydrol. Proc., accepted (August 2002).

Price, J.S. and Whitehead, G. 2001. Developing hydrological thresholds for *Sphagnum* recolonization on an abandoned cutover bog. Wetlands 21: 32–42.

Quinty, F. and Rochefort, L. 1997. Peatland restoration guide – Guide de restauration des tourbières. Québec, Canadian Sphagnum Peat Moss Association, p. 21.

Quinty, F. and Rochefort, L. 1996. Plant Reintroduction on a Harvested Peat Bog. Northern Forested Wetlands: Ecology and Management, pp. 133–145, CRC/Lewis Publishers, Boca Raton, FL.

Ratcliffe, D. 1977. A Nature Conservation Review. Cambridge University Press, London.

Robinson, M. 1985. The hydrological effects of moorland gripping: a re-appraisal of the Moor House research. J. Envir. Manag. 21: 205–211.

Rochefort, L. and Bastien, D. 1998. Réintroduction de sphaignes dans une tourbière exploitée et abandonnée: évaluation de divers moyens de protection contre la dessiccation. Ecoscience 5: 117–127.

Rochefort, L., Quinty, F., Campeau, S. Johnson, K.W., Vitt, D.H. and Malterer, T.J. this issue. North American approach to the restoration of *Sphagnum* dominated peatlands. Wetl. Ecol. & Manag.

Romanov, V.V. 1968. Hydrophysics of Bogs. Israel Program for Scientific Translations, Jerusalem, p. 299.

Rothwell, R.L., Silins, U. and Hillman, G.R. 1996. The effects of drainage on substrate water content at several forested Alberta peatlands. Can. J. Forest Res. 26: 53–62.

Sacot, C. and Rochefort, L. 1996. Tolèrance des sphaignes à la dessiccation. Cryptogamie, Bryologie et Lichenologie 17: 171–183.

Schlotzhauer, S.M. and Price, J.S. 1999. Soil water flow dynamics in a managed cutover peat field, Québec: 1. Field and laboratory investigations. Water Res. Res. 35: 3675–3683.

Schothorst, C.J. 1977. Subsidence of low moor peat soil in the western Netherlands. Geoderma 17: 265–291.

Schothorst, C.J. 1982. Drainage and behaviour of peat soils. *In:* de Bakker, H. and van den Berg, M.W. (eds.), Proceedings off the Symposium on Peatlands Below Sea Level. ILRI Publication 30: 130–163. Institute of Land Reclamation and Improvement, Wageningen.

Schouwenaars, J.M. 1993. Hydrological differences between bogs and bog relicts and consequences for bog restoration. Hydrobiologia 265: 217–224.

Schouwenaars, J.M. 1990. A study on the evapotransiration of *Molini caerulea* and *Sphagnum papilosum*, using all weighable lysimeters. *In:* Schouwenaars, J.M. (ed.), Problem Oriented Research on Plant-Soil-Water Relations. PhD. Thesis, Agricultural University, Wageningen, The Netherlands.

Schouwenaars, J.M. 1988. The impact of water management on groundwater fluctuations in a disturbed bog relict. Agric. Water Manag. 14: 439–449.

Schouwenaars, J.M. 1988a. Hydrological research in disturbed bogs and its role in decisions on water management in the Netherlands. *In:* Proceedings of the International Symposium on the Hydrology of Wetlands in Temperate and Cold Regions. Publications of the Academy of Finland 4/1988, Vol. 1: 170–177.

Schouwenaars, J.M. and Vink, J.P.M. 1992. Hydrophysical properties of peat relicts in a former bog and perspectives (sic) for *Sphagnum* regrowth. Int. Peat J. 5: 15–28.

Siegel, D.I., Reeve, A.S. Glaser, P.H. and Romanowicz, E.A. 1995. Climate-driven flushing of pore water in peatlands. Nature 374: 531–533.

Smart, J. 1983. The Plant Ecology of Revegetated Peat Cuttings in Ombrotrophic Mires with Special Reference to Thorne Moors, South Yorkshire. PhD thesis, University of Sheffield, Sheffield, England.

Spieksma, J.F.M., Moors, E.J., Dolman, A.J. and Schouwenaars, J.M. 1997. Modelling evaporation demand from a drained and rewetted peatland. J. Hydrol. 199: 252–271.

Spieksma, J.F.M. 1999. Changes in the discharge pattern of a cutover raised bog during rewetting. Hydrol. Proc. 13: 1233–1246.

Tallis, J.H. and Meade, R. 1997. Blanket mire degradation and management, *In:* Tallis, J.H., Meade, R. and Hulme, P.D. (eds.), Blanket Mire Degradation: Causes, Consequences and Challenges, pp. 212–216, Macaulay Land Use Research Institute, Aberdeen.

Van Seters, T.E. and Price, J.S. 2001. The impact of peat harvesting and natural regeneration on the water balance of an abandoned cutover bog, Québec. Hydrol. Proc. 15: 233–248.

Van Seters, T.E. and Price, J.S. 2002. Towards a conceptual model of hydrological change on an abandoned cutover bog, Qúébec. Hydrol. Proc. 16: 1965–1981.

Van Wirdum, G. 1991. Vegetation and Hydrology of Floating Rich-Fens, doctoral thesis, University of Amsterdam, Amsterdam.

Vasander, H., Tuittila, E.-S., Lode, E., Lundin, L., Ilomets, M., Sallantaus, T., Heikkilä, R., Pitkänen, M.-L. and Laine, J. this issue. Restoration of peatlands in Northern Europe. Wetl. Ecol. & Manag.

Verry, E.S. 1988. The hydrology of wetland and man's influence on it. *In:* Proceedings of the Symposium on the Hydrology of Wetlands in Temperate and Cold Regions, pp. 41–61, Academy of Finland, Joensuu.

Waddington, J.M. and Warner, K.D. 2001. Atmospheric CO, sequestration in restored mined peatlands. Ecoscience 8: 359–368.

Waddington, J.M. and Price, J.S. 2001. Effect of peatland drainage, harvesting, and restoration on atmospheric water and carbon exchange. Phys. Geogr. 21: 433–451.

Wheeler, B.D. 1988. Chemical conditions and revegetation of cutover raised mires. *In:* Fojt, W. and Meade, R. (eds.), Cutover lowland raised mires. pp. 48–58. Nature Conservancy Council publication No. 24, Peterborough.

Wheeler, B.D. 1999. Water and plants in freshwater wetlands. *In:* Baird, A.J. and Wilby, R.L. (eds.), Eco-Hydrology: Plants and Water in Terrestrial and Aquatic Environments, pp. 127–180, Routledge, London.

Wheeler, B.D. and Shaw, S.C. 1995. Restoration of Damaged Peatlands. Her Majesty's Stationery Office, London, p. 211.

Whinam J., Hope, G.S., Clarkson, B.R., Buxton, R.P., Alspach, P.A. and Adam, P. this issue. *Sphagnum* in peatlands of Australia: Their distribution, utilization and management. Wetl. Ecol. & Manag.