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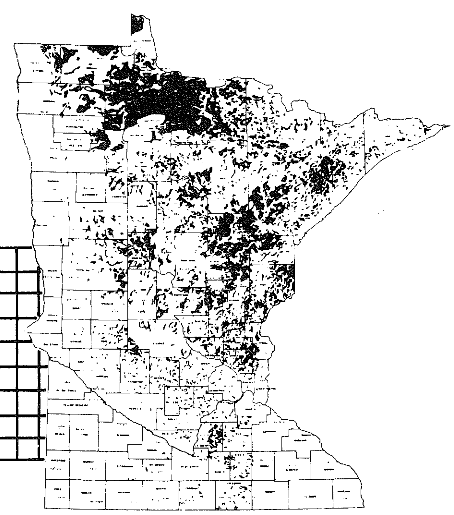
MINNESOTA PEAT PROGRAM

The Water Resources of Peatlands: A Literature Review

1980



Minnesota Department of Natural Resources

THE WATER RESOURCES OF PEATLANDS

A LITERATURE REVIEW

by

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Prepared for

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Minnesota Peat Program

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WATER QUANTITY

Hydrologic Characteristics of Peat

- Bulk density generally increases with depth and degree of decomposition but decreases with higher fiber content.
- Less decomposed peats have higher water contents at saturation than more decomposed peat.
- Greater bulk densities result in lower porosity, water retention, water yield coefficients, and hydraulic conductivities.

Hydrologic Cycle of Peatlands

- Evapotranspiration differs for various wetland plants with alder and willow being higher than sedge which are greater than tamarack, while all are greater than open water evaporation.
- Annual evapotranspiration from bogs is typically less than from fens but both exhibit higher evapotranspiration than mineral watersheds.
- Evapotranspiration is reduced when the water table is lowered.
- The groundwater moves rapidly in surface layers of peatlands and slowly in deeper layers. Groundwater movement in peatlands is poorly understood.
- Water table levels exhibit seasonal patterns in fluctuation being highest in the spring and receding in the summer with periodic rises due to rainstorms.
- Peatlands do delay stormwater runoff but do not capture and hold precipitation and slowly release water over time. Most runoff from peatlands in northern climates occurs in spring as snowmelt. Runoff from fens is more evenly distributed throughout the year than from bogs.
- The source of water to bogs is precipitation while fens receive water by both precipitation and groundwater.

Effects of Drainage on Water Quantity

- Drainage increases peat subsidence producing increases in bulk density and decreases in porosity, permeability, water content, and total water holding capacity. Subsidence is probably greater on bogs and on thin peat deposits than on fens and thick deposits.

- Clearing of peatlands decreases water yield coefficients at the surface of the peat.

- Drainage generally reduces evapotranspiration especially during the summer and in dry years. The amount of ET reduction is greater immediately after drainage and on densely drained areas.

- Drainage lowers the groundwater table with the influence of drainage extending further from the ditch in bogs than in fens. Drainage also modifies the pattern of groundwater flow so that movement is toward the ditch from both surface and deep layers.

- Drainage increases the storage capacity of a peatland. To achieve desired drainage norms, ditches must be spaced closer in fens than in bogs.

- Drainage increases interception on forested peatlands due to increased vegetative growth while clearing eliminates interception.

- The effect of drainage on runoff varies with the spacing, age, and type of drainage; the peat or peatland type; the vegetation type; and climate. Peak and minimum flows increase as ditch spacing decrease but flows decrease with increasing age of drainage. Ditches yield greater peak and average runoff and lower minimum flows than do drains. Annual and summer runoff from a drained bog is greater than from a drained fen while drained woodless bogs have higher annual runoff than drained forested bogs. Differences between the runoff from drained vs. undrained areas are less in wet years than in dry years.

- Generally, drainage has been found to increase annual, minimum, and seasonal runoff while decreasing peak runoff and distributing the flow more evenly throughout the year.

- Drainage appears to delay runoff response by increasing the storage in peat.

- Research on the effects of peat mining on runoff and other components of the hydrologic cycle does not exist.

Effects of Timber Harvesting on Water Quantity

- East-west strip cuts on black spruce peatlands increase snow accumulation and delay snowmelt.
- Timber harvesting reduces interception resulting in a rise in net precipitation.
- Timber harvesting reduces evapotranspiration with the reduction being more pronounced initially.
- Timber harvesting is not likely to change groundwater levels in fens but could increase water levels in bogs in wet years and decrease levels in dry periods.
- Generally, timber harvesting increases runoff with the effect tapering off over time.

WATER QUALITY

Water Quality of Natural Peatlands

- Bog waters are more acid and ion-poor than fens because fens receive more dissolved nutrients from the groundwater. Also, bog waters are usually of a darker color and contain more humic and fulvic acids than fens.
- Inherent seasonal variations in the concentrations of many ions in peatland waters occur with high flows producing higher values of pH, dissolved oxygen, and potassium and lower concentrations of conductivity, acidity, calcium, manganese, iron, aluminum, and sulfate.

Chemical Content of Peat

- Fens generally contain higher amounts of plant nutrients than bogs such as: nitrogen, phosphorus, potassium, calcium, magnesium, and sodium.
- Numerous heavy metals have been found in peatlands throughout the world including: iron, manganese, zinc, copper, lead, chromium, nickel, mercury, and uranium.
- The herbicide DDT and the breakdown form DDD have been found in Canadian peatlands.

- The availability and solubility of plant nutrients vary with pH and oxygen. Fens are expected to have greater availability of: nitrogen, phosphorus, potassium, sulfur, calcium, magasium, iron, boron, copper, and molybdenam while bogs could have greater availability of manganese and zinc.

Significance of Water Quality Parameters

- Based upon the literature, discharge waters from natural peatlands have the potential of exceeding the toxic limits for fish for the constituents: pH, zinc, and dissolved oxygen; exceeding drinking water standards for iron and color; and exceeding the MPCA fisheries and recreation standards for pH, color, and dissolved oxygen.

- The concentrations of heavy metals in peatland runoff waters are essentially unknown.

Effects of Agriculture on Water Quality

- Agricultural activities which include: drainage, cultivation, and fertilization appear to increase nitrogen, phosphorus, and potassium concentrations in discharge and soil solution waters.

- Cultivation and fertilization appear to increase the specific conductivity of soil solution and discharge waters.

- Agricultural practices modify the relative amounts of various forms of nitrogen in discharge waters.

Effects of Forestry on Water Quality

- Forest drainage is likely to increase pH, calcium, sodium, potassium, manganese, and possibly conductivity; and decrease organic carbon and acidity and possibly iron, ammonium, and phosphate with the magnitude of change being greater if the ditch intercepts mineral soil.

- Forest drainage is likely to increase suspended sediment immediately after drainage followed by a reduction over time with periodic rises during rain storms.

- Fertilization with NPK of drained forested peatlands increases phosphorus and decreases iron in peatland waters. Fertilization with NPK and lime ($CaCO_3$) increases pH and calcium and reduces iron in drainage waters, and reduces ammonium and nitrate in the peat.

Effects of Peat Mining on Water Quality

- Drainage and mining of bogs may increase pH, calcium, magnesium, bicarbonate, sulfate, and humic and fulvic acids and decrease chloride and nitrate concentrations in discharge waters.

- Drainage and mining of fens may increase pH, magnesium, sulfate, nitrate, and humic and fulvic acids and decrease calcium and bicarbonate concentrations in discharge waters.

- With increased age of drainage and mining, pH, iron, calcium, magnesium, bicarbonate, and humic and fulvic acid concentrations increased due to increased aeration.

- Studies of the effects of peat mining on water quality are rare.

INTRODUCTION

Increased pressures for the use of Minnesota's peatlands raises the question of the potential effects of expanded use on Minnesota's water resources. Preliminary investigations determined that peat mining could affect water quality and water quantity, but that baseline and post harvesting field monitoring were needed to reach definite conclusions (Brooks and Predmore 1978). During 1977, the project "Water Resources of Peatlands" was begun to provide field documentation and fill in the gaps in information. This report is a comprehensive review of the literature on the water resources of peatlands.

I. WATER QUANTITY

A. HYDROLOGIC CHARACTERISTICS OF PEAT

An understanding of the physical and hydrologic characteristics of peat is necessary to evaluate the movement of water through and from peatlands. The physical properties of peat including; bulk density, porosity, void ratio, and fiber content, regulate water movement but are not actually hydrologic expressions. The hydrologic characteristics of peat include: water content, hydraulic conductivity, water yield coefficient, water retention, and capillarity.

These characteristics are important in defining the three major peat types used in soil classification in the U.S. These peats are distinguished according to the degree of decomposition using fiber content (USDA 1968). Fibric is derived from Latin meaning fibre-containing and sapric is taken from Greek meaning rotted (Farnham 1968). Descriptions of these three peats are:

Fibric

These least decomposed of all organic soil materials contain very high amounts of fiber that is well preserved and readily identifiable as to botanical origin. They have low bulk densities and high water contents when saturated. They often occur in the very acid, raised bogs of the Boreal forests and consist largely of sphagnum moss peat. These materials have fiber contents (unrubbed) exceeding 2/3 of the volume with colors of light yellowish brown, dark brown, or reddish brown.

Hemic

These are intermediate in degree of decomposition between fibric and sapric materials and are partly altered both physically and biochemically. They have intermediate values for fiber content, bulk density and water content. Fiber contents average between 1/3 and 2/3 of the volume before rubbing and are colored dark grayish brown to dark reddish brown.

Sapric

These are the most decomposed of the organic soil materials and have the least amount of fiber, highest bulk density, and least water content at saturation. They commonly occur on the surface of peats that have been drained and cultivated or on well-aerated areas or areas rich in calcium in undrained peatlands. Fiber contents average less than 1/3 of the volume (unrubbed) and they are commonly very dark grey to black in color.

1. Bulk Density

Bulk density (Db) is defined as the mass of dry material per unit volume and is commonly expressed as g/cm^3 (Black et al. 1965). The method of determining bulk density is by obtaining a known volume of sample (in situ) using a corer and weighing the sample after drying to 105° C (Black et al. 1965). Boelter and Blake (1964) recommend taking a saturated bulk density sample for peat soils. Bulk density is determined from the formula (Buckman and Brady 1960):

$$Db = \frac{\text{oven-dry weight}}{\text{volume}}$$

Generally bulk density is lowest at the surface of a peatland and increases with depth which is related to the peat types in the soil profile. Bulk densities for peat soils are much lower than those for mineral soils (Table 1).

Table 1. Bulk densities of major peat types and mineral soils.

Peat type	Db[a] (g/cm^3)	Mineral Soil	Db[b] (g/cm^3)
Fibric	< .075	Sand	1.6
Hemic	.075 - .195	Silt	1.4
Sapric	> .195	Clay	1.2

[a] From Boelter 1968, 1969.

[b] From Buckman and Brady 1960.

Boelter (1964(b), 1966, 1968, 1969, 1975) has investigated the bulk densities of Minnesota peat soils and developed most of the interelationships for bulk density and other hydrologic characteristics (Table 2). It is evident that bulk density is related to fiber content, porosity, hydraulic conductivity, water yield coefficient, and water retention.

The relationship between bulk density or fiber content and the water content of peat is not consistent at various suctions (Figure 1). At low bulk densities, more water is contained in peat when saturated than for more dense peats. However, that water is held loosely and is easily given up to drainage (increased suction). More dense peats hold their water more tightly. Given bulk density (X), the

Table 2. Hydrologic characteristics of several Minnesota peats (Boelter 1975).

Peat type	Sampling depth	Bulk density	Total porosity	Water content 0.1 bar	Water yield coefficient	Hydraulic conductivity	Fibre content 0.1 mm
	cm	g/cc	%	% vol.	cc/cc	10^{-5} cm/sec	%
Sphagnum moss peat							
Live, undecomposed mosses	0-10	0.010	99.6	14.0	0.86	*	94
Undecomposed mosses	15-25	0.040	89.4	29.8	0.60	3810.00	87
Undecomposed mosses	45-55	0.052	91.6	43.9	0.48	104.00	76
Moderately decomposed with woody inclusions	35-45	0.153	86.5	63.6	0.23	13.90	52
Woody Peat							
Moderately decomposed	35-45	0.137	87.2	60.3	0.27	496.00	59
Moderately well decomposed	60-70	0.172	89.4	70.2	0.19	55.80	32
Herbaceous peat							
Slightly decomposed	25-35	0.069	86.8	30.2	0.57	1280.00	75
Moderately decomposed	70-80	0.156	87.0	73.9	0.13	0.70	40
Decomposed peat							
Well decomposed	50-60	0.261	82.4	74.8	0.08	0.45	25

* Too rapid to measure with techniques used.

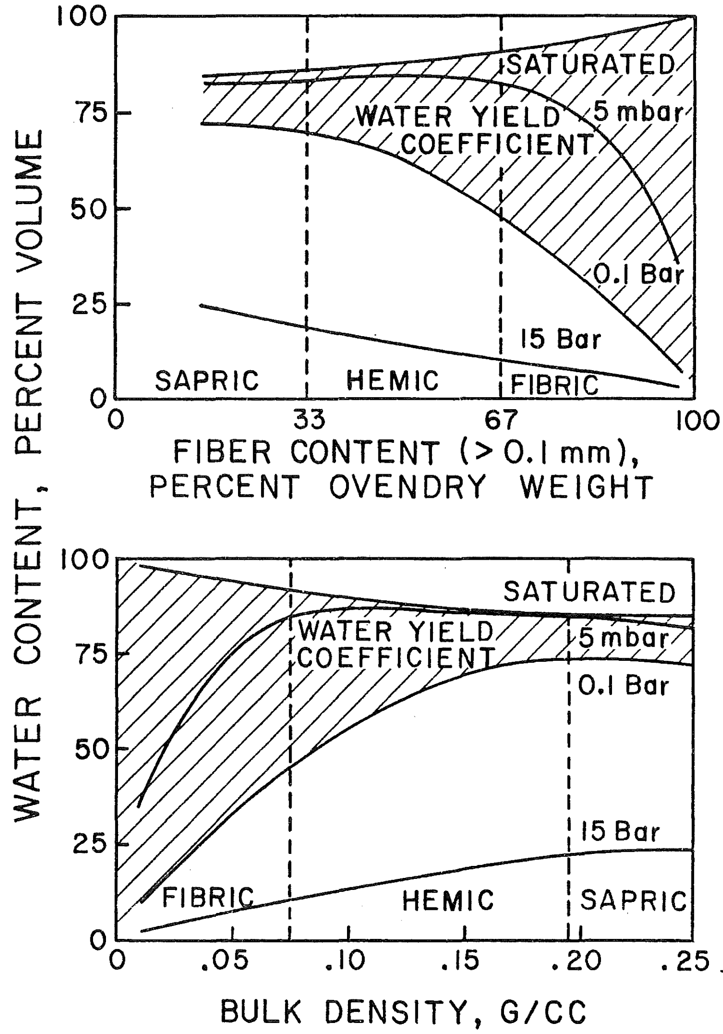


FIGURE 1 : RELATIONSHIPS AMONG FIBER CONTENT, BULK DENSITY, AND WATER CONTENT FOR DIFFERENT TYPES OF PEAT AT VARIOUS SUCTIONS (FROM BOELTER, 1969).

water content at saturation (Y_1) can be determined as described by Boelter (1969):

$$Y_1 = 99 - 123.45X + 252.92X^2 \qquad (R^2 = .66)$$

The water content at 0.1 bar suction (Y_2), the approximate field capacity, can be similarly determined by:

$$Y_2 = 2.06 + 719.35X - 1809.68X^2 \qquad (R^2 = .88)$$

By knowing the bulk density, the fiber content (Y_3) can be estimated by:

$$Y_3 = 98.87 - 494.51X - 803.80X^2 \qquad (R^2 = .85)$$

The relationship between bulk density or fiber content and hydraulic conductivity is given in Figure 2. As the peat becomes more dense, the hydraulic conductivity decreases.

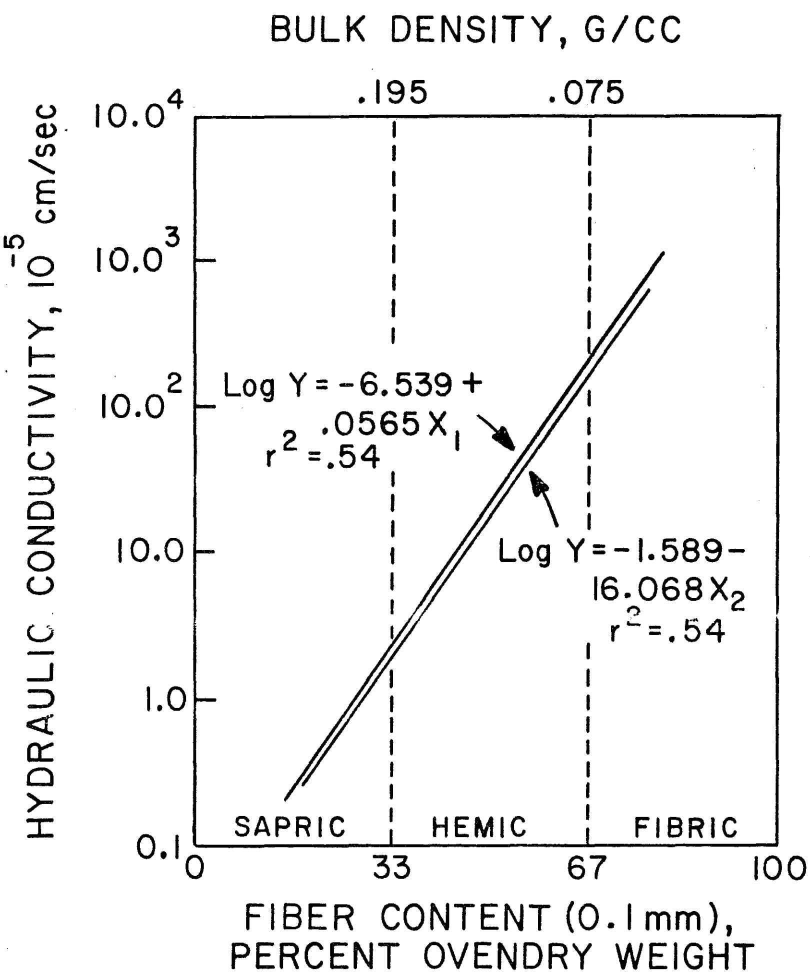


FIGURE 2 : RELATIONSHIP BETWEEN BULK DENSITY OR FIBER CONTENT AND HYDRAULIC CONDUCTIVITY FOR DIFFERENT PEAT TYPES (FROM BOELTER, 1969)

In summary, bulk density increases with depth and degree of decomposition but decreases with higher fiber content. Greater bulk densities result in lower porosity, water retention, water yield coefficients, and hydraulic conductivities.

2. Fiber Content

The U.S. peat classification system uses fiber content to define the three major types of peat defining fibers as particles greater than 0.1 mm in size. The more decomposed peats have fewer fibers (Farnham 1968). Fiber content varies from 25 to 94 percent for Minnesota peats (Table 2). Since fiber content is inversely proportional to bulk density, high fiber content peats hold more water at saturation but less water when unsaturated than low fiber content peats. This relationship is the mirror image of the bulk density - water content relationship (Figure 1). This difference occurs because decomposed peats with low fiber content have smaller pores which hold water more strongly than less decomposed peats (Boelter 1968). It is evident in Table 2 that water storage, water yield coefficients, and hydraulic conductivities are also related to fiber content.

3. Porosity and Void Ratio

The total porosity (P) or pore space is that portion of the soil occupied by air and water (Black et al. 1965). Porosity is determined from the bulk density (Db) and particle density by:

$$P = \frac{(1 - Db)\ 100}{Pd}$$

where particle density (Pd) = weight of soils/volume of solids (Buckman and Brady 1960). As shown in Table 2, the porosities of peat vary from about 82 to nearly 100 percent. Generally, porosity is greatest at the peatland surface and decreases with depth. Porosity is directly related to the peat type and peat soils have much higher total porosities than do mineral soils (Table 3). Like fiber content, porosity various inversely with bulk density.

Table 3. Total porosities for major peat types and mineral soils.

Peat type	Total Porosity[a] percent	Material	Total Porosity[b] percent
Fibric	>90	Gravel	30-40
Hemic	85-90	Sand	35-40
Sapric	<85	Till	40-50

[a] From Boelter 1968

[b] From Boelter 1966

Void ratio is a term related to porosity that is sometimes found in peat literature. The void ratio is the ratio of the volume of spaces to the volume of solids and can be determined from porosity (P) as described by Irwin (1968):

$$\text{Void Ratio} = \frac{P}{(100-P)}$$

or as indicated by Dasberg and Neuman (1977) for a fully saturated peat by:

$$\text{Void Ratio} = \frac{\text{Wet weight} \times P}{100}$$

Void ratios have been found to be less at the surface of peats (about 2:1) than in deeper peat (10 to 17:1) (Dasberg and Newman 1977).

4. Water Content

The amount of water in peat can be expressed as: a) oven-dry weight, b) wet weight, or c) volumetric water content. Traditionally, soil water content has been expressed on a dry weight basis (Buckman and Brady 1960). According to Black et al. 1965, water content based on a dry weight basis is obtained by:

$$\% \, WC_{OD} = \frac{(\text{Wet Wt.} - \text{Dry wt.})\ 100}{\text{Dry wt.}}$$

For peat soils, water contents expressed on an oven-dry basis may be as high as 2000 percent (Boelter 1968). Water content based on the wet weight of soil is obtained by:

$$\% \, WC_{WW} = \frac{(\text{Wet wt.} - \text{Dry wt.})\ 100}{\text{Wet wt.}}$$

Water contents based on the wet weight of soil are always less than 100 percent and for peat soils vary from 62 percent for well decomposed peat to 95 percent for undecomposed sphagnum moss (Boelter 1968). Water content expressed on a volumetric basis is obtained by multiplying the oven-dry water content by the bulk density or by the formula:

$$\% \, WC_{V} = \frac{\text{Db } (\text{Wet wt.} - \text{Dry wt.})\ 100}{\text{Dry wt.}}$$

Volumetric water contents for peat vary from 60 to 97 percent and are the preferred expression, according to Boelter and Blake (1964). Gravimetric methods are recommended for accurate water content determinations of peat soils because resistance blocks are unsatisfactory at low suctions and hydrogen ions in peat interfere with neutron scatter methods (Boelter 1975). Volumetric water contents at saturation decrease with increasing decomposition and bulk density (Figure 1).

5. Hydraulic Conductivity

Hydraulic conductivity (K) is the apparent velocity of water flowing through a saturated soil in response to a hydraulic gradient (Irwin 1968). Hydraulic gradient is the change in head or water surface elevation over some distance. Hydraulic conductivity is important in controlling the rate at which water at some depth is supplied to the soil surface for evaporation. Also, hydraulic conductivity is one factor controlling the rate of groundwater movement after drainage (Rycroft et al 1975a).

Hydraulic conductivity is most widely used in application to Darcy's Law for flow through a porous media:

$$Q = K\ A\ dh/dl$$

where: Q = flow rate ($cm^3\ sec^{-1}$), K = hydraulic conductivity ($cm\ sec^{-1}$), A = cross-section area (cm^2), and dh/dl = hydraulic gradient of head (h cm) over distance (l cm). With a hydraulic gradient of 1.0, i.e., a water table having a slope of 45^{o}, the hydraulic conductivity (K) equals the rate of water movement. Such steep water table slopes are rarely achieved in peatlands except immediately adjacent to drainage ditches.

There are two field methods and one laboratory method used for estimating hydraulic conductivity in peatlands: a) auger-hole method, b) seepage-tube method, and c) lab cores (Rycroft et al. 1975a).

a. Auger-hole Method.

This method, considered to be the easier of the two field methods, provides hydraulic conductivity in an unlined well by charting the water level recovery after water is removed. Hydraulic conductivity is determined from the equation:

$$K = \frac{2.3r\,A_n}{(2c+r)\ (t_j - t_i)}\ \log_{10} \frac{h_i}{h_j}$$

where K = hydraulic conductivity (cm sec^{-1}), r = well radius (cm), A_n = shape factor = rc/0.19, c = well depth below water table (cm), h_i = initial water level (cm) at time t_i in sec, and h_j = water level at time t_j.

b. Seepage-tube (piezometer) method.

This method differs from the first in that a lined well with a cavity below the end of the well is used (Rycroft et al. 1975a). Hydraulic conductivity is estimated from Kirkham's Equation (1946):

$$K = \frac{\pi r^2}{A}\ \frac{\ln h_i/h_j}{(t_j - t_i)}$$

where K = hydraulic conductivity (cm sec^{-1}), r = radius of cavity (cm), h_i = initial water level in cm at time t_i in sec, and h_j = water level at time t_j. This method measures horizontal hydraulic conductivity and is especially useful in layered peat soils where the conductivity may change through the profile.

c. Lab core method.

Hydraulic conductivity determined in the laboratory usually involves measuring water movement through a soil core and applying Darcy's Law (Boelter 1965):

$$K = \frac{Q\Delta L}{At\Delta H}$$

where K = hydraulic conductivity (cm/sec), Q = volume of water (cm^3) passing through core in time t (sec), A = cross-section area of core

(cm^2), ΔL = core length (cm), and ΔH = core length plus head of water (cm). Either horizontal or vertical K may be determined by this method depending upon the orientation of the collected soil core.

d. Comparison of Methods.

Boelter (1965) found that laboratory determinations of hydraulic conductivity were significantly higher than field values using the seepage-tube method, attributing the difference to leakage along core walls. Both Boelter (1965) and Rycroft _et al_. (1975b) recommend field rather than lab determinations and Rycroft prefers the seepage-tube method while Baden (1976) and Irwin (1968) prefer the auger-hole method which gave them more reproduceable results. Boelter (1965) and Rycroft _et al_. (1975a) found vertical K to be approximately equal to horizontal K, yet Irwin (1968) found a significant difference between hydraulic conductivities for the two directions. Rycroft _et al._ (1975b) has also studied the variability of hydraulic conductivity determinations with time and initial head. Conductivity varied linearly with initial head and non-linearly with time and suggested that peat particle migration during the test was responsible for this "non-Darcian" behavior.

Boelter (1968) has found hydraulic conductivities for Minnesota peats to vary from 3810 to 0.47 x 10^{-5} cm/sec (Table 2). Hydraulic conductivity decreases with increasing decomposition (Table 4). According to Boelter (1965, 1975), the hydraulic conductivity for hemic peats is comparable to fine sands or sandy loams while

Table 4. Hydraulic conductivities for major peat types (from Boelter 1968).

Peat type	Hydraulic Conductivity 10^{-5} cm/sec
Fibric	>180
Hemic	2.1 - 180
Sapric	<2.1

conductivities for sapric peats are lower than for clay or glacial till. Elsewhere, Sturges (1968) found very low hydraulic conductivities for a Wyoming mountain bog ranging from 0.19 to 0.28 x 10^{-5} cm/sec. Irwin (1968) determined the average hydraulic conductivity for Holland marsh, Ontario peats to be 57 x 10^{-5} cm/sec which is comparable to Boelter's hemic peat.

Hydraulic conductivity is inversely related to bulk density as shown in Figure 2 (Boelter 1964b). This relationship is very useful since bulk density is easier to obtain than hydraulic conductivity. Given bulk density (X), hydraulic conductivity (Y) may be determined from (Boelter 1969):

$$\text{Log } Y = 0.0565\, X - 6.539$$

In general, as the degree of decomposition increases, hydraulic conductivity decreases (Boelter 1975; Rycroft et al 1975a). Dasberg and Newman (1977) have related hydraulic conductivity to the consolidation of the peat due to management practices by:

$$K = C_V\, A_V\, \delta_W/(1+e)$$

where K = hydraulic conductivity, C_V = coefficient of consolidation, A_V = consolidation index, δ_W = specific weight of water, and e =

void ratio. Hydraulic conductivity decreases when the peat is compressed. Romanov (1961) reports a useful relationship for the variation of hydraulic conductivity in a peat profile by:

$$K = A/(Z+1)^{M}$$

where K = hydraulic conductivity, Z = depth, and A and M are coefficients for a specific peatland.

6. Water Yield Coefficient.

The water yield coefficient is a measure of the amount of water that is removed (added) from a peat profile when the water table is lowered (raised) (Boelter 1968). The term water yield coefficient is synonymous with the terms specific yield, storage coefficient, coefficient of drainage, and coefficient of groundwater level found elsewhere in the literature (Boelter 1966; Dooge 1975; Heikurainen 1964). The most common use of this expression is in water balance studies where the volumetric change in storage can be obtained from a corresponding change in water level.

Several methods have been used to determine water yield coefficients. Both Heikurainen (1963) and Bay (1967c) used correlations of precipitation input with corresponding rises in water table levels to determine water yield coefficients. Vorob'ev (1963) used the indirect means of deriving coefficients by measuring the pore volume distribution of peat soils. Boelter (1968) determined coefficients from water retention curves in the laboratory. For this method, the water yield coefficent would be the difference between the percent water contents at saturation and at 0.1 bar suction, the point where all gravitational water is removed. This difference also represents the maximum amount of water that would be removed by drainage (Boelter 1975). These water yield coefficients are shown in Figure 1 as the hatched area.

According to Boelter (1975), water yield coefficients for Minnesota peat soils vary from 0.08 to 0.86 with less decomposed peats having higher coefficients (Table 2). Table 5 shows water yield coefficients for the three major peat types. To illustrate the use of water yield coefficients, assume that a coefficient of 0.50 was determined for a fibric peat from water retention curves or from a correlation between a number of observed precipitation events and corresponding water table responses. A 30 cm fall in the water table over a month would yield a change in storage of 0.50 x 30 cm = 15 cm. A 30 cm drop in a hemic peat with a coefficient of 0.25 would yield 7.5 cm of water and a sapric peat with a coefficient of 0.10 would yield 3 cm of water.

Table 5. Water yield coefficients for major peat types (Boelter 1968).

Peat Type	Water Yield Coefficient (%)
Fibric	>42
Hemic	15-42
Sapric	<15

Boelter (1968) reported that water yield coefficients were highly correlated (R^2 = 0.89) to bulk density. This relationship could be very useful in determining water yield coefficients and predicting the effect of drainage on change in storage.

7. Water Retention

Soil water retention refers to the amount of water (percent) retained by a soil under various suctions (Buckman and Brady 1960). Water retention curves for various Minnesota peats show that undecomposed sphagnum peat retains more water at saturation than does the more decomposed peats but contains less water at suctions greater than 0.005 bars (5 MBARS) (Figure 3). Field capacity is the water content at 0.1 bar and the wilting point is the water content at 15 bar (Boelter 1968). The difference between water contents at 0.1 bar and 15 bar is termed "available water" and the difference between water contents at saturation and 0.1 bar is the water yield coefficient which is also referred to as "gravitational water" (Buckman and Brady 1960).

Dooge (1975) indicated that a fall in the water table from the surface to a depth of 5 cm (about 2 inches) produces a suction of 5 MBars which would reduce the water content of an undecomposed peat from 97 to 64 percent while only decreasing the water content of a decomposed peat from 83 to 82 percent. To create 0.1 bar suction, the water table would have to be lowered 100 cm (3.3 feet) or to produce the wilting point at 15 bar, the water table would have to be lowered 15,000 cm (492 ft.).

The relationship between bulk density or fiber content and water retention is also shown in Figure 1. As bulk density increases (fiber content decreases), the water is held more tightly.

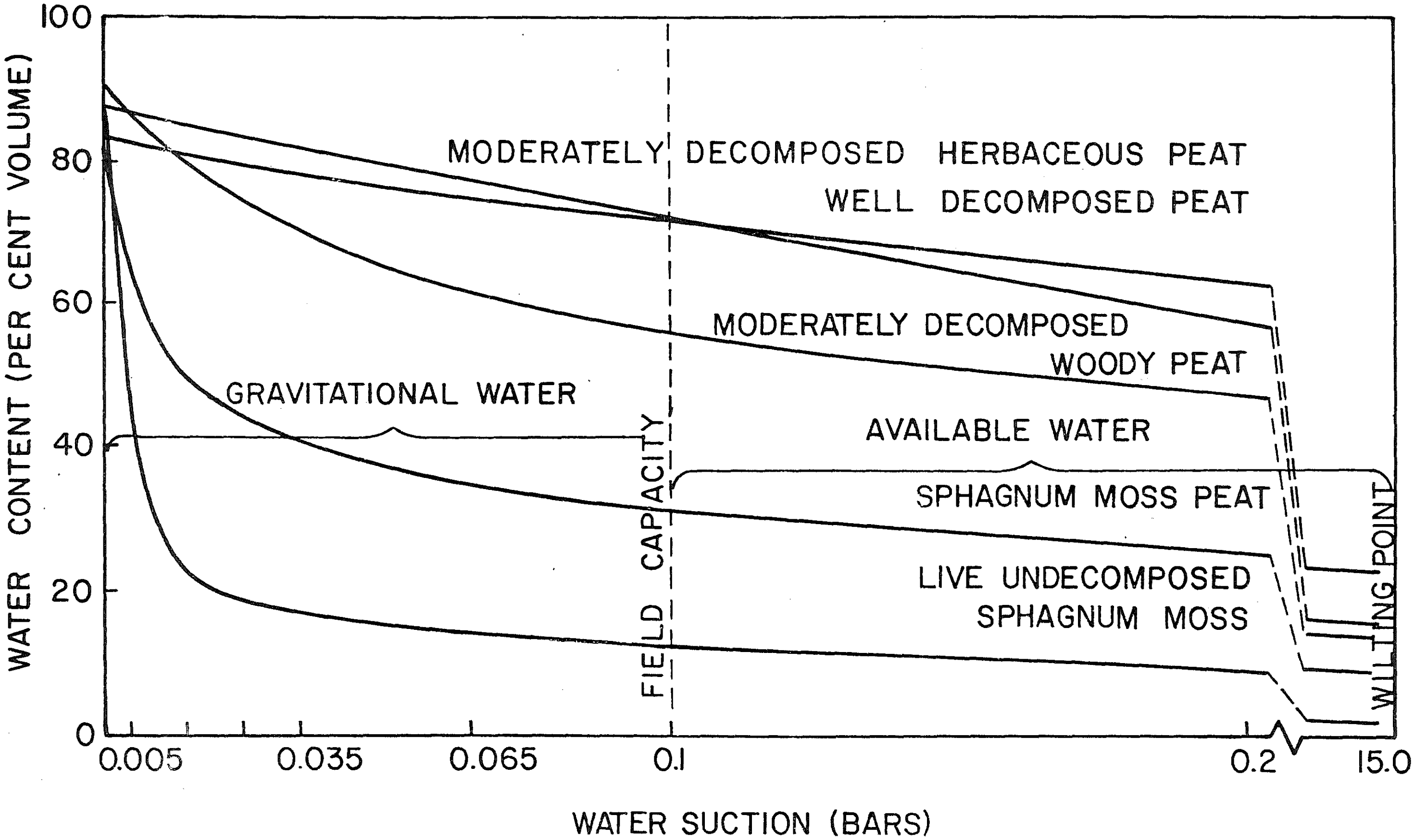


FIGURE 3: WATER RETENTION CURVES FOR SEVERAL MINNESOTA PEATS.
(FROM BOELTER 1968)

8. Capillarity

The height to which water will rise in a soil is termed the "capillary rise" (Buckman and Brady 1960). Romanov (1961) reports that the maximum capillary rise for peat is 20 cm while Kravchenko (1963) reports 60 cm for cultivated peat soils. Should the water table fall beneath these levels, evaporation could be reduced.

9. Summary

There are many physical and hydrologic characteristics of peat that control water movement within and from peatlands and most of these terms are interrelated. Table 6 summarizes these characteristics together with their general purpose. The interrelationships

Table 6. Summary of physical and hydraulic characteristics of peat.

Characteristic	Units	To Determine
Bulk density (Db)	g/cm^3	degree of decomposition, peat type, K, water yield coefficient, fiber content, water content at saturation.
Fiber content	%	peat type
Porosity (P)	%	amount of air and water in peat
Water content	%	water in peat
Hydraulic conductivity (K)	cm/sec	flow rate in peat
Water yield coefficient	%	change in storage
Water retention	%	wilting point, available water, water yield coefficient
Capillarity	CM	rise of water in peat

among characteristics are shown in Figure 4. This self-interaction matrix indicates that either bulk density or fiber content are the most important terms and that if either term was obtained, it would be possible to derive all other characteristics.

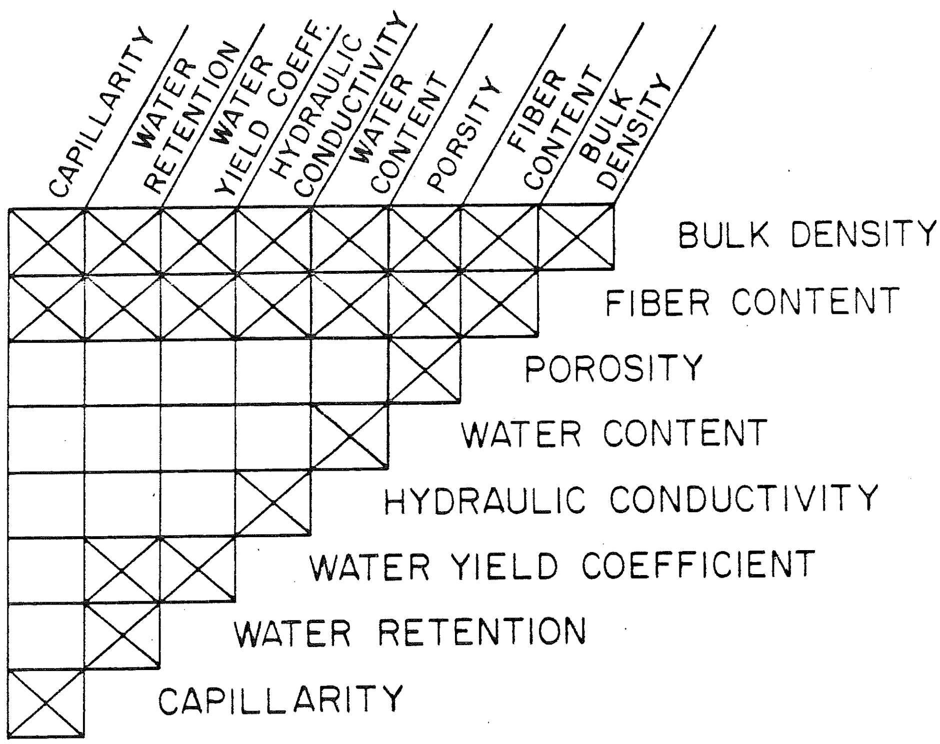


FIGURE 4: SELF-INTERACTION MATRIX OF THE PHYSICAL AND HYDROLOGIC CHARACTERISTICS OF PEAT.

B. HYDROLOGIC CYCLE OF PEATLANDS

1. Introduction

The hydrologic cycle of peatlands is similar to that cycle for any watershed system except that special cases have been identified. Two distinct hydrologic peatland systems based on the source of water, are: ombrotrophic and minerotrophic (Figure 5).

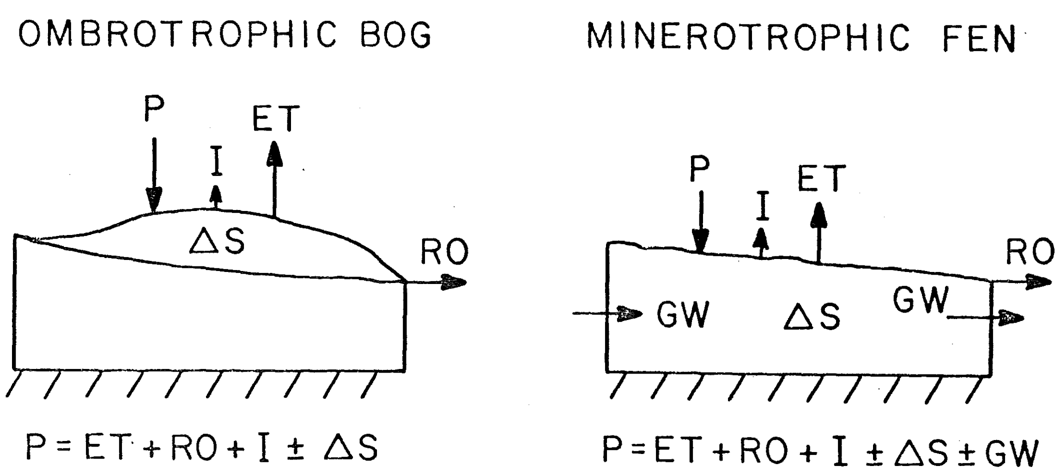


FIGURE 5: HYDROLOGIC CYCLE ON A OMBROTROPHIC BOG AND A MINEROTROPHIC FEN.

Ombrotrophic peatlands are so named because they are mineral-nourished by rain (Gorham 1967). Typically, the waters of these bogs are ion-poor and very acid. Ombrotrophic bogs are isolated from the regional groundwater system.

Minerotrophic peatlands derive their name by being mineral-nourished from ion-rich groundwater (Sjörs 1961; Gorham 1967; Boelter and Verry 1977; Romanov 1961; Moore and Bellamy 1974). The regional groundwater system is often a major component of the hydrologic cycle on these peatlands.

2. Precipitation

Precipitation is a major input into any peatland. For ombrotrophic bogs, atmospheric precipitation is the sole input since the bog lies above the regional groundwater system.

3. Evapotranspiration

Evapotranspiration is usually the greatest water loss from a peatland. Several methods of estimating evapotranspiration have been used in peatland studies including:

a. Bottomless lysimeters
b. Thornthwaite PET
c. Pan evaporation
d. Water balance
e. Change in groundwater table

a. Bottomless lysimeters

Bottomless lysimeters are usually metal cylinders placed vertically in peat to a depth of impermeable soil which serves as a seal. Evapotranspiration is determined by monitoring precipitation and water table changes and converting those changes to actual water loss (Kohnke et al. 1940; Van Bavel 1966; Spier 1962; Hudson 1965; and Harold 1966). Frequently, water yield coefficients are used to make these conversions. An undisturbed soil core is preferred and water levels should be maintained at the same depth both inside and outside the lysimeter (Bay 1966).

Bonde et al. (1961) used galvanized steel cattle tanks, 45 inches in diameter, two feet deep with vertically pleated walls to measure evapotranspiration from various wetland plants in Minnesota (Table 7). The evapotranspiration from all species, except tamarack, exceeded open water evaporation. The greatest evapotranspiration was from the brush which exceeded the shrubs.

Bay (1966) used bottomless lysimeters in Minnesota peatlands but these cylinders were constructed of 16 ga. sheet metal that were driven two feet into the peat. ET losses were determined from water

Table 7. Evapotranspiration as a percent of open water evaporation for seven wetland plant species in Minnesota (Bonde et al. 1961).

	ET as percent of Open Water Evaporation - - - - - Year - - - - -		
Species	1958	1959	1960
Alder	---	---	288
Willow	---	---	159
Cattail	122	---	---
Sedge	112	149	156
Formerly cattail now marshgrass & sedge	---	121	134
Tamarack	---	---	93
Open water	100	100	100
Precipitation	78	106	---

table changes by pumping tests. The ratio of the volume of water to water table change was found to range from 0.91 near the surface to 0.62 at lower depths. These ratios agree with water yield coefficients obtained by Boelter (1965). Lysimeter leakage was found to be immeasurable both by filling with water and by covering with plastic. Other similiar lysimeters have been constructed in peatlands by Chapman (1965).

Bay (1966), Dooge (1975), and Van Bavel (1961) have concluded that evapotranspiration measured from lysimeters is actual rather than potential evapotranspiration. Dooge (1975) further concludes that lysimeters provide the only direct measurement of ET from a vegetated soil surface which is important because ET decreases appreciably when the water table is lowered.

b. Thornthwaite PET

Potential evapotranspiration (PET) from peatlands can be obtained using the methods developed by Thornthwaite and Mather (1957). The procedure for determining monthly PET uses mean monthly temperature and station latitude in the following relationships:

$$PET = 1.6 \left(\frac{10t}{I}\right)^{a}$$

$$I = \Sigma i \text{ and } i = (t_m/5)^{1.514}$$

$$a = 6.75 \times 10^{-7}\,(I)^{3} - 7.71 \times 10^{-5}\,(I)^{2} + 1.792 \times 10^{-2}\,(I) + 0.49239$$

where PET = potential evapotranspiration (cm), t = mean monthly temperature (^{o}C), t_m = normal mean monthly temperature (^{o}C), and i = monthly heat index. Monthly values of PET are then adjusted for day length using correction factors based upon station latitude.

c. Pan Evaporation

Evaporation from standard U.S. Weather Bureau evaporation pans has been used to estimate evapotranspiration. The procedure involves daily readings of water levels in the pan with a hook gage (USDA 1979). Pan evaporation is then corrected with coefficients which vary geographically in the United States (Kohler et al. 1955). A pan coefficient of 0.7 is frequently quoted as an average adjustment to approximate evaporation from lakes and ponds. No coefficients that pertain to peatlands were found in the literature, although Bay (1966) found the following relationship:

Lysimeter ET = 0.16 + 0.90 * (Pan evaporation)

d. Water Balance

Water balance techniques for estimating evapotranspiration usually involve the measurement of all other components of the water balance equation and calculating ET as the residual. This residual method of obtaining ET frequently ignores groundwater inflow and deep seepage (Bay 1967c). Bay (1967c) used the water balance approach for Minnesota peatlands in the form:

$$ET = P - Ro - \Delta S$$

where ET = evapotranspiration, P = precipitation, Ro = runoff, and ΔS = change in storage. Chapman (1965) used a variation of the water balance equation for periods when runoff rates were equal at the beginning and the end of a period:

$$E = \frac{1}{t}\left(P - \frac{100\ Ro}{A}\right)$$

where E = evaporation (cm/day), t = days, P = precipitation (cm), Ro = runoff (M^3), and A = watershed area (M^2). For longer periods Chapman (1965) used the relationship:

$$E = P - Ro - \frac{W}{3}$$

where E = evaporation, P = precipitation, Ro = runoff, W/3 = change in water table divided by three since a 3 cm drop in the water table yields 1 cm of water. This second relationship by Chapman is actually identical to that used by Bay (1967c) and is the most common form.

e. Change in Groundwater Table

Heikurainen (1963) used the change in the groundwater table to estimate ET from peatlands by the equation:

$$G = CW$$

where G = fall (rise) of GW table; C = coefficient of groundwater table (water yield coefficient), and W = amount of water causing change. Heikurainen (1963) determined C from laboratory tests and developed an expression for C as a function of depth (Z):

$$C = 1.112 + 0.131Z + 0.010Z^2$$

Laboratory tests were used because the author felt precipitation could be better controlled. Later, Heikurainen and Laine (1968) recommended that lysimeters be used to determine 'C'. As an example of this method, assume that the coefficient of GW table was 3.1 and that the water table dropped 9mm in one day, the estimated evapotranspiration would be 9mm/3.1 = 2.9mm/day.

f. Other methods

There are several energy budget methods that could be used to estimate ET from peatlands based upon vapor transfers from the surface or by heat flow (Romanov 1961). These methods are quite difficult to instrument but give good results.

Bergland and Mace (1972) determined albedos for black spruce and sphagnum-sedge vegetative types in Minnesota. Albedo is the percent of shortwave radiation that is reflected (Rose 1966). The albedo for black spruce was found to be 6 to 8 percent while sphagnum-sedge was 11.6 to 16.1 percent, concluding that the absorbed energy is 5 to 10 percent greater in black spruce bogs than in sphagnum-sedge bogs.

g. Comparison of ET Methods.

Bay (1966) compared lysimeter ET to Thorthwaite PET, Pan evaporation, and Hamon methods and determined the following relationships:

Lysimeter ET = 0.08 + 1.03 (Thornthwaite PET)

Lysimeter ET = 0.16 + 0.90 (Pan evaporation)

Lysimeter ET = 1.24 (Hamon ET)

Bay concluded that actual ET in high water table areas approached potential ET. Bay (1967c) also compared the water balance method to Thornthwaite PET. Over a six-year period, annual ET determined from a water balance varied from 88 to 116 percent of Thornthwaite PET. Bay (1967c) suggested that PET would exceed actual ET during dry years when the water table would be low and temperatures high. Monthly estimates of Thornthwaite PET varied greatly from water balance ET and were not recommended but Thornthwaite PET could be used to approximate a water balance for ungaged peatlands on a long-term seasonal basis.

The effect of the water table position on evapotranspiration has been noted by several investigators. Romanov (1961), emphasizing the importance of capillary flow on evapotranspiration, stated that transpiration would be reduced when the water table fell below the rooting level at about 30-35 cm and evaporation would be reduced when the water table fell below 45 cm. Heikurainen and Laine (1968) found that the change in groundwater level method compared favorably with pan evaporation except when the water table was deep. Boelter (1972a) found sphagnum evapotranspiration in controlled plots to be reduced by 50 percent when the water table was lowered 0.3 meters from the surface, concluding that capillarity was broken and the plants desiccated. Paivanen (1976) observed in Finland during dry summers that water balance ET was only 41 to 67 percent of regional PET while during rainy summers was 85 to 101 percent of PET, concluding that lower water tables in dry years reduced ET.

h. Differences in ET by plant species.

It has already been shown that ET by various wetland plant species range from 288 to 93 percent of open water evaporation (Table 7). Heikurainen (1963) determined that birch stands transpire as much as coniferous trees during the growing season on peatlands in Finland. Eggelsmann (1975b) found that ET from a peat grassland was greater than from a heather raised bog in the fall and winter but a sphagnum bog had even greater ET. In comparing peatland types, Dooge (1975) states that ET from bogs is less than that from fens by 5 to 15 percent. Baden and Eggelsmann (1968) found that ET from a raised bog in Germany was 5 percent higher than that from a mineral watershed.

4. Groundwater

The groundwater component of the hydrologic cycle is perhaps the most important yet the least understood. Groundwater has a major influence on peatland formation and groundwater chemistry affects the type of vegetation found on a peatland. There are several aspects of groundwater in peatlands that have hydrologic significance including: water table fluctuations, groundwater flow, and groundwater storage.

a. Water Table Fluctuations.

The water table is the upper boundary of groundwater where the water pressure is equal to atmospheric pressure (Rose 1966). Several investigators have monitored water table fluctuations in Minnesota and noted seasonal patterns. Manson and Miller (1953) studied 8 peatlands in Minnesota over a number of years using about 70 observation wells and 5 recording wells. The general trend was that the groundwater rose rapidly after spring breakup, remaining near the peat surface during spring, and then receded rapidly during the summer months. At the end of the summer, the water table leveled off and remained low during the winter. Bay (1967a, 1968, 1970) observed a similar trend in seasonal water table fluctuations (Figure 6). The interruptions in groundwater recession are caused by precipitation. Manson and Miller (1953) also noted that when precipitation was above or below normal, the water table was correspondingly high or low. The range of water table depths observed was from near the surface in spring to a depth of 1.2 to 1.8 meters

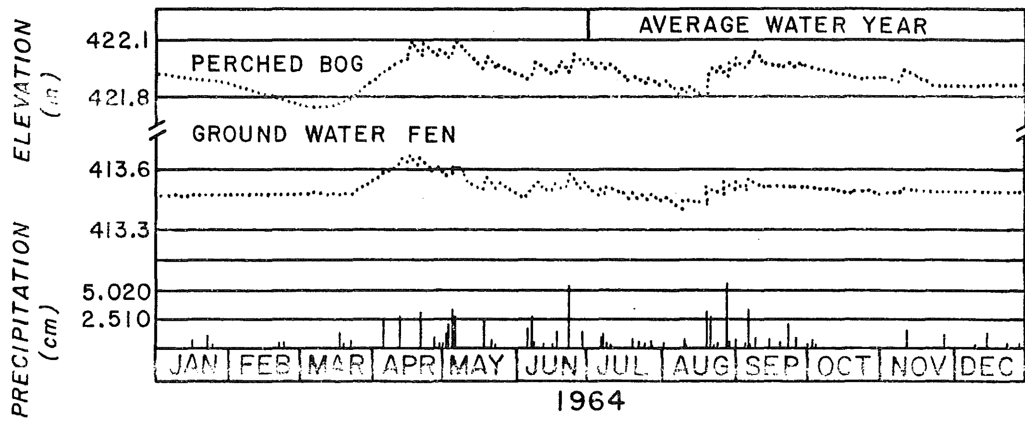


FIGURE 6: WATER LEVELS IN A PERCHED BOG AND GROUND WATER FEN (BAY, 1968)

(4 to 6 feet) in fall and winter. Water tables were also found to be nearer the surface where peat deposits were thicker, for example in the center of a lake-filled peatland.

Bay (1967a) compared water table fluctuations in different types of peatlands and found that the water table remained nearer the surface in groundwater fens than in perched bogs due to continued recharge to the fen from the regional aquifer. Fens also fluctuated less than bogs and runoff from both types was highly related to the position of the water table. Chapman (1965) also found runoff to be related to the water table height in England noting that most of the water moved through the upper 8 cm. Findings contrary to Bay are reported by Podzharov (1975) who states that water table fluctuations were greater for transitional areas and fens than for high bogs (raised bogs). He also stated that water table levels were not related to summer rainfall and large areas with thick deposits were found to have less fluctuations than smaller shallow peat deposits.

b. Groundwater flow.

Groundwater flows in response to hydraulic head. Most of this water movement occurs in the upper layer of peatlands described by Romonov (1961) as the "active layer." He defined this zone as: "the layer of most intense exchange of moisture with the atmosphere, including absorption of precipitation, evaporation into the atmosphere, and runoff of excess water." In this zone water table fluctuations, freezing and thawing occur. The lower boundary of the active layer is the depth where hydrologic characteristics and the degree of decomposition vary rapidly. The thickness of this zone may be from about 30 to 50 cm and is distinguished by high hydraulic conductivities and low bulk densities, less than 0.076 g/cm^3.

Tritium concentrations in peat have been used to describe the presence of the active layer. Tritium in atmospheric precipitation rose highly during the 1960's due to atmospheric testing of nuclear weapons with maximum levels occurring in 1963. Gorham and Hofstetter (1971) measured tritium levels in a Minnesota peat profile and found most of the tritium above the 180 cm depth with maximum concentrations at the 18 cm depth. These results indicate that most of the water moves laterally within peat rather than downward.

The flow rate within peat can be estimated by using hydraulic conductivities and hydraulic gradients. Romanov (1961) estimated flow rates in Eurasian peats to be from 0.86 to 0.00086 cm/day. Rates of water movement in Minnesota peats were estimated to be as high as 36 M/day for surface undecomposed moss peat to 0.49 cm/day for deeper decomposed peats (Bay and Klawitter 1963). These values are higher than other rates reported for peat probably because a hydraulic gradient of 1:1 was used. In Germany, for example, Baden (1976) determined rates of 0.004 to 1.25 cm/day.

Using the hydraulic conductivities from Table 4 and assuming an average hydraulic gradient of 0.001 cm/cm, flow rates were calculated for the major peat types and are presented in Table 8.

Table 8. Calculated flow rates for major peat types.

Peat Type	Hydraulic Conductivity 10^{-5} cm/sec	Calculated Flow Rate cm/day
Fibric	>180	>.147
Hemic	2.1-180	.0017-.147
Sapric	<2.1	<.0017

Sparling (1966) reports methods for actually measuring flow rates in peatlands and suggests using either salt dilution or dye testing. For extremely low flow rates at the surface of a peatland, Sparling suggested using a grided white ceramic plate with capillary tube filled with dye positioned just above the plate. The device is placed just below the water surface and is checked later for migration of the dye.

Very few studies of groundwater flow direction have been made in peatlands. Sander (1976) modeled the groundwater flow in a Minnesota peatland using electric analog techniques. To develop the model, water table levels and outflow rates were measured. The model was fitted by adjusting the contributing watershed from a surface delineation of 19 ha (48 acres) to 1680 ha (4150 acres) and by directing 80 percent of the surface discharge through upwelling "windows" in the peat which were observed on the site (Figure 7). Sander concluded from these studies that Darcy's Law may not apply to flow through the uppermost layer of peat since the flow may actually be surface runoff in interconnected channels.

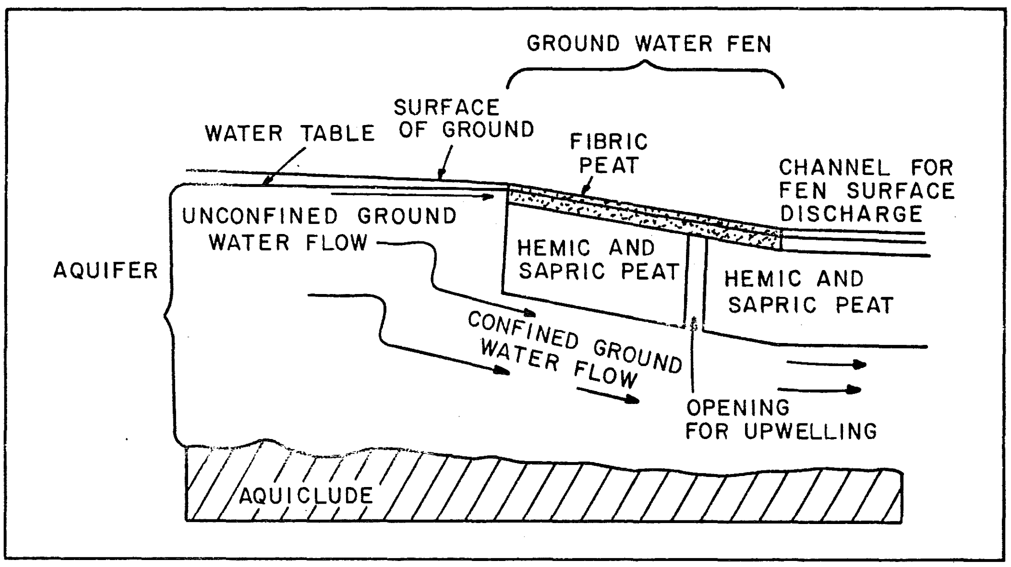


FIGURE 7: SCHEMATIC CROSS SECTION OF GROUND WATER FLOW IN A FEN (FROM SANDER, 1976)

Newman and Dasberg (1977) developed groundwater flow nets for a portion of the Hula Basin peatland in northeastern Israel. This 150 Km^2 peatland is surrounded by mountains and drained by the Jordon River. Peat depths average from two to six meters and the area was drained during the years 1952 to 1957 by two main diversion channels and lateral ditches. To develop flow nets, a regional network of 36 piezometers and 9 observation wells was established at 9 locations together with a local network of 37 piezometers at 12 points on a 70 x 120M plot (Dasberg and Newman 1977). The flow net diagram for January 13, 1975, indicates a mild regional gradient in the peatland with most of the flow moving vertically upward to the water table (Figure 8). This condition persists most of the year due to groundwater inflow from the surrounding mountains except when rainstorms or floods occurred in the winter. The effect of such a rainstorm is shown in the flow net on February 12, 1975, when a gradient reversal in the upper 5 meters caused a downwand groundwater flow. Within a few weeks the gradient returned to an upward direction. Based upon these flow nets, the groundwater contribution to the peatland was estimated to be 40 percent of precipitation.

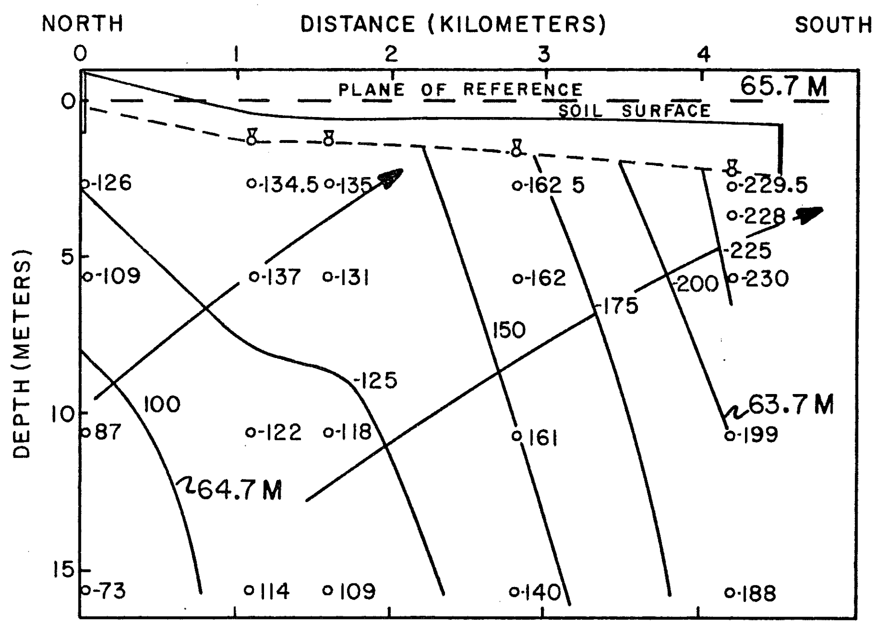
Newman and Dasberg (1977) also measured tritium levels in a peat profile and concluded that groundwater free of tritium reaches the peat from below and mixes with water high in tritium at a shallow depth of one to two meters.

Dixon et al. (1966) used piezometers to determine that artesian flow was occurring vertically upward from a sand acquifer through a silt layer into a Wisconsin peatland. The average upflow rate was estimated to be 0.28 cm/day.

c. Groundwater storage

A high water storage capacity in peatlands is one of their characteristics. The change in storage over time is obtained from water table fluctuations and water yield coefficients (Section I.A.7). Boelter (1964a) notes that undecomposed moss peat contains more than 90 percent water by volume at saturation and releases 50 to 80 percent

JANUARY 13, 1975



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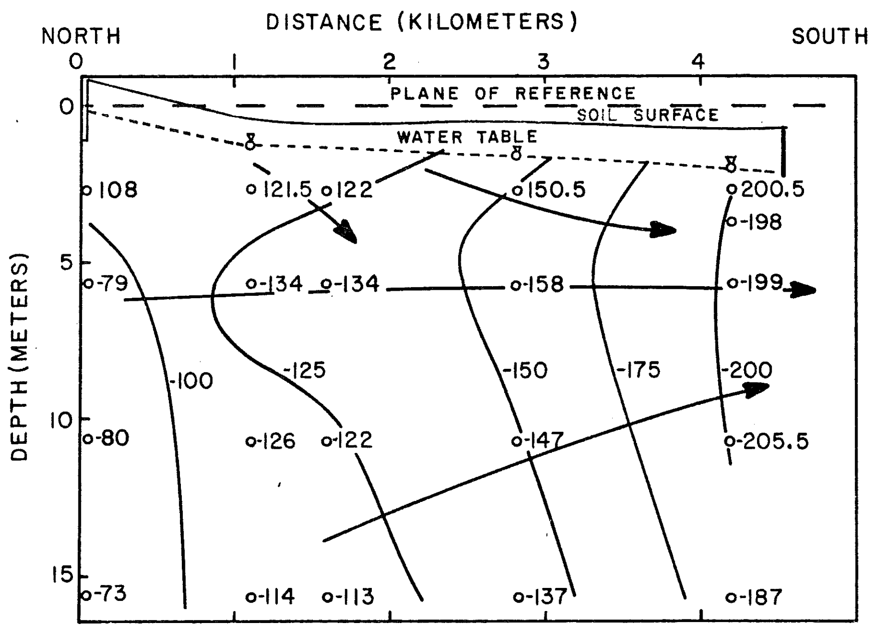

FIGURE 8: HYDRAULIC-HEAD DISTRIBUTION IN THE HULA BASIN IN A N-S DIRECTION ON JANUARY 13, 1975 AND FEBRUARY 12, 1975 (NEWMAN AND DASBERG, 1977)

of this stored water to drainage. Herbaceous and decomposed peats contain 80 to 90 percent water but release only 10 to 15 percent to drainage.

5. Interception

Interception loss is the amount of precipitation which does not reach the ground surface because it is intercepted by plants and evaporated back into the atmosphere (Hewlett and Nutter 1969). Interception (I) is also the difference between gross precipitation (Pg) and net precipitation (Pn) where net precipitation is throughfall (Pt) plus stemflow (Ps):

$$I = Pg - Pn$$

$$Pn = Pt + Ps$$

Throughfall is measured with precipitation gages under the vegetative canopy and stemflow is determined with collars around plant stems that collect the water moving down.

Verry (1976) reported the following relationships for throughfall and stemflow (mm) for black spruce in Minnesota:

$$Pt = 1.600 + 0.934\ (Pg)$$

$$Ps = 0.004 + 0.001\ (Pg)$$

These relationships were based on a 63 to 73 year old black spruce stand having 11 to 15 meter heights, a basal area of 26 to 30 M^2/ha, and a crown cover of 68 percent. Black spruce interception was 29 percent of precipitation for a dry year and 23 percent of precipitation for a wet year. Heikurainen (1964) reported that interception for forested peatlands in Finland ranged from 7 to 23 percent of precipitation, being greater in older stands. Later, Heikurainen (1975) measured an interception of 5 percent of precipitation in a natural peatland.

No interception values were found in the literature for the shrubs and mosses which often dominate peatland vegetation. Interception by such plants could be quite high.

6. Infiltration

Infiltration is the process by which water enters the soil surface (Hewlett and Nutter 1969). The number of infiltration studies on peat soils are few although Keane and Dooge (1975) have conducted a laboratory study of infiltration on a raised bog peat and a blanket bog peat using Phillip's equation in Ireland. They obtained a maximum infiltration with an initial water content of 85 percent. For either higher or lower initial water contents, the infiltration rate was less.

7. Runoff

Runoff as used here is the amount of water leaving a peatland through an outlet channel. When evaluating the runoff from a watershed, both the seasonal distribution of runoff and the response of runoff to storms is of interest.

a. Seasonal distribution.

Most of the runoff from natural peatlands in northern climates occurs in the spring due to snowmelt (Verry and Boelter 1975). Bay (1969) observed that 20 to 25 percent of annual precipitation is stored in the snowpack in northern Minnesota and 66 percent of annual runoff occurred during spring snowmelt. Examples of the seasonal distribution of runoff are shown in Figure 9 for Minnesota and in Figure 10 for Scotland. In Minnesota, runoff may cease in the winter

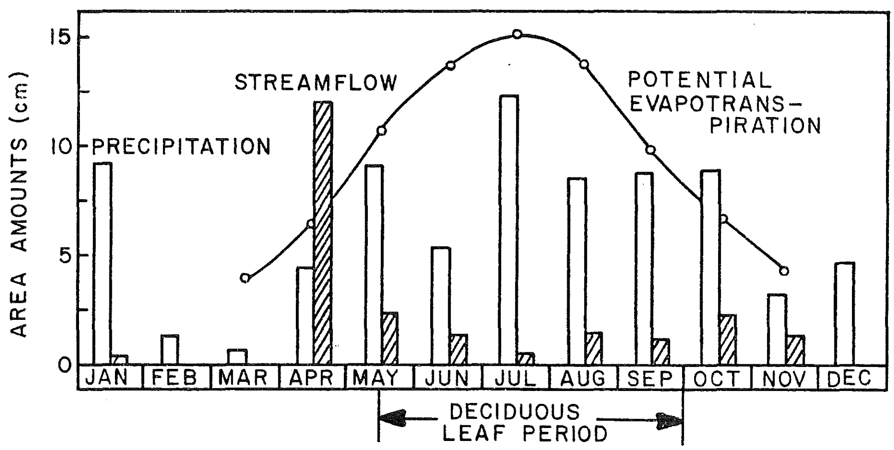

FIGURE 9: MONTHLY PRECIPITATION, STREAMFLOW, AND POTENTIAL EVAPOTRANSPIRATION FOR A PERCHED BOG WATERSHED, 1969 (MODIFIED FROM VERRY AND BOELTER 1975)

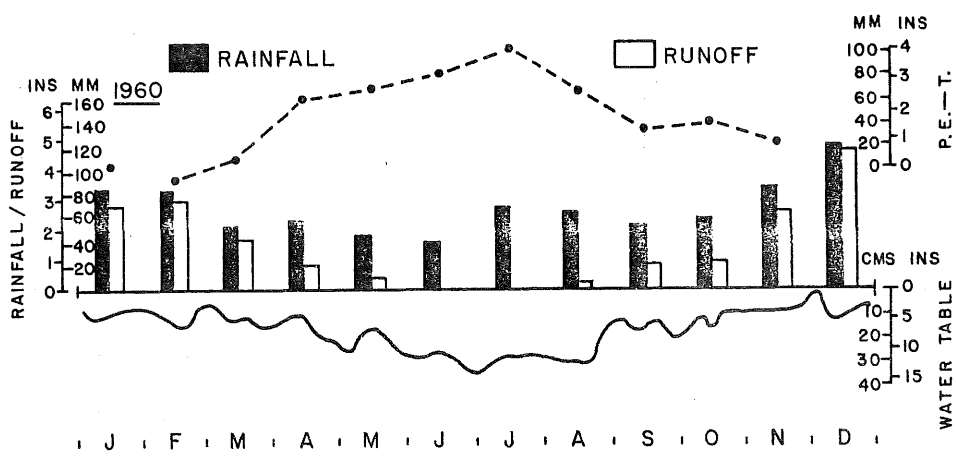

FIGURE 10: RAINFALL, RUNOFF, EVAPOTRANSPIRATION AND WATER TABLE LEVELS FOR A RAISED BOG IN SCOTLAND (FROM ROBERTSON, et. al., 1968)

and become low during the summer due to evapotranspiration losses even though precipitation may be quite high. In Scotland, most of the runoff occurs in the winter months but may cease during the summer. Annual runoff was found to be 56 percent of precipitation for this raised bog in Scotland (Robertson et al 1968). Fens have more uniform runoff than bogs because they are fed by the regional groundwater system (Figure 11). The flow from the fen is nearly constant 70 percent of the time while runoff from the bog is more variable and even ceased 20 percent of the time. Zero flow occurred because the water table in the bog dropped below the outlet (Bay 1969).

b. Stormflow.

The runoff from peatlands occurring after rain storms can be characterized as having long drawn out recessions that delay stormflow (Bay 1969). Two stormflow hydrographs for a small perched bog in northern Minnesota are shown in Figure 12. The recession of the hydrograph during the dormant season approached a straight line quickly and remained above pre-storm levels due to temporary groundwater storage. The recession of the growing season hydrograph continued to drop due to daily evapotranspiration losses. Lag times, the time from the centroid of precipitation to the peak discharge, were found to vary from one to three and one-half hours (Bay 1969). Robertson et al (1968) also observed a delayed runoff response from peatlands during the summer months and attributed the lag to available storage due to lower water tables.

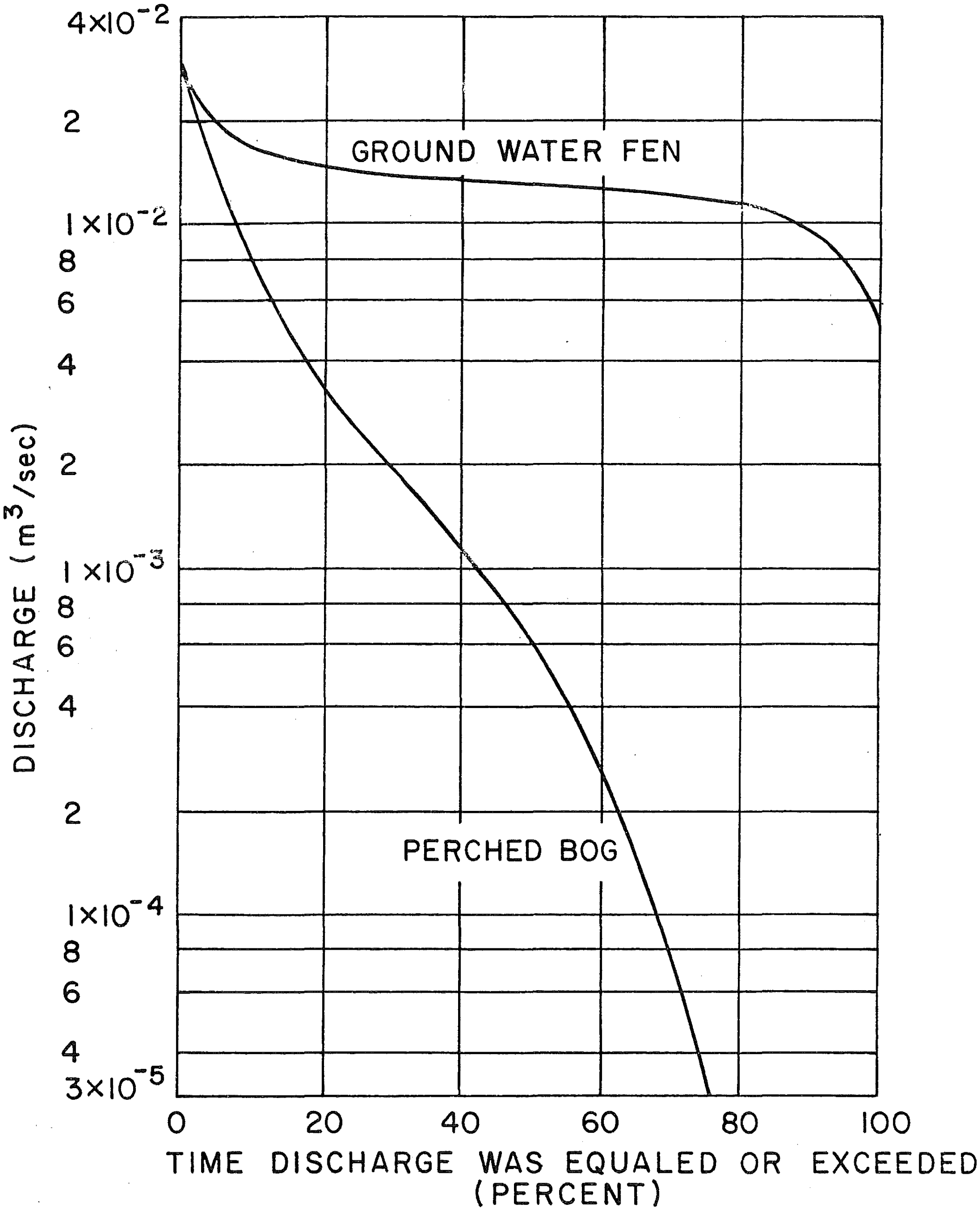

FIGURE 11: FLOW DURATION CURVES FOR A GROUND WATER FEN AND A PERCHED BOG WATERSHED. (FROM: BOELTER AND VERRY, 1977)

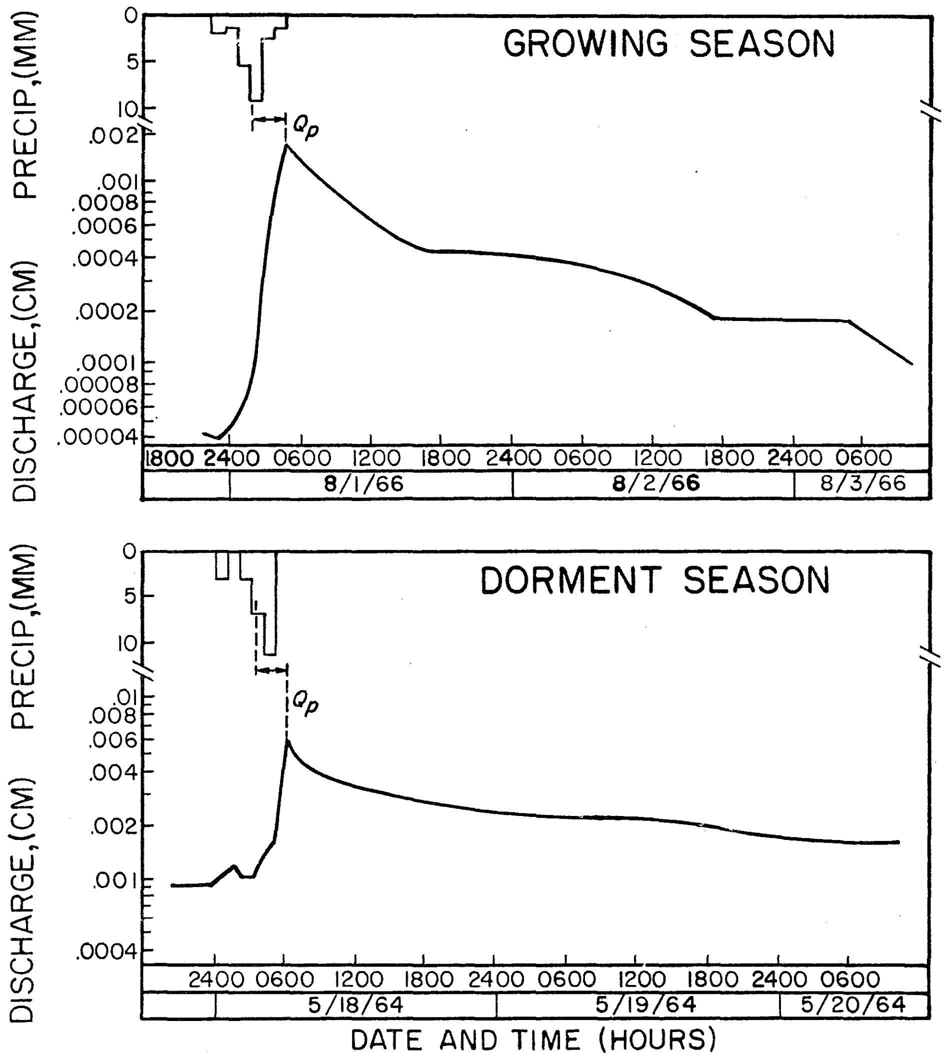

FIGURE 12: HYDROGRAPHS AND HYETOGRAPHS FOR TWO STORMS FROM A SMALL BOG IN NORTHERN MINNESOTA (FROM BAY, 1969).

8. Water Balance of Peatlands

Previous discussions have been on various components of the water balance equation. There have also been several studies of the complete water balance of peatlands. A water balance equation that can be applied to many hydrologic systems is:

$$P = ET + Ro + I \pm \Delta S \pm GW$$

where P = gross precipitation, ET = evapotranspiration, Ro = runoff, I = interception, S = change in storage, and GW = change in groundwater. For peatlands many have used the equation:

$$P = Ro + ET \pm \Delta S$$

(Romanov 1961; Chapman 1965; Bay 1976c; Heikurainen 1974; Dooge 1975; and Eggelsmann 1975). Schendel (1975) used residual rather than ΔS. An additional deep seepage component was suggested by Burke (1975) but was not used while Hommik and Modissoon (1975) added the term "change in groundwater" ($\pm\Delta GW$). Dooge (1975) suggested different equations for bogs and fens, including an extra inflow (I) term on the left side of the fen equation.

The results of several water balance studies are presented in Table 9. The water balance by Bay (1967c) was for the growing season, Burke's (1975) was for eleven months while the others were annual. Runoff, evapotranspiration, and residual values (P-Ro-ET) from the original studies have been converted to a percent of precipitation basis to provide comparison. In Europe, bog runoff varied from 43 to 54 percent of precipitation while ET varied from 44 to 61 percent. The European studies determined ET by lysimetry while Bay (1967c) used the water balance ($ET = P-Ro-\Delta S$). Residual amounts were both negative and positive indicating a groundwater and/or change in storage loss or gain respectively. Burke (1975) and Eggelsmann (1975) also presented monthly water balances. Schendel (1975) observed that ET from a bog in grass was greater

Table 9. Runoff, evapotranspiration, and residual components of water balance equations for various peatlands.

% of Precipitation				
Ro	ET	Resid	Peatland Type	Author, year
42.9	61.0	- 3.9	coastal bog - grass	Schendel, 1975
54.4	49.5	- 3.9	coastal bog - crops	Schendel, 1975
47.2	44.2	+ 8.6	blanket bog	Burke, 1975
37.8	66.4	- 4.2	grassland raised bog	Burke, 1975
--	71.9	--	sphagnum bog	Eggelsmann, 1975b
32.4	68.4	- 0.8	heather raised bog	Eggelsmann, 1975b
40.4	64.3	- 4.7	mineral soil	Eggelsmann, 1975b
22.7	88.2	-10.9	perched bog	Bay, 1967c

than from a bog in agricultural crops with the result that runoff was lower from the grassed area. Schendel (1975) also correlated runoff (mm) to precipitation (mm) for various seasons with the following results:

$$\text{Winter } Ro = 1.14P - 122.70 \qquad R^2 = 0.94$$

$$\text{Summer } Ro = 0.49P - 92.87 \qquad R^2 = 0.81$$

$$\text{Annual } Ro = 0.80P - 241.95 \qquad R^2 = 0.78$$

Eggelsmann (1975) observed that peatlands have a higher evapotranspiration and less runoff than mineral soil watersheds.

9. Summary

The hydrologic cycle of peatlands varies for the peatland type whether bog or fen. Fens receive a major contribution of inflow waters from groundwater sources while bogs are fed only by precipitation. Evapotranspiration varies for different types of vegetation and ET from fens is generally greater than that from bogs. Since ET is reduced when the water table is lowered, most peatland researchers recommend lysimeters for ET measurements. Groundwater movement in peatlands is not well understood although it is commonly recognized that water moves rapidly in surface layers and slowly in deeper layers. Upwelling is also a little known possibility in fens. Water table levels show a seasonal pattern in fluctuations and runoff is related to these levels. Both interception and infiltration have not been sufficiently studied in peatlands. Interception may vary substantially with the plant species and infiltration rates are basically unknown. Most of the runoff from peatlands in northern climates occurs in the spring due to snowmelt. The runoff from fens is more evenly distributed throughout the year than from bogs. Stormflow from peatlands is somewhat delayed with long drawn out recessions.

The preceeding discussions have concentrated on undisturbed peatlands. The following two sections will describe the effects of various management activities on water quantity.

C. EFFECTS OF DRAINAGE ON WATER QUANTITY

1. Introduction

The drainage of a peatland results in changes to the peat itself and therefore to the hydrologic characteristics that describe the peat. Drainage also modifies evapotranspiration, runoff, and groundwater. When drainage is accompanied by clearing, additional changes occur, especially to interception and evapotranspiration.

2. Effects on Hydrologic Characteristics

The foremost effect of drainage on peat is to provide conditions that allow subsidence. Stephens and Stewart (1977) have described six causes of peat subsidence:

a. Shrinkage due to desiccation

b. Consolidation due to loss of buoyant forces of groundwater

c. Compaction by equipment or tillage

d. Wind erosion

e. Burning

f. Biochemical oxidation

The first three causes of subsidence increase the bulk density and reduce the volume of peat without a loss in mass. The last three result in a loss in peat matter. Biochemical oxidation is estimated to be responsible for 65 percent of the total subsidence in the Florida Everglades as compared to 20 percent for the Netherlands and 13 percent for parts of the B.S.S.R. (Stephens and Stewart 1977). Van der Molen (1975) has suggested that biochemical oxidation is the main source of subsidence in fens.

Annual subsidence rates for fens can be estimated from a method developed by Stephens and Stewart (1977):

$$S_x = (0.0169D - 0.1035)2^{(T_x-5)/10}$$

where S_x = subsidence (cm/yr), D = average water table depth (cm), and T_x = average annual soil temperature at 10 cm (^{o}C). From this equation it is evident that subsidence increases as the depth to the water table and soil temperature increase. This increase in subsidence with increasing water table depth was substantiated by Boelter (1972a) who measured subsidence rates in Minnesota of 1.4 cm/yr with the water table at 0.3 m and 2.4 cm/yr with the water table at 0.6 m on a fibric peat.

Eggelsmann (1975) provided an excellent summary of the effects of drainage on the physical and hydrologic characteristics of peat. Bulk density and subsidence increase after drainage while porosity and permeability decrease (Figure 13). The rate of change in these characteristics becomes less as the time of drainage increases. Eggelsmann also indicated that thick peat deposits have less compaction than thinner deposits and dense peats subside less than loose peat. Therefore, subsidence would probably be greater on bogs than on fens.

Piascik (1976) also observed that deep drainage on peat increased bulk density by subsidence and compaction and decreased porosity, water content, and total water holding capacity. Novikov et al (1972) found that clearing of natural peatlands decreased water yield coefficients at the surface by a factor of three times due to the loss of live undecomposed materials.

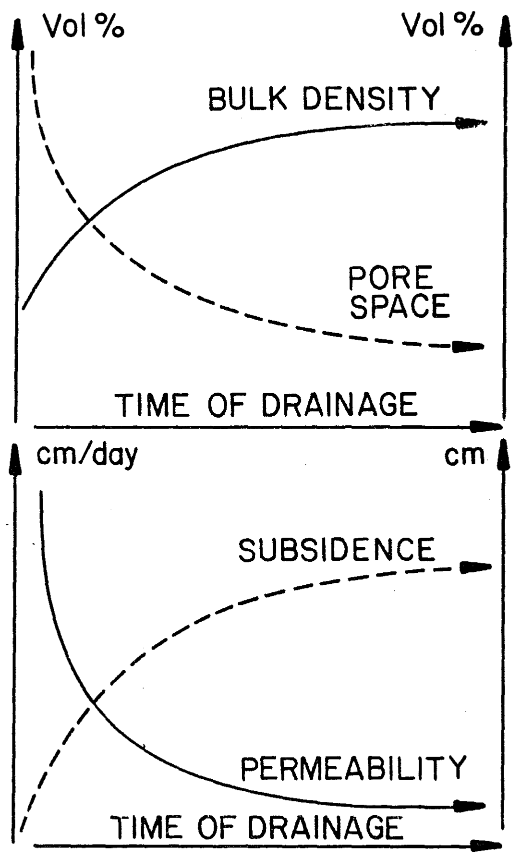

FIGURE 13: CHANGE IN PHYSICAL CHARACTERISTICS OF PEAT WITH TIME OF DRAINAGE (FROM EGGELSMANN, 1975).

3. Effects on Water Balance Components

a. Evapotranspiration.

Drainage results in a reduction of evapotranspiration mainly because the groundwater table is lowered and less water is available for ET (Romanov 1961; Boelter 1972a; Dooge 1975; Heikurainen 1975; Mikulski and Lesniak 1975; and Paivainen 1976). The amount of decrease in annual ET has been estimated to be 15 percent by Mikulski and Lesniak (1975) and 22 to 31 percent by Heikurainen (1975). However, the magnitude of ET reduction varies with drain spacing, peatland type, vegetative type, climate, and the age of drainage.

Huikari (1968) evaluated forest drainage in Finland with drain spacings of 5, 10, 20, 40, 60, 80, and 100 meters and found that ET was greatest on the widest strip which is also the strip having the least water table reduction. Vompersky (1974) also observed that wider drain spacings had greater ET from pine forested peatlands in the USSR.

The effect of drainage on ET also varies with the peatland and vegetative types. Romanov (1961) concluded that drainage and cultivation increases annual ET from raised bogs with most of the increase occurring in the summer. Fens were unaffected. Nonetheless, ET from a drained forested fen was greater than from a drained woodless bog (Vompersky 1974). In comparing ET from drained bogs with different vegetation, Baden and Eggelsmann (1968) found grassland raised bogs had higher ET in spring and summer and lower ET in fall and winter than heather raised bogs, although the annual ET was nearly equal on both areas.

Climate has also been found to influence the ET response to drainage. During wet years, ET from drained areas was greater than from natural areas while during dry years the opposite occurred (Hommik and Madissoon 1975; Heikurainen et al 1978). This reversal is explained by the drained area having increased forest production which would transpire more when water was not limiting as during a wet year but would transpire less with lower water tables in a dry year.

Heikurainen (1975) investigated the effect of the age of drainage on ET changes. Immediately after drainage, evaporation (E) decreased due to the lowering of the water table. After some years, transpiration (T) increased as the forest stand developed. The combined effect was that ET was reduced immediately after drainage but recovered later for areas that were drained for more than 10 years. Schuch (1976) also compared ET on drained areas of different ages. As Table 10 indicates, the old forest had greater ET than the young forest and both were greater than either the natural raised bog or the cultivated area.

In summary, the general effect of drainage is to reduce ET, especially during the summer and in dry years, with a greater ET reduction on densely drained areas and newly drained areas.

Table 10. Evapotranspiration as a percent of precipitation from cultivated, natural, young and old forested peatlands (Schuch 1976).

Peatland Type	ET as % of Precipitation
Cultivated	39
Natural	46
Young forest	69
Old forest	75

b. Groundwater.

The most direct impact that drainage has on groundwater is to lower the water table. Ditching also reduces the range of water table fluctuations (Manson and Miller 1955). The magnitude and extent of water table lowering varies with the peat type and ditch spacing. The effect of the peat type on water table drawdown is illustrated in Figures 14 and 15. For the moderately decomposed hemic peat (Db = 0.17 g/cm^3), the water table drawdown extended to only about five to ten meters from the ditch (Figure 14). However,

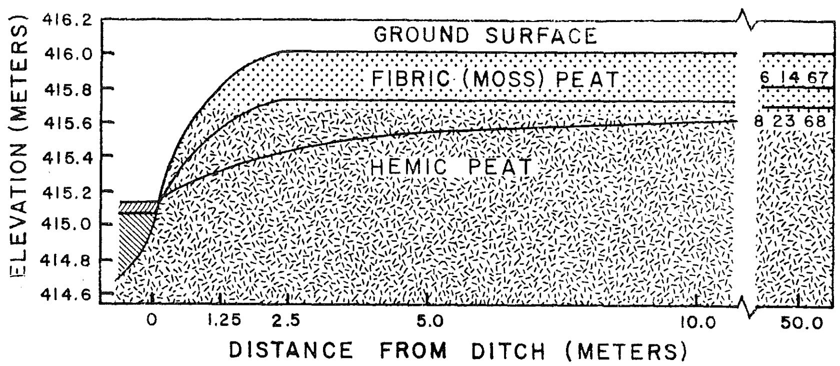

FIGURE 14: WATER TABLE DRAWDOWN IN MARCELL BOG WITH HIGH WATER TABLE (6/14/67) AND LOW WATER TABLE (8/23/68) CONDITIONS. (FROM: BOELTER, 1972a)

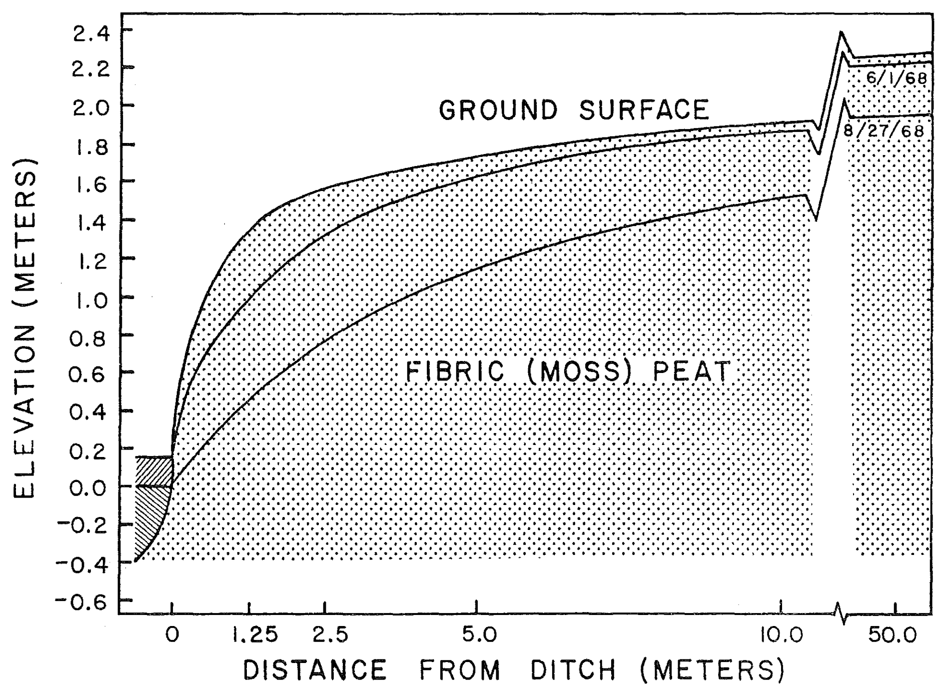

FIGURE 15: WATER TABLE DRAWDOWN IN THE FLOOD-WOOD BOG WITH HIGH WATER TABLE (6/1/68) AND LOW WATER TABLE (8/27/68) CONDITIONS. (FROM: BOELTER, 1972a)

the water table drawdown in an undecomposed fibric peat (Db = 0.08 g/cm^3), extended almost 50 meters from the ditch (Figure 15). Boelter (1972a) concluded that lowering the water level in a ditch in hemic peat did not appreciably lower the water table in the adjacent peatland. Figure 16 illustrates the effect of ditch spacing

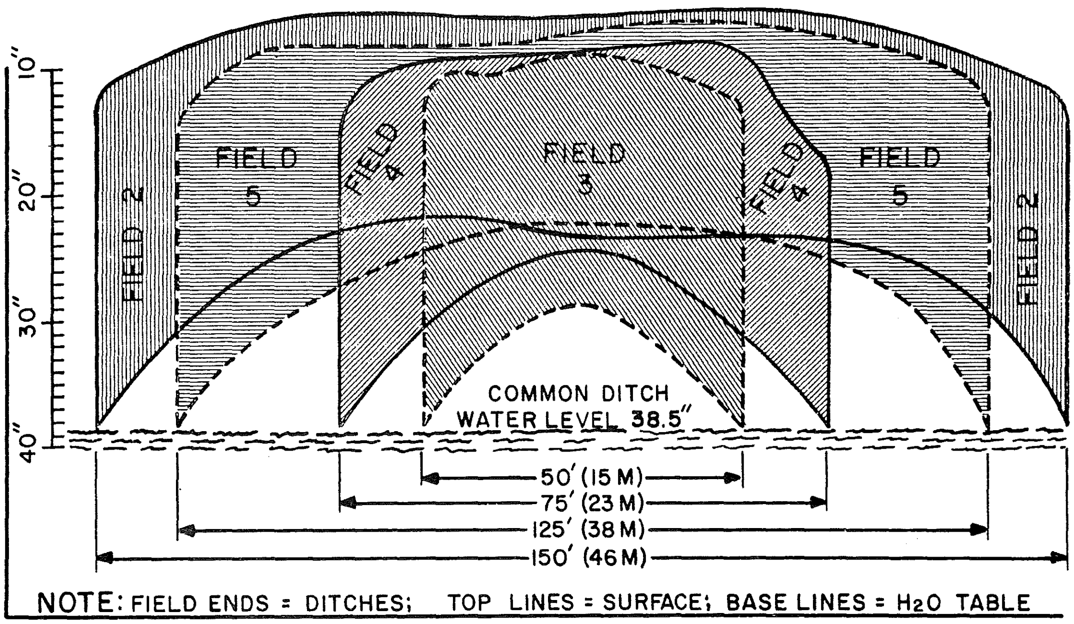

FIGURE 16: WATER TABLE PROFILES FOR VARIOUS DITCH SPACINGS AT THE PINE ISLAND BOG ON JULY 3, 1967 (FROM U.S.D.A., 1967).

on water table drawdown, for a sphagnum bog. Little difference in water table depth in the center of the field occured until ditch spacings were reduced to 15 meters. Huikari (1968) also investigated the effect of drain spacing on water table drawdown. The greatest drawdown occurred with a 5 meter spacing with gradual rises in the water table as the spacing increased to 60 meters. The 80 and 100 meter spacings had water table drawdowns slightly greater than the 60 meter but less than 40 meter spacings. Baden (1976) described the general shape of the water table after drainage as a convex curve having a gradient of 1:1 near the ditch with a logarithmetric decrease in gradient further from the ditch.

Drainage not only alters the shape of the water table but also the pattern of groundwater movement (Figure 17). After drainage, water flowed toward the ditch from both surface and deeper layers of peat. Boelter (1972b) estimated the rate of water movement into the ditch using Darcy's Law as 4×10^{-7} m^3/sec per 100 meters of ditch. Hommik and Madissoon (1975) concluded from water balance studies that drainage increased the groundwater contribution to an area. Burke (1975) observed that drainage resulted in an increased storage in the peat of about 30 percent of precipitation.

There are several methods for determining drain or ditch spacing in peatlands including:

1) Nomographic methods
2) Ellipse equation
3) Electric analog analysis
4) Drainage norms

The method of drainage norms is the most common. Drainage norms are most often considered to be the average depth to the water table during the growing season (Heikurainen 1964). Meshechok (1968) defined the drainage norm as the mean of three monthly water table levels for the four months of June to September. Since the norm may vary from year to year depending on precipitation, drainage norms can be plotted versus precipitation and corrected to normal precipitation.

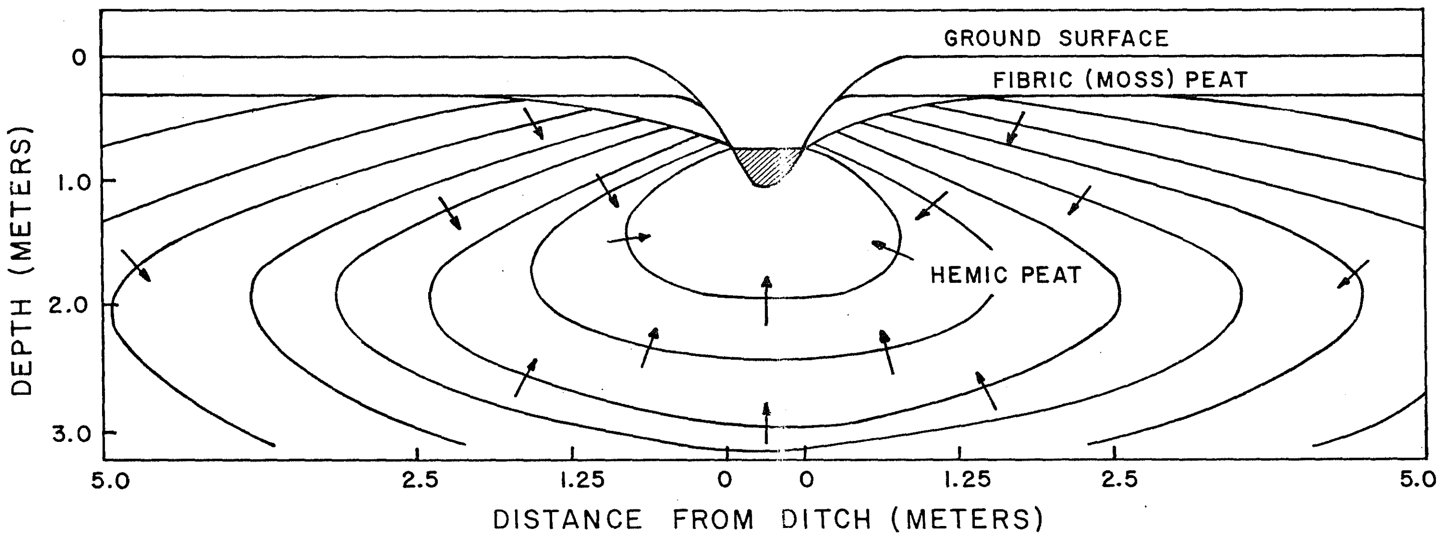

FIGURE 17: FLOW NET SHOWING LINES OF EQUAL HYDRAULIC HEAD AROUND AN OPEN DITCH IN THE MARCELL BOG ON AUGUST 10, 1967. INTERALS BETWEEN LINES REPRESENT HYDRAULIC HEAD DIFFERENCES OF 0.06m. WATER FLOWS PERPENDICULAR TO THESE LINES AS INDICATED BY THE ARROWS. (FROM: BOELTER, 1972a)

A 30 cm drainage norm has been recommended for tree growth.

Heikurainen (1964) developed regression equations to determine drainage norms for certain effective ditch spacings on ombrotrophic and minerotrophic peatlands:

Ombrotrophic $Y = 116.44 - 3.431X + 0.038X^2$ $r = 0.87$

Minerotrophic $Y = 80.279 - 1.618X - 0.011X^2$ $r = 0.85$

where Y = drainage norm expressed as percent of ditch depth and X = effective ditch spacing (meters). The ombrotrophic equation is good for spacings of 10 to 45 meters and a ditch depth of less than 0.9 meters. The minerotrophic equation applies to spacings of 16 to 67 meters and ditch depths less than 0.8 meters. To correct for precipitation departures from normal, Meshechok (1968) suggested:

$$X_n = X_{300}\ (300/P_n)$$

where X_n = effective ditch spacing (meters) for normal precipitation (P_n) in millimeters for the months June through September and X_{300} = effective ditch spacing (meters) for a precipitation of 300 mm. Drainage norms are presented in Figure 18 and Table 11. For example, to achieve the desired 30 cm drainage norm for tree growth, ditches 80 cm deep should be spaced 46 meters apart in an ombrotrophic bog (from Figure 18) and 30 meters apart in a minerotrophic fen (from Table 11). As expected, closer ditch spacings are required for the more decomposed peat.

Ditch capacity can also be considered in drainage design projects. Finland uses the 20-year peak discharge for determining ditch capacity by the formula:

Table 11. Drainage norm in cm for various ditch intervals and depths (precipitation June - September = 300 mm) (from Meshechok 1968).

Ditch depth in cm	Effective ditch distance in metres								
	5	10	15	20	25	30	35	40	45
	Drainage norm (h) in cm								
	For minerotrophic peat								
20	16.8	14.0	12.1	10.6	9.4	8.4	7.5	6.7	6.0
30	25.2	21.0	18.2	15.9	14.0	12.5	11.2	10.1	9.1
40	33.6	28.0	24.2	21.2	18.7	16.7	15.0	13.4	12.1
50	42.0	35.0	30.3	26.5	23.4	20.9	18.7	16.8	15.1
60	50.4	42.0	36.3	31.8	28.1	25.1	22.4	20.1	18.1
70	58.8	49.0	42.4	37.1	32.8	29.3	26.2	23.5	21.1
80	67.2	56.0	48.4	42.4	37.4	33.4	29.9	26.8	24.2
90	75.6	63.0	54.5	47.7	42.1	37.6	33.7	30.2	27.2
100	84.0	70.0	60.5	53.0	46.8	41.8	37.4	33.5	30.2
	For ombrotrophic peat								
20	18.6	16.5	14.5	12.7	11.1	9.9	8.9	8.1	7.6
30	27.8	24.8	21.8	19.1	16.7	14.9	13.4	12.2	11.4
40	37.1	33.0	29.0	25.4	22.2	19.8	17.8	16.3	15.2
50	46.4	41.3	36.3	31.8	27.8	24.8	22.3	20.4	19.0
60	55.7	49.5	43.5	38.1	33.3	29.7	26.7	24.4	22.8
70	65.0	57.8	50.8	44.5	38.9	34.7	31.2	28.5	26.6
80	74.2	66.0	58.0	50.8	44.4	39.6	35.6	32.6	30.4
90	83.5	74.3	65.3	57.2	50.0	44.6	40.1	36.6	34.2
100	92.8	82.5	72.5	63.5	55.5	49.5	44.5	40.7	38.0

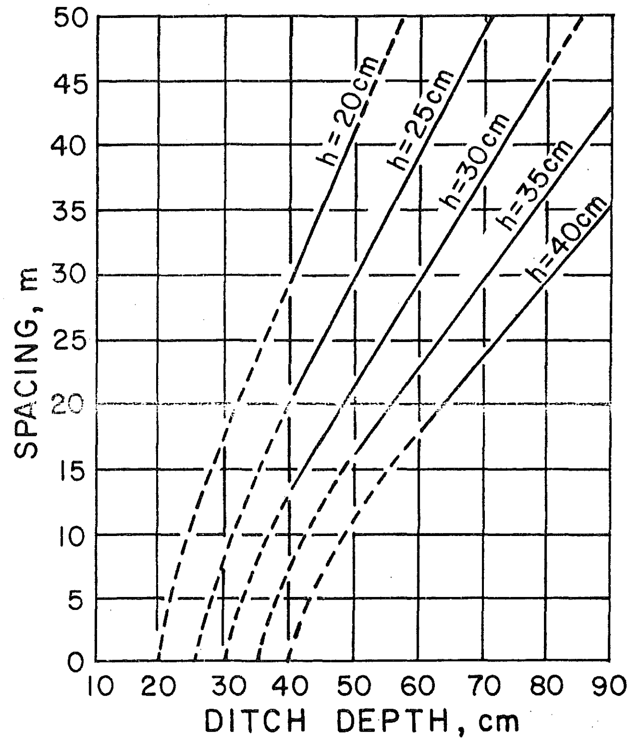

FIGURE 18: ATTAINMENT OF CERTAIN DRAINAGE NORMS (h) FOR AN OMBROTROPHIC BOG FOR VARIOUS DITCH SPACINGS AND DEPTHS. (PRECIPITATION FROM JUNE THROUGH SEPTEMBER = 300 mm) (FROM: HEIKURAINEN, 1964)

$$Q_{20} = AC\ (RS)^{.5}$$

where Q_{20} = 20 year peak discharge, A = cross-section area of ditch, C = speed coefficient, R = hydraulic radius, and S = ditch slope. A trial and error method is used to obtain the desired design discharge (Heikurainen 1964).

Wertz (1968) has developed a method of obtaining drain spacings based on the water content of peat:

$$X = 23.8\ (1/(90.5 - W)\ q)^{0.5}$$

where X = drain spacing in meters, W = water content $\leq 89\%$, and q = flow rate per unit area (l/sec/ha).

In summary, drainage lowers the groundwater table and local groundwater movement is modified to flow toward the ditch. The influence of water table drawdown extends further into bogs than fens and therefore to achieve a desired drained depth, closer ditch spacings are needed in fens than in bogs. Drainage also increases the storage capacity of the peat.

c. Interception

The effect of drainage alone on interception is dependent on the vegetative response. Heikurainen (1975) determined that interception increased 5 to 6 percent of precipitation on newly drained areas and 20 percent of precipitation on older drained areas. Later, Heikurainen et al. (1978) stated that drainage results in higher interception on peatlands due to increased vegetative response. This response only applies to the situation when the vegetation is not cleared. Vegetation clearing, associated with peat mining projects, would result in complete elimination of interception with increased precipitation reaching the peat surface.

d. Runoff

The effect of drainage on runoff is one of the most controversial questions in peatland hydrology. McDonald (1973) discusses this controversy and presents two philosophies:

(1) Because drainage increases the natural drainage system, the sensitivity of runoff to precipitation is increased.

(2) Because drainage produces water storage capacity, the sensitivity of runoff to precipitation is reduced.

Those who believe drainage increases runoff sensitivity (Conway and Millar 1960) have observed the following changes in the storm hydrograph (Figure 19):

(1) Time of concentration is reduced

(2) Peak discharge is increased

(3) Recession curve is steeper

(4) Base flow is lowered

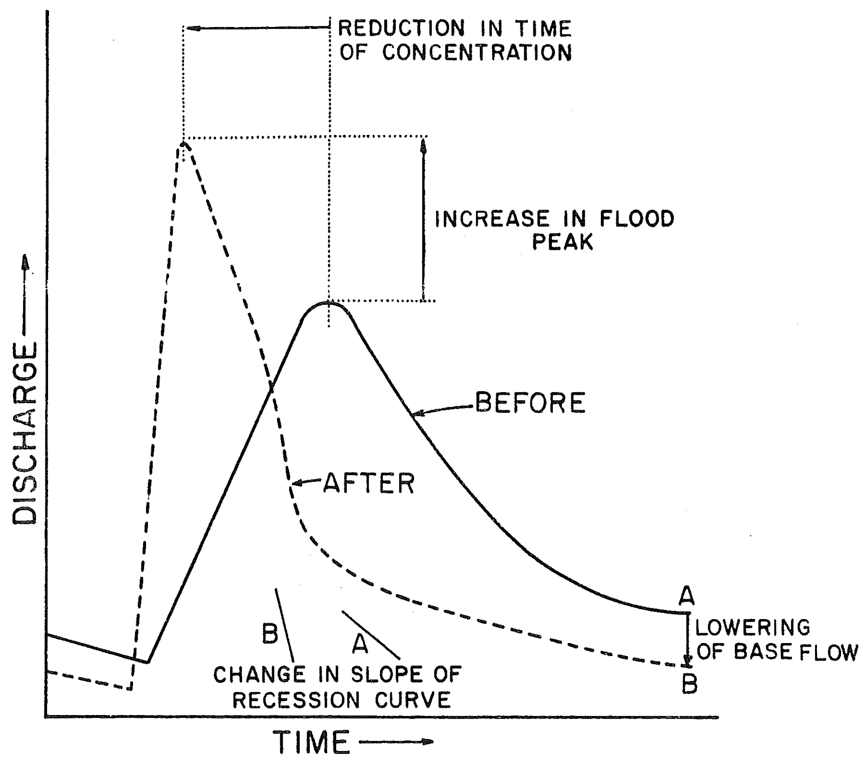

FIGURE 19: DIAGRAM OF HYDROGRAPHS BEFORE AND AFTER DRAINAGE ASSUMING INCREASED RUNOFF SENSITIVITY (FROM McDONALD, 1973).

Contradictory findings that drainage results in a reduction of runoff sensitivity (Burke 1963) would produce these changes in the storm hydrograph (Figure 20):

(1) Time of concentration is increased

(2) Peak discharge is reduced

(3) Recession curve is flatter

(4) Base flow is increased

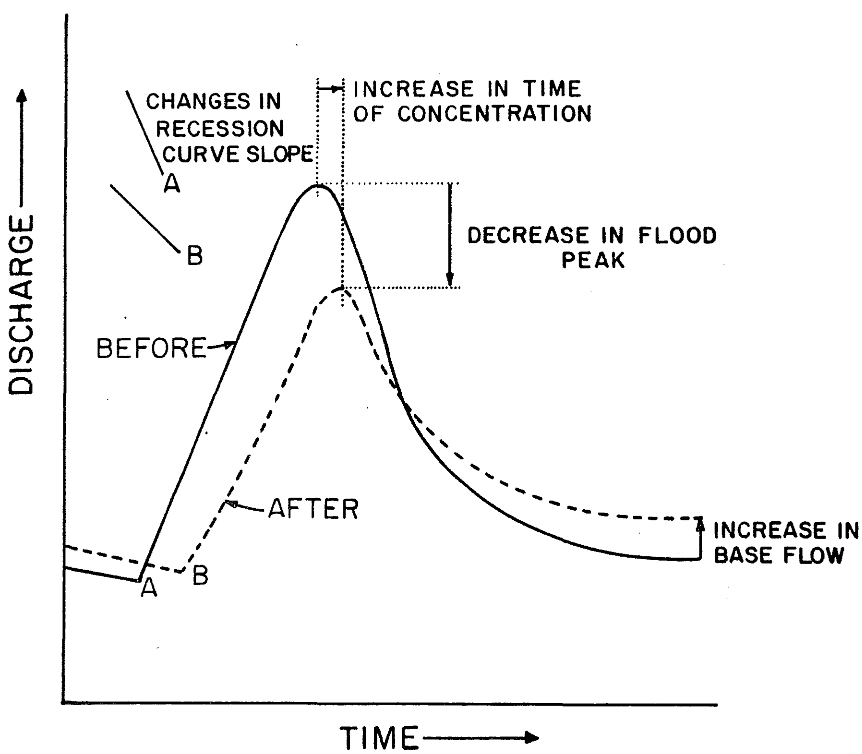

FIGURE 20: DIAGRAM OF HYDROGRAPHS BEFORE AND AFTER DRAINAGE ASSUMING DECREASED RUNOFF SENSITIVITY (FROM McDONALD, 1973).

McDonald attributed these opposing results to different peat types and study areas, indicating that when sphagnum peat is drained increased runoff occurred due to subsidence. Actually, the effect of drainage on runoff varies with the spacing, age, and type of drainage; the peat or peatland type, the vegetation type and climate. Table 12 summarizes the results of 24 studies investigating the effects of drainage on runoff. In most cases, drainage increased annual, minimum, and seasonal runoff and decreased peak runoff. Increases in annual flows ranged from a slight rise to 60 percent. Peak flows decreased up to 30 percent although Seuna (1974) and Mustonen and Seuna (1975) reported increases in peak flows. Minimum flows increased 20 to 150 percent, usually during the summer low flow period. To better understand these varying responses of runoff to drainage the effects of drainage spacing, age, and type; peatland type; vegetation type; and climate on runoff response are discussed.

1. Ditch spacing.

Generally, runoff increases as ditch spacing decreases (Ferda and Novak 1976; Heikurainen 1964; Huikari 1968; and Vompersky 1974). Ferda and Novak (1976) found in a Czechoslovakian transition peatland, drained for forestry, that drainage increased peak flow 15 to 56 percent for the 30 meter spacing while slight decreases in peak flow were observed for the 60 and 100 meter spacings (Figure 21). Minimum flows were also observed to increase after drainage with greater increases occurring for closer ditch spacings. Narrow ditch

Table 12. Effects of drainage on runoff.

	WATERSHED CHARACTERISTICS				FLOW CHARACTERISTICS*					
NO.	PEATLAND TYPE	DRAINAGE TYPE	DITCH SPACING	USE	ANNUAL	PEAK	MINIMUM	SEASONAL	LOCATION	AUTHOR
1.	Unknown	Ditch	50-90M	Forestry	↑	↓		↑ Summer	Finland	Heikurainen, 1964
2.	Unknown	Ditch	Unknown	Forestry	↔ Recent ↓ Old	↓			Finland	Heikurainen, 1975
3.	Unknown	Ditch	Unknown	Forestry		↓	↑		Finland	Heikurainen, 1976
4.	sedge bog	Ditch	Unknown	Forestry		↓	↑		Finland	Heikurainen, et al. 1978
5.	Pine bog	Ditch	5,10,20,40, 60,80,100M	Open Forestry	↑ As spacing↓		↑ <40M open <20M forest		Finland	Huikari, 1968
6.	Open bog	Ditch Plough Drain	40% of Wtsd.	Forestry	↑ 43%	↑ 31% ↑131%	↑	All ↑ months	Finland	Seuna, 1974
7.	Open bog	Ditch Plough Drain	80M/ha 275M/ha	Forestry	↑ 43%	↑ 31% spring ↑131% storm	↑	All ↑ months	Finland	Mustonen and Seuna, 1975
8.	Sedge bog H2-3	Ditch, Drain Narrow, Ditch	Unknown	Forestry	Open> Drain> Narrow				Finland	Paivenen, 1976
9.	Bog Fen	Ditch	55,108,220M	Forestry				↓ summer	USSR	Vompersky, 1974
10.	Valley Bogs	Ditch	33% of wtsd	Unknown	↓	↓			USSR	Moklyak et al. 1975
11.	Unknown	Unknown	6 wtsds. 6-25% drained	Agric.	↑ (5)		↑	↑ Spring	USSR	Bulavko and Drozd, 1975
12.	High Bog Fen	Ditch	MODEL	Agric.	↓ Slight ↑				USSR	Hommik and Madissoon, 1975
13.	Bog	Theoretical			↑ Initial				BSSR	Lundin, 1975
14.	Bog	Ditch	60-200M	Forestry		↓	↑		Latvia	Ayre, 1977
15.	Unknown	Ditch	16 wtsds 5-25% drained	7-49% peatland Ag & For.	↑ 10-20% (9) ↔ (7)	↓17-30% (7) 10-30% (6)	↑ 20-130% (8) ↑ 30-150% (16)	Winter Summer	BSSR	Klueva, 1975
16.	Transition	Ditch & Drains	30,60,100M	Forestry		↑ <30M ↓ 60-100M	↑ All		Czecho- slovakia	Ferda and Novak, 1976
17.	Bog Fen	Ditch	800M	Agric. & Forestry	↑ 20% fen			↑Summer Fall	Poland	Mikulski and Lesniak, 1975
18.	Raised bog	Tile drains	Unknown	Natural Agric.	Approx. Same	↓ 2-4X			W. Germany	Baden and Eggelsmann, 1968
19.	Raised bog	Tile drains	Unknown	Natural Agric.		Delayed ↓ flatter			W. Germany	Eggelsmann, 1975 a
20.	Raised Bog	Ditch	Unknown	Agric. Forestry	↔Same ↓				W. Germany	Schuch, 1976
21.	H5-6	Ditch Plough Drain	25,50,100 ft. 8,12,16,20 ft.	Agric.	Approx. Same	↓	↑		Ireland	Burke, 1968
22.	Blanket Bog H5	Ditch	< 3.5M	Agric.	↑ 60%	↓	↑	↑	Ireland	Burke, 1975
23.	Bog Fen	Literature Review				↓ Bog		↑Spring Fen	Ireland	Dooge, 1975
24.	Bog	Literature Review			↑	↑↑↓	↑↓		Scotland	McDonald, 1973

* ↑Increase; ↓Decrease; ↔ No change.

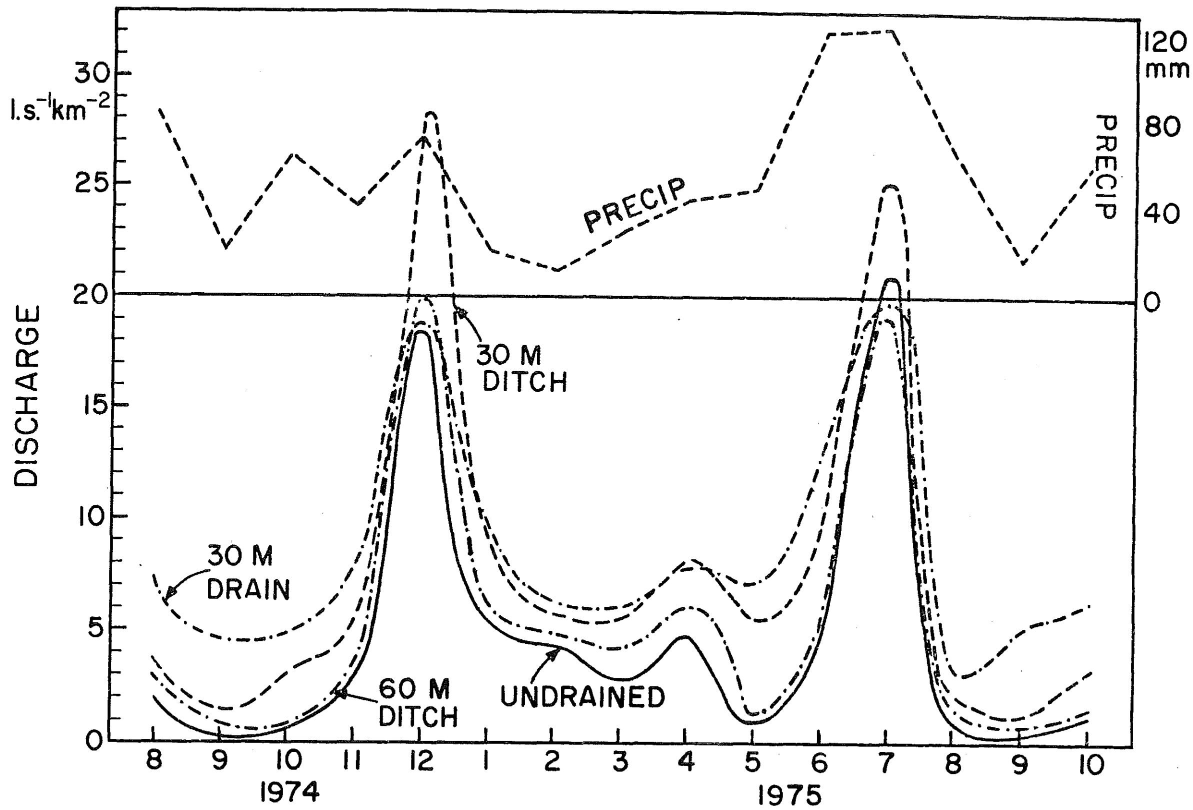

FIGURE 21: RUNOFF FROM A NATURAL AND A DRAINED PEATLAND IN CZECHLOSLOVAKIA (FROM FERDA AND NOVAK, 1976).

spacing is also needed to increase peak runoff from forested peatlands in Finland (Heikurainen 1964; Huikari 1968). As Figures 22 and 23 indicate, ditch spacings of less than 20 meters were needed to actually increase runoff as compared to predrained conditions (Huikari 1968). Vompersky (1974) also evaluated the effect of ditch spacing on runoff in the USSR and reports that average runoff for a drained, oligotrophic bog would be 0.070, 0.057, and 0.048 l/sec/ha for ditch spacings of 56, 108, and 220 meters respectively.

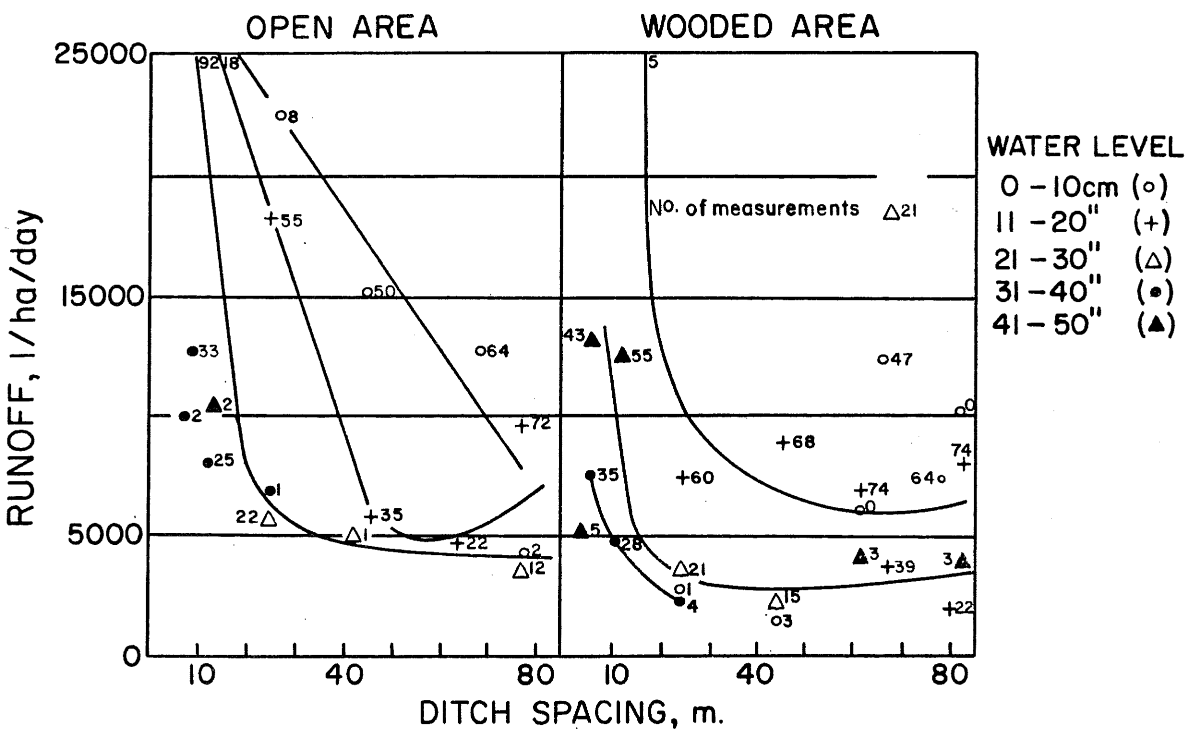

FIGURE 22: DITCH SPACING, SURFACE RUNOFF, AND GROUND WATER LEVEL FOR OPEN AND WOODED AREAS DURING A WET YEAR IN FINLAND (FROM HUIKARI, 1968)

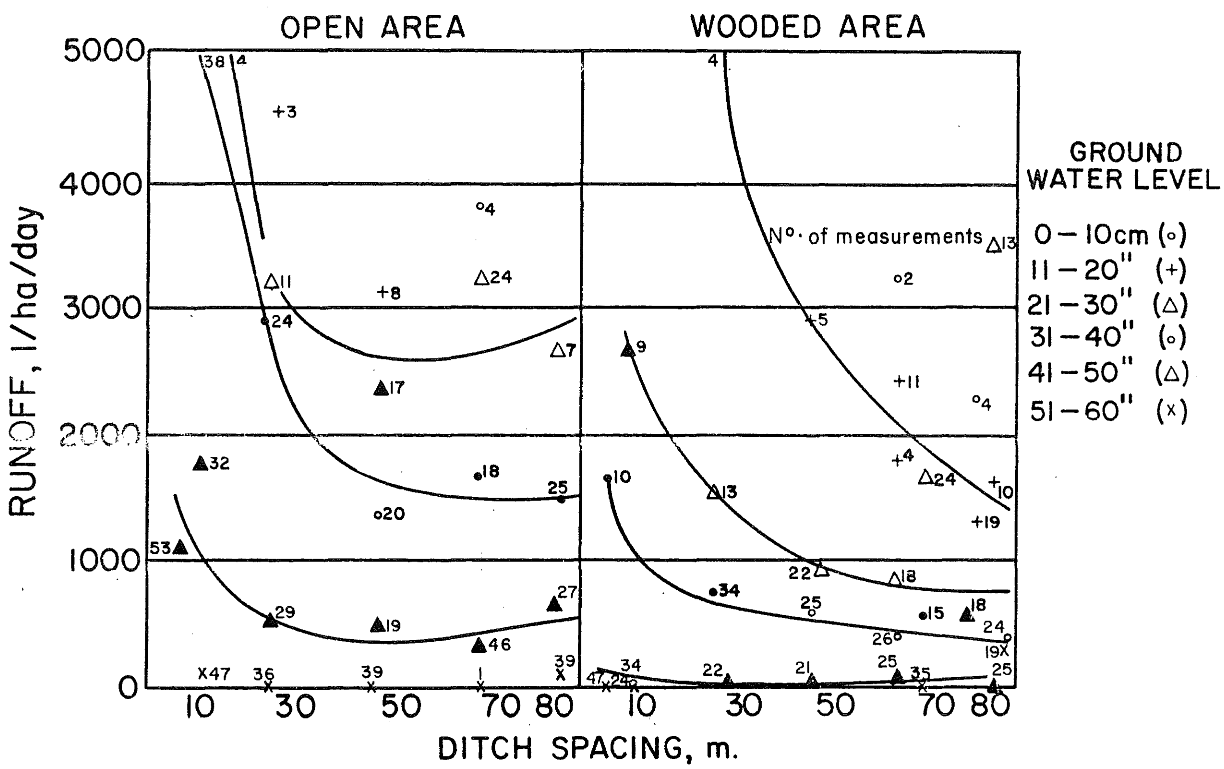

FIGURE 23: DITCH SPACING, SURFACE RUNOFF, AND GROUND WATER LEVEL FOR OPEN AND WOODED AREAS DURING A DRY YEAR IN FINLAND (FROM HUIKARI, 1968).

2. Age of Drainage.

The effects of forest drainage on runoff vary with the age of drainage. In comparing recently drained (less than 10 years old) to older drained areas, Heikurainen (1975) found that annual runoff decreased from 35 to 25 percent of precipitation with increasing age. It was concluded that as the tree stand develops, lower flood peaks of longer duration would occur due to increased water storage capacity and delayed snowmelt. Similar findings have been made by Schuch (1976) in West Germany, who obtained a decrease in runoff from 30.6 to 25.3 percent of precipitation for young and old drained forests respectively.

3. Type of Drainage.

Paivanen (1976) compared runoff from plots drained by three methods: a) ordinary open ditches, b) plastic pipe drains, and c) narrow, vertical-walled ditches. Average runoff was greater for the ordinary ditch and least for the narrow ditch, however, the duration of runoff was greater for the pipe drains and narrow ditches. Ferda and Novak (1976) showed that ditches spaced at 30 meters yielded higher peak flows but lower minimum flows than drains at the same spacing (Figure 21).

4. Peatland type.

Vompersky (1974) compared area-runoff from a drained minerotrophic fen and a drained oligotrophic bog and noted that summer, fall, and total annual runoff were greater from the bog. McDonald (1963) feels that the reason for greater runoff from drained bogs is due to greater subsidence resulting in increased bulk density. Hommik and Madissoon (1975) developed a model to predict runoff from complex peatland watersheds:

$$Ro = 0.11\ A_b + 0.43\ A_n + 0.56\ A_u + 0.83\ A_{db} + 0.05\ B_d + 0.97\ B_u$$

$$- 0.31\ C + 9.6\ q - 31.6 \qquad (R = 0.90)$$

where Ro = runoff, A_b = % of watershed as fen, A_n = % as bog, A_u = % as wet soil with brush, A_{db} = % as drained fen, B_d = % as woods on normally wet soils, B_u = % as woods on excessively wet soils, C = % as on open soils of normal wetness, and q = mean minimum runoff (daily runoff below 95 percent exceedance value). From this equation it is apparent that if the watershed was 80 percent bog and 20 percent fen, the runoff would be twice as much as a watershed having 80 percent fen and 20 percent bog. This equation ignores the variables: % as drained bog and regional groundwater contributions to fens.

5. Vegetative Type

Mustonen and Seuna (1975) and Vompersky (1974) concluded that drainage of an open bog would result in a greater increase in annual runoff than would drainage of a forested bog due to increased transpiration on the forested area. Schuch (1976) also compared runoff from drained forested, drained cultivated, and natural raised bogs (Table 13). Runoff was less from the drained forested area as compared to either the natural or cultivated areas. Eggelsmann (1975) also compared runoff and groundwater levels from different vegetative types. A grassland raised bog had a higher peak runoff and lower

Table 13. Runoff as a percent of precipitation from cultivated, natural, young and old forested peatlands (Schuch, 1976).

Peatland Type	Runoff as a % of Precipitation
Cultivated	60.6
Natural	54.0
Young forest	30.6
Old forest	25.3

minimum flow than a heather covered bog (Figure 24). The peak flow from the grassland was delayed as compared to the heather bog.

6. Climate

The effect of drainage on runoff varies for wet versus dry years. Burke (1968) observed that during prolonged dry periods, runoff was negligable from both drained and natural areas. When rainfall followed a long dry period, runoff started and reached its peak sooner on natural areas, although the total runoff was greater for the drained area. The greater storage capacity of the drained area was said to cause the attenuating effect. During prolonged wet periods, total runoff was similar from both drained and natural areas but the flow was more uniform from the drained area and peak flow was greater on the undrained area. Heikuri (1968) observed that runoff from drained forested areas was 58 percent of open area runoff during wet years but was only 22 percent of open area runoff during dry years (Figures 22 and 23). As expected, runoff comprises a greater percentage of precipitation during a wet year (36-37%) than in a dry year (5-23%) (Paivanen 1976).

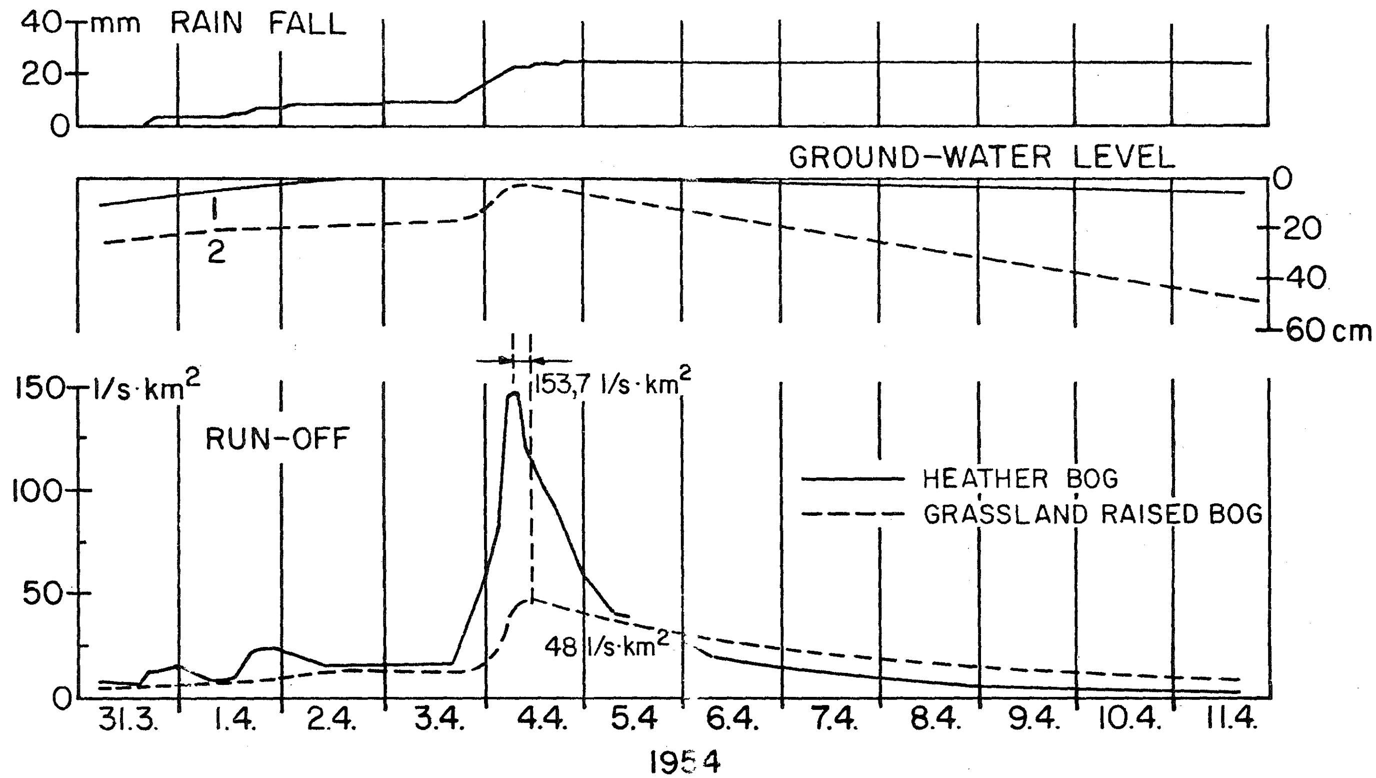

FIGURE 24: RAINFALL, GROUND-WATER LEVEL, AND RUNOFF FOR A HEATHER BOG AND A GRASSLAND RAISED BOG (FROM EGGELSMANN, 1975).

In summary, drainage generally appears to decrease peak runoff and increase annual, minimum and seasonal runoff, thus redistributing the flow more evenly throughout the year. The effect of drainage on runoff varies with the spacing, age, and type of drainage; the peat or peatland type; the vegetative type; and climate. Runoff increases as ditch spacing decreases and decreases with the age of drainage. Total annual runoff from drained bogs is greater than from fens and is also greater from woodless bogs than forested bogs. Differences in runoff changes are more pronounced in dry years. No studies could be found on the effect of drainage associated with peat mining on runoff.

4. Summary

The general effects of drainage on water quantity are presented in Table 14. Two notable exceptions to this summary are that clearing accompanied by drainage would eliminate interception rather than increase it and evapotranspiration may increase on some forested bogs.

Table 14. Summary of the effects of drainage on water quantity.

Parameter	Effect: Increase	Effect: Decrease
Hydrologic Characteristics		
subsidence	X	
bulk density	X	
porosity		X
water content		X
water holding capacity		X
water yield coefficients		X
Water Balance Components		
evapotranspiration		X
groundwater level		X
groundwater storage	X	
interception	X	
peak runoff		X
annual runoff	X	
minimum runoff	X	
runoff timing		X

D. EFFECTS OF TIMBER HARVESTING ON WATER QUANTITY

1. Introduction

Timber harvesting on peatlands is a common practice in many European countries and also occurs to a lesser extent in the United States. Since timber harvesting often accompanies drainage projects for peat mining operations, this topic is included in the literature review. Harvesting of timber on peatlands has been found to affect precipitation, interception, evapotranspiration, groundwater levels, and runoff.

2. Precipitation

Bay (1958) evaluated the snow accumulation and melt for five cutting methods in Minnesota:

east-west strip cut

shelterwood

tree selection

1/2 acre clearcut

uncut control

The E-W strip cuts were on a black spruce peatland with cut strips 30 meters wide and leave strips of 46 meters. Snow accumulation was highest on the 1/2 acre clearcut and second on the E-W strip cut. The E-W strip cut resulted in the slowest snowmelt of the five methods due to shading. These results indicate that different methods of timber harvesting on peatlands result in changes to the precipitation that reaches the site as well as affecting the timing of snowmelt.

3. Interception

Paivanen (1968) reports that timber harvesting reduces interception on peatlands. For example, a Scots pine stand with 50 percent canopy coverage in Finland has interception values of 13.6 to 30.6 percent of precipitation. This interception would be eliminated by timber harvesting with a subsequent rise in net precipitation.

4. Evapotranspiration

Generally, a reduction in evapotranspiration would be experienced by timber harvesting on peatlands (Paivanen 1968). This effect would be more pronounced initially and would be reduced as a new forest stand developed.

5. Groundwater

During the first growing season after clearcutting strips on a black spruce bog in Minnesota, no change in groundwater levels were observed (Bay 1970). However, lower water levels in these strip cuts occurred during the second and third years after harvesting (Boelter 1974). It was concluded that timber harvesting would not be expected to change water levels on fens due to continuous groundwater inputs but on perched bogs water levels could change, the direction depending upon climate. During dry periods, the water table in the bog would be lower after timber harvesting due to increased evaporation by surface winds and by increased transpiration by invading sedges which remove water from lower depths. During wet periods, water tables in harvested bogs would be higher due to reduced interception. Strip cutting would appear to increase water table fluctuations.

Paivanen (1968) observed a rise in the groundwater table during the summer after clearcutting on both Norway spruce and Scots pine peatlands.

6. Runoff

Timber harvesting on drained areas generally results in an increase in runoff which is expected since interception and evapotranspiration are reduced and groundwater levels rise (Paivanen 1968; Heikurainen *et al*. 1978).

7. Summary

The general effects of timber harvesting on water quantity are presented in Table 15.

Table 15. Summary of the effects of timber harvesting on components of the water balance equation.

Water Balance Component	Effect Increase	Effect Decrease
Net Precipitation	X	
Interception		X
Evapotranspiration		X
Groundwater level	X	
Runoff	X	

II. WATER QUALITY

A. INTRODUCTION

The quality of waters leaving peatlands is influenced by the quality of waters entering peatlands, whether it be precipitation or groundwater, and the changes that occur to that water within the peatland system. Such changes are caused by vegetation, peat types, the chemical content of the peat, microbial activity, and also by water quantity. In order to differentiate among these potential changes it would be helpful to classify peatlands as "Hydrologic Peatland Types." Also, to be able to predict changes in water quality as a result of peatland disturbances it is first necessary to realize the water quality of natural peatlands. Finally, since there are so many water quality parameters to describe the peatland water system, an understanding of the importance of these parameters is needed.

B. HYDROLOGIC PEATLAND TYPES

There are numerous peatland classification systems that have been developed based upon floristic, physical, chemical, or hydrologic characteristics (Sjörs 1950; Romanov 1961, Heinselman 1963; Moore and Bellamy 1974). Two such classification systems, based upon the quality of waters within the peatland are given in Table 16. In the United States, the most commonly used hydrologic peatland types are:

1. Ombrotrophic bog
2. Transition
3. Minerotrophic fen

Table 16. Classification of peatlands based on mean concentrations of major ions for western Europe and Scandanavia.

Hydrological Mire[a]		pH	HCO_3	CI	SO_4	Ca	Mg	Na	K	H	Total
					Total Major Ions meg/l						
Type 1		7.5	3.9	0.4	0.8	4.0	0.6	0.5	0.05	0	10.25
2	fen	6.9	2.7	0.5	1.0	3.2	0.4	0.4	0.08	0	8.28
3		6.2	1.0	0.5	0.7	1.2	0.4	0.5	0.02	0	4.32
4		5.6	0.4	0.5	0.5	0.7	0.2	0.5	0.04	0.01	2.85
5	transition	4.8	0.1	0.3	0.5	0.3	0.1	0.3	0.07	0.03	1.70
6		4.1	0	0.4	0.4	0.2	0.1	0.3	0.04	0.14	1.58
7	bog	3.8	0	0.3	0.3	0.1	0.1	0.2	0.04	0.16	1.20
Sjörs types[b]											
Extreme rich fen		7.7	2.3	0.2	0.4	1.8	0.9	0.2	0.02	-	5.9
Transitional fen		5.8	0.9	0.1	0.03	0.9	0.02	0.05	0.01	-	1.9
Intermediate fen		4.8	0.6	0.01	0.06	0.6	0.03	0.08	0.01	0.02	1.4
Transitional poor fen		5.5	0.1	0.04	0.04	0.1	0.03	0.06	-	-	0.38
Intermediate poor fen		4.4	0	0.03	0.05	0.06	0.03	0.08	-	0.4	0.29
Extreme poor fen		3.9	0	0.06	0.07	0.07	0.02	0.05	-	0.13	0.40
Moss		3.8	0	0.04	0.13	0.04	0.05	0.09	0.01	0.16	0.50

- denotes less than 0.01 milli equivalents per litre.

[a] From Moore and Bellamy 1974.

[b] From Sjörs 1950.

Ombrotrophic bogs are very acid and ion-poor, being fed by precipitation. Minerotrophic fens are nourished by nutrients in groundwater and are therefore ion-rich and only slightly acid to slightly basic (Sjors 1961; Gorham 1967; Boelter and Verry 1977). The water quality characteristic differentiating bogs and fens in the United States include pH, specific conductivity, and calcium concentration (Table 17). These values are for waters leaving peatlands rather than for water within peatlands as were used in the classification systems in Table 16.

Table 17. Characteristic water quality indicators of streamflow from bogs and fens in Minnesota (Boelter and Verry 1977).

Parameter	Bog	Fen
pH	3-4	4-8
Specific conductivity (mmhos/cm)	<80	>100
Calcium (mg/l)	4-5	>15

C. WATER QUALITY OF NATURAL PEATLANDS

Water quality studies of natural peatlands have been of two major types; those that investigate pools and depressions within the peatland and those that examine the runoff waters leaving peatlands. Table 18 summarizes the studies relating to the water quality of peatland pools by country and peatland type. Typically, the purpose of such studies is to investigate the ecological nature of the peatland by relating water chemistry to vegetation, peat type, or peatland type. The influence of groundwater on fen water chemistry is evident with fens being much more ion-rich than bogs. Also,

Table 18. Summary of the water quality of natural peatland pools.

Parameter	Finland[a]		Czechloslovakia, Poland, and USSR[b]			England[c]		
	Bog	Fen	Bog	Transition	Fen	Bog	Transition	Fen
pH	3.75	4.27	2.8-4.2	3.5-5.4	6.6-8.5	3.7-4.4	4.1-6.9	6.1-7.6
Conductivity μmhos/cm^3	23.4	69.5				54-89kc	45-79kc	43-119kc
Total Hard ^{o}dH			0.2-2.0	2.0-4.0	2.0-8.0			
Carbonate Hard ^{o}dH			0	0.1-1.2	1.5-7.0			
Carbonic Acid			20-70	20-70	4-30			
HCO_3 mg/l			0	0-20	20-90	0	0-14.9	6.6-71.4
K mg/l	.59	1.38				.2-2.0	.12-1.52	.17-1.75
Ca mg/l	2.50	4.91	0.6-4.5	2.5-30.0	20-60	1.3-1.8	2.1-7.3	2.3-17.5
Mg mg/l	.82	2.00						
AL mg/l	.72	.80						
Fe mg/l	.26	1.22	0.3-5.0	0.1-10.0	2.0-3.5			
Na mg/l	.52	1.43				4.5-9.6	3.5-6.3	3.6-6.0
CL mg/l	2.31	4.50						
MN mg/l	.047	.051						
Zn mg/l	1.86	2.69						
B mg/l	.044	.053						
Mo mg/l	.074	.072						
SiO_2 mg/l			1.5-40.0	0.1-3.5	0.2-4.5			
SO_4 mg/l			0.5-15.0	8-55	35-75			
Ammonia-N mg/l	3.47	2.00						
NO_3-N mg/l			2.0-14.0	0.5-15.0	3.5-18.0			
Total-P mg/l	.039	.053						
PO_4P mg/l			0.01-0.12	0.02-0.25	0.08-0.35			
Total ions mg/l			<30	35-80	122-220			

[a] Tolonen and Seppanen 1976.

[b] Pietsch 1976.

[c] Gorham 1956.

Table 19. Summary of the water quality of runoff waters from natural peatlands.

Parameter	Minnesota, USA[a] Perched Bog	Minnesota, USA[a] GW Fen	Czechloslovakia[b] Transition	Canada[c] Bog	Canada[c] Transition	Canada[c] Fen	USSR[d] Bog	USSR[d] Fen
pH	3.3-3.9	6.2-6.8	3.5	4.2-5.9	4.3-7.4	6.7-7.2	3.5-5.9	5.3-7.1
Conductivity μmhos/cm	38-64	77-173	202 ms	27-32	22-305	328-370		
Color PL/Co	183-653	36-164						
O_2 mg/l			5.1-7.8	.7-6.3	.7-3.9			
Suspended sediment mg/l							8-20	15-35
Humic & Fulvic acid mg/l							85-140	16-45
Acidity mg/l	24.2-72.2	-	2.0 mual/l					
Alkalinity mg/l	-	26.2-82.2	0					
HCO_3 mg/l							1-18	50-420
K mg/l	.7-1.9	.7-1.5	.8	.3-.6	.7-3.9	2.6-3.7		
Ca mg/l	1.4-3.4	7.6-25.6	2.7	.8-1.0	1.6-41.0	25.8-43.0	1.2-8.0	12-100
Mg mg/l	.61-1.33	1.95-3.81		.3-.6	.3-14.3	12.7-20.0	.1-2.8	.5-10.0
AL mg/l	.36-1.22	.10-.22						
Fe mg/l	.55-2.15	.50-1.46	4.9					
Na mg/l	.3-.9	1.0-3.0	1.1	.2-1.7	.4-2.8	2.0-2.1		
CL mg/l	0-1.5	0-.8	5.2	1.4-2.0	1.6-6.8	3.6-6.7	2-9	1-6
MN mg/l	.01-.11	.02-.14	.10					
Cu mg/l	0-.11	.18-.54						
Pb mg/l	<.05	<.05						
Zn mg/l	0-.19	0-.28						
F mg/l				0	0-1.3	.3-1.4		
SiO_2 mg/l	.6-4.8	.9-8.9	11.0	0-2	0-6	6-8		
SO_4 mg/l	2.4-6.8	1.8-10.2	19.8				2-8	3-16
Total-N mg/l	.7-1.98	.29-.87						
Organic-N mg/l	.29-1.09	.11-.55						
Ammonia-N mg/l	.06-.84	.01-.29	.4					
NO_3-N mg/l	0-.45	.03-.17	0	1.2-1.6	1.9-7.9	6.2-9.6	.1-1.0	0-1
NO_2-N mg/l	0-.006	0-.006	0				.1	.1
Total-P mg/l	.01-.37	.05-.13						
PO_4-P mg/l			.18					

[a] Verry 1975.

[b] Ferda and Novak 1976

[c] Walmsley and Lavkulich 1975

[d] Largin 1976

fen waters appear to be richer in the nutrients: nitrogen, phosphorus, and potassium. Heinselman (1970) studied the water quality of peatland pools in Minnesota, USA, and found similar results with bogs having a pH of 3.3 to 3.6, calcium of 1.8 to 2.6 mg/l, and magnesium of 0 to 0.3 mg/l. Fens again were richer with a pH of 5.2 to 6.0, calcium of 2.6 to 6.4 mg/l, and magnesium of 0.4 to 4.7 mg/l.

The water quality of runoff waters from natural peatlands is summarized in Table 19 by country and peatland type. Table 20 summarizes the water quality characteristics that would be greater for fens and greater for bogs based upon these studies. The remaining characteristics have concentrations that are similar for bogs and fens.

Table 20. List of water quality characteristics that have greater concentrations in fens or bogs.

Fen > Bog	Bog > Fen
pH	Color
Conductivity	O_2
Suspended sediment	Humic and fulvic acid
Alkalinity	Acidity
HCO_3	Fe
Ca	Na
Mg	CL
Cu	SiO_2
Zn	Total-N
SO_4	Organic-N
NO_3-N	Ammonia-N
PO_4-P	Total-P

Seasonal differences in concentrations have been noted by several investigators. In peatland pools, Gorham (1956) observed that concentrations of many constituents increased in dry weather due to evaporation. In runoff waters from peatlands, concentrations of many constituents appear flow related. For example, during periods of high flow in spring and summer storms, higher values of pH and O_2 were observed while other constituents had lower values including: conductivity, acidity, K, Ca, MN, Fe, Al, and SO_4 (Sparling 1966; Walmsly and Lavkulich 1975; Verry 1975). Verry (1975) also stressed the importance of considering flow rates when evaluating water quality by showing that annual yields (Kg/ha/yr) of many constituents remained constant among several bogs even though the watershed areas and annual discharges varied substantially. Verry (1975) also showed by weighting concentrations by the amount of annual streamflow that organically derived nutrients had similiar values for perched bogs and groundwater fens but that minerals derived from groundwater solution were higher in the fen.

D. CHEMICAL CONTENT OF PEAT

The quality of waters leaving peatlands is influenced not only by the quality of input waters but also by the chemistry of the peat itself. There have been numerous studies of the chemical content of peat that have focused on essential plant nutrients, heavy metals, and trace metals. Also, the availability (solubility) of many nutrients has been explored. Walmsley and Lavkalich (1975) have reported the chemical content of several Canadian peatlands (Table 21).

Table 21. Nutrient content of Canadian raised bog and fen peatlands (Walmsley and Lavkulich 1975).

		Raised Bog (Fibric)	Fen (Hemic)
Ash	%[a]	1.1	12.1 - 18.9
Total N	%	.7	2.2 - 2.6
Ca	%	.17	.90 - 1.88
Mg	%	.0097	.12 - .14
Na	%	0	.021 - .044
K	%	.0039	.012 - .020
C/N Ratio		65	11 - 20
pH ($CaCl_2$)		3.8	6.1 - 6.4

[a] ppm = percent x 10^4.

As expected, fen peats are richer in nutrients. The order of abundance for exchangeable cations is: Ca > Mg > Na > K. Another important nutrient, phosphorus, is quite low in peat and Brown and Farnham (1976) have reported C:P ratios of 300 to 500 for Minnesota peat.

Of increasing interest is the heavy metal content of peatlands due to the great absorptive capacity of peat. Pakarinen and Tolonen (1976) reported the content of seven heavy metals in sphagnum for several countries (Table 22).

Table 22. Heavy metal content of sphagnum from several countries (Parkarinen and Tolonen 1976).

Heavy Metal	Finland	Sweden	Germany	Canada
Fe ppm[a] dry wt.	230-319	360-1150	350-780	82-546
MN ppm dry wt.	99-169	70-114	49-65	51-77
Zn ppm dry wt.	34-47	190-218	120-184	79-180
Cu ppm dry wt.	4.5-6.3	10.6-14.3	0.3-9.4	2.8-3.2
Pb ppm dry wt.	5.6-19	56-78	81	.7-7.3
Cr ppm dry wt.	1.6-5.0	2.8-3.7	2.5-2.8	.5-2.4
Ni ppm dry wt.	1.2-3.4	2.1-2.3	1.7-1.9	.2-.8

[a] % dry wt. = ppm x 10^{-4}

Lead (Pb) shows a regular gradient from low levels in sparsely populated Canada to highly industrialized Germany. Levels of lead in peat from Britain have been as high as 500 ppm (Parkarinen and Tolonen 1976). South Finland was substantially higher in lead, iron, zinc, and nickel than north Finland. Sapek (1976) found concentrations of lead in Poland peatlands to range from 10 to 60 ppm and cadmium from 0.15 to 1.6 ppm, and observed that higher concentrations occurred in the upper peat layers. Bysiek et al. (1972) also found mercury in Finish peats ranging from 0.012 to 0.39 ppm while coal ranged from 0.07 to 33.0 ppm and petroleum from 1.9 to 21.0 ppm. Uranium and arsenic have also been found in Finish peats which may relate to geologic mineralization in the vicinity (Yliruokanen 1976).

Peat may also contain small amounts of trace metals. Sillanpää (1972) analyzed the chemical content of a raised bog profile in Finland for some heavy metals along with several trace metals (Table 23).

Table 23. Heavy metal and trace metal concentrations of a raised bog peat profile in Finland (Sillanpää 1972).

Metal	Concentration in descending order
Al	.05-9%
Fe	.05-5%
Mn	10-700 ppm
Sr	5-300 ppm
V	1-200 ppm
Cr	1-150 ppm
Zn	10-100 ppm
Cu	1-90 ppm
Ni	2-50 ppm
Pb	2-40 ppm
Co	.5-30
Mo	.25-5
Sn	.2-2.5

The high levels of aluminum and iron coincided with a high ash content of 90 percent at the base of the profile. Generally, maximum concentrations occurred at the bottom of the profile and the surface had concentrations that were slightly greater than the middle portions of the profile. Metal concentration appeared directly proportional to ash content. Peat, being an active ion-exchange material has a selective affinity for various cations. The general order of affinity for divalent cations is: Cu > Co > An > Ni > Ba > Sr > Ca > Mg and for monovalent cations is: Ag > Ti > Cs > K > Na > Li with the affinity being greater for the divalent cations than the monovalent.

Herbicides may also be found in peatlands. Pheeney and Radforth (1972) found traces of DDT and DDD, its anaerobic breakdown state, in bogs in N.B., Canada, even though one of the bogs had never been sprayed. Traces were found at depths as great as 170 cm. Even though DDD is a dechlorination of DDT, it has a much greater toxicity and stability than its parent.

The availability (solubility) of plant nutrients contained within peat vary with pH and Eh. Figure 25 shows the availability of 12 plant nutrients as a function of pH with wider bands indicating greater availability. Sphagnum fibric peats, commonly found in the surface horizon of bogs have pH's ranging from 3.0 to 4.5 while hemic peats, commonly found in the surface horizon of fens or lower horizons of bogs have pH's greater than 4.5 and generally greater than 5.0 (Farnham and Finney 1965; Farnham 1968). Since many of these plant nutrients are also important water quality constituents the availability of each will be discussed.

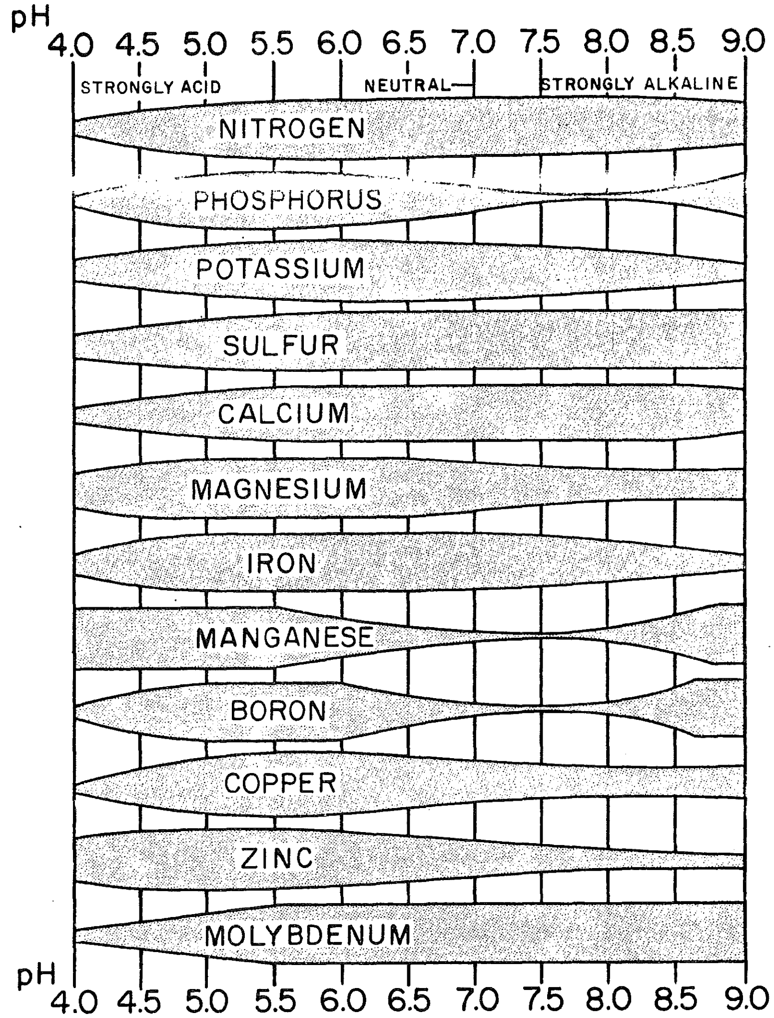

FIGURE 25: THE AVAILABILITY OF VARIOUS PLANT NUTRIENTS IN ORGANIC SOILS AS A FUNCTION OF pH (FROM LUCAS AND DAVIES, 1961).

1. Nitrogen

Nitrogen availability in peat is influenced by the total nitrogen content of the soil and the pH. Organic soils with a pH of less than 4 generally have nitrogen contents less than 1 percent while peats with pH's greater than 5 contain more than 2 percent nitrogen.

Below a pH of 5 microbial activity is reduced with a marked reduction in nitrate-nitrogen release (Lucas and Davies 1961).

2. Phosphorus

Phosphorus is most available around a pH of 5.5 with availability dropping below pH 5.5, frequently due to the greater solubility of iron and alumimum at these lower pH's which combine with phosphorus. Very acid peats contain as little as 0.01 percent phosphorus while peats that receive abundant calcium have about 0.05 percent phosphorus.

3. Potassium

Potassium availability is less affected by pH than phosphorus availability and is more a function of total potassium content in the peat. Very acid peats usually have less than 0.1 percent potassium while peats supplied by abundant calcium average 0.15 percent potassium. Very high levels of calcium may reduce potassium availability.

4. Sulfur

In raised bog peats, sulfur is usually less than 0.1 percent while fens may average 0.4 percent. Sulfur content varies in the peat profile with SO_4 generally increasing with depth to just above the permanent water table. Below this layer, under reducing conditions, sulfur is found as sulfide and possibly free sulfur. If this layer becomes aerated, such as by drainage, sulfide can be oxidized to sulfate resulting in a lowering of the pH.

5. Calcium

Calcium availability is less below pH 5.5 and calcium is generally more abundant in fens (Table 22).

6. Magnesium

Magnesium is similiar to calcium in behavior and Lucas and Davies (1961) propose that Ca/Mg ratios in peat profiles are useful in determining the level at which a bog has risen above the influence of groundwater. Fen peats developed under the influence of groundwater have Ca/Mg ratios of about 5:1 while bog peats developing under the influence of precipitation have Ca/Mg ratios of less than 1.

7. Iron

Iron availability drops below pH 5.0 but usually there is sufficient iron in all peats due to humic acids which maintain iron availability. Iron solubility is also dependant on oxygen concentrations with the reduced state (ferrous iron) being more soluble at low oxygen concentrations. Ferric iron is found in solution only below a pH of 3 (Sparling 1966).

8. Manganese

Although fens contain more manganese (0.02 percent) than bogs (0.001 percent), the availability of this nutrient declines sharply when pH rises above 5.5. Manganese solubility is similarly oxygen dependent. The reduced manganous (Mn^{2+}) is soluble while the oxidized state (MnO_2) is insoluble (Hem 1970).

9. Boron

Ca/B ratios are important in determining boron availability in peat soils. Low ratios occur when the pH is less than 5.0 such as in acid raised bogs.

10. Copper

Copper is less available in very acid peat soils and availability drops sharply at pH's below 5.5

11. Zinc

Typically, raised bog peats contain less zinc but has greater zinc availability than more basic peats. For highly basic peats, zinc may form zinc hydroxide and complex zinc compounds which are insoluble causing reduced availability.

12. Molybdenum

Molybdenum availability drops below pH 5.5 and peats that are high in iron content are particularly low in molybdenum availability.

In summary, fens have greater nutrient availability of N, P, K, S, Ca, Mg, Fe, B, Cu, and Mo; while bogs have greater availability of Mn and Zn. Since this means that the solubility of these nutrients is also higher, the potential for these elements leaving the respective peatland in runoff water is greater.

E. SIGNIFICANCE OF WATER QUALITY PARAMETERS

There are numerous constituents of water that have been used to describe the quality of water. For all but the water quality specialist, numerical values assigned to these constituents are almost meaningless. This section is provided to clarify the significance of water quality parameters. These constituents are organized into the categories: major constituents, heavy metals, organic constituents, and related properties.

Major Constituents

1. pH

The pH of a substance is the negative log, base 10, of the hydrogen ion activity in moles per liter. A pH of 7 indicates neutral or pure water; a pH greater than 7 indicates alkalyne water which normally occurs when carbonate and bicarbonate ions are present; and a pH less than 7 implies acid water which often occurs when CO_2 is excessive. Carbon dioxide is one of the most important reactions establishing pH. When carbon dioxide enters water either from the atmosphere or by respiration, carbonic acid is formed which dissociates into bicarbonate and then carbonate and H^+ ions are liberated influencing pH:

$$CO_2 + H_2O \rightleftarrows H_2CO_3 \rightleftarrows H^+ + HCO_3^- \rightleftarrows H^+ + CO_3^{--}$$

The pH at any one time is an indication of the balance of chemical equilibria in the water (Hem 1970).

The pH of water is very important for fish and other aquatic life. There are numerous studies on the pH range prefered by individual fish species and generally the toxic limits for most species are pH's less than 4.8 and greater than 9.2. Most freshwater fish tolerate the pH range of 6.5 to 8.4. A pH of 4 is toxic to the algae *Asplanchna* but *Fragilaria, Asterionella,* and *Aphanizomemon* will tolerate a pH as low as 2.5. Algae are destroyed at pH's greater than 8.5 (McKee and Wolf 1963; McKim et al. 1975).

Jeglum (1971) has used pH to separate Canadian peatlands into fertility classes:

Fertility Class	pH Range
Very oligotrophic	3.0 - 3.9
Oligothrophic	4.0 - 4.9
Mesotrophic	5.0 - 5.9
Eutrophic	6.0 - 6.9
Very eutrophic	7.0 - above

As expected, the oligotrophic bogs are quite acid. Clymo (1964) has offered several explanations for the very low pH of sphagnum bogs. First, H^+ ions in rainfall, in areas of industrial pollution, may supply a significant part of the total hydrogen ion accumulation in a bog. Second, sulfur-metabolizing bacteria may produce $SO_4^=$ and H^+ increases during dry periods. Third, sphagnum plants secrete organic acids but these probably do not account for much of the total hydrogen ion activity. Fourth, sphagnum plants may produce H^+ ions metabolically and these hydrogen ions are readily replaced by other cations. Clymo felt that cation exchange in sphagnum was the greatest cause of low pH's in bogs. Gorham (1956, 1957) stated that another source of H^+ ions could be the dissociation of sulfuric acid in water.

2. Acidity

Acidity is the capacity of water to neutralize hydroxyl ions. Linked to pH, acidity is caused by a presence of free carbon dioxide, carbonic acid, free H^+ ions, and by some acids like sulfuric acid, bisulfate (HSO_4) and phosphoric acid. Acidity is determined by adding a base until pH 8.3 is obtained (Hem 1970). McKee and Wolf (1963) also report that humic acids may attribute to acidity.

The effect of acidity alone on fish has not been investigated but waters receiving acid mine drainage, having a pH of 4.0 and a total acidity of 100 mg/l, could not sustain fish life (McKee and Wolf 1963).

3. Alkalinity

Alkalinity, the opposite of acidity, is the capacity of water to neutralize hydrogen ions (acid). Alkalinity is also linked to pH and is caused by the presence of carbonate, bicarbonate, hydroxide, and weak acids like carbonic acid which is formed when carbon dioxide is dissolved. Organic acids may also contribute to alkalinity. The ratio of any of these carbon dioxide species found in water at any one time is determined by pH (Figure 26). For example, at a pH of 7, there would be about 18 percent of the total dissolved species of CO_2 as carbonic acid (H_2CO_3) and 82 percent as bicarbonate (HCO_3-). A high alkalinity is generally associated with high pH, hardness, and excessive dissolved solids. High buffering is also associated with high alkalinity (McKee and Wolf 1963; Hem 1970).

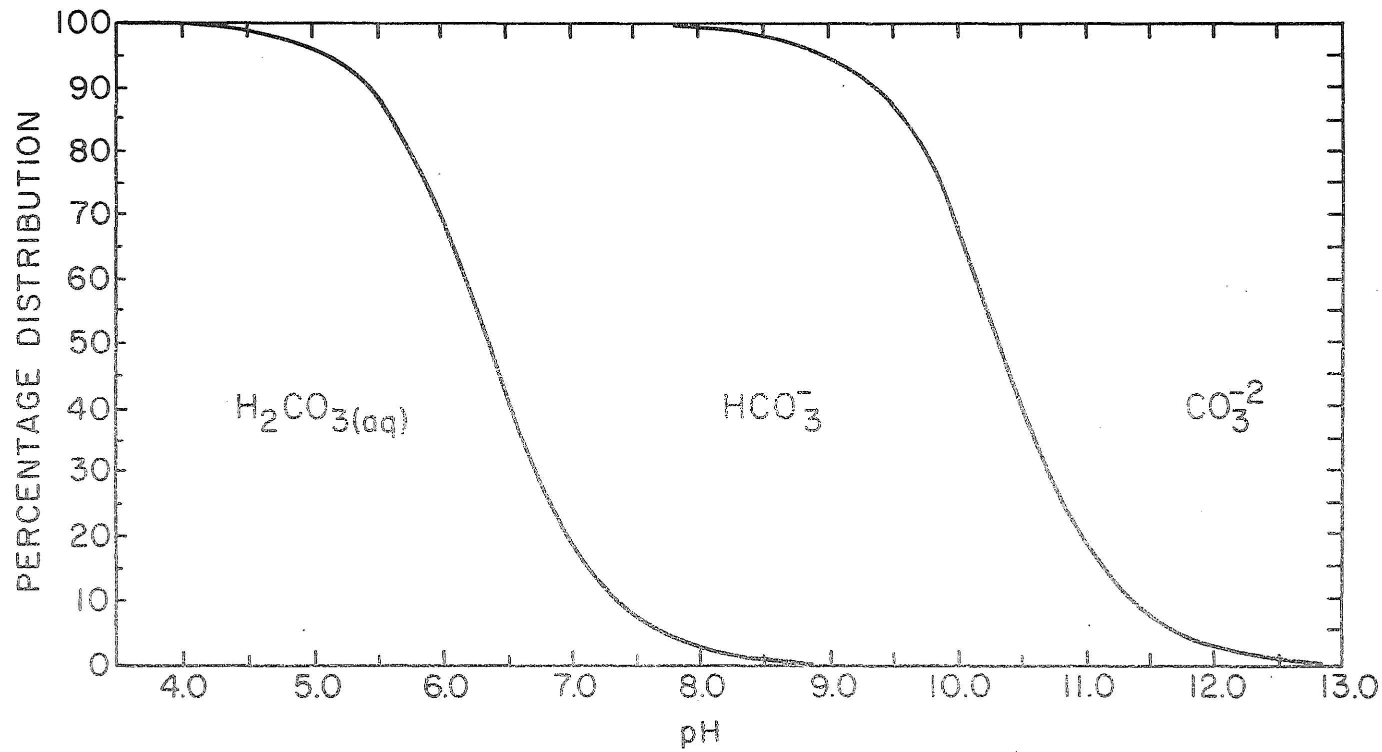

FIGURE 26: PERCENTAGES OF TOTAL DISSOLVED CARBON DIOXIDE SPECIES IN SOLUTION AS A FUNCTION OF pH (FROM HEM, 1970).

Most streams have alkalinities less than 200 mg/l while some groundwater may exceed 1000 mg/l when calcium and magnesium concentrations are high. The range of alkalinity in peatlands appears to be 26 to 82 mg/l for fens and almost zero for bogs (Table 20).

High alkalinity is undersirable for many industrial process waters including: beer, carbonated beverages, and pulp and paper making. Alkalinity is generally not lethal to adult fish when in insufficient concentrations to raise the pH above 9.0. A high alkalinity actually reduces the toxicity of copper sulfate to fish and serves as a buffer to prevent sudden changes in pH. The best waters for supporting a well diversified aquatic life have a pH range of 7 to 8 and an alkalinity range of 100 to 120 mg/l (McKee and Wolf 1963; Hem 1970).

4. Specific Conductivity

Conductivity is the ability of a substance to conduct electrical current while specific conductivity is the conductance of a cube of water one centimeter on a side, expressed as μmhos/cm. Conductivity increases with greater dissolved solids and with temperature, therefore, all values are corrected to $25^{\circ}C$. The relationship between total dissolved solids and specific conductivity has been expressed empirically by Hem (1970) as:

$$TDS\ (ppm) = A \times Spec.\ Cond.\ (\mu mhos/cm)$$

where A = 0.55 to 0.75. Conductivities of peatland waters have often been corrected and reduced to compensate for the high hydrogen ion content. Such values appear as Kcorr in the literature (Sjörs 1950; Gorham and Pearsall 1956).

Specific conductivities greater than 2000 μmhos/cm are not considered good for a well mixed fish fauna and values greater than 4000 μmhos/cm exceed the upper limit tolerated by fish. Generally, peatlands have low conductivities ranging from 20 to 370 μmhos/cm with fens being greater than bogs (Table 20). Verry (1975) has used specific conductivity to differentiate between peatland types with fens having conductivities greater than 80 μmhos/cm and perched bogs having less than 80 μmhos/cm. In comparison, oligotrophic lakes have conductivities less than 200 μmhos/cm and eutrophic lakes more than 200.

5. Nitrogen

The sources of nitrogen include: a small amount from weathering of rocks, the atmosphere, soils, and biological materials. Nitrogen is found in several organic forms: ammonia, organic, and nitrite-nitrogen and also as inorganic nitrate (Figure 27). Nitrogen gas (N_2) in the atmosphere is converted to organic nitrogen by nitrogen

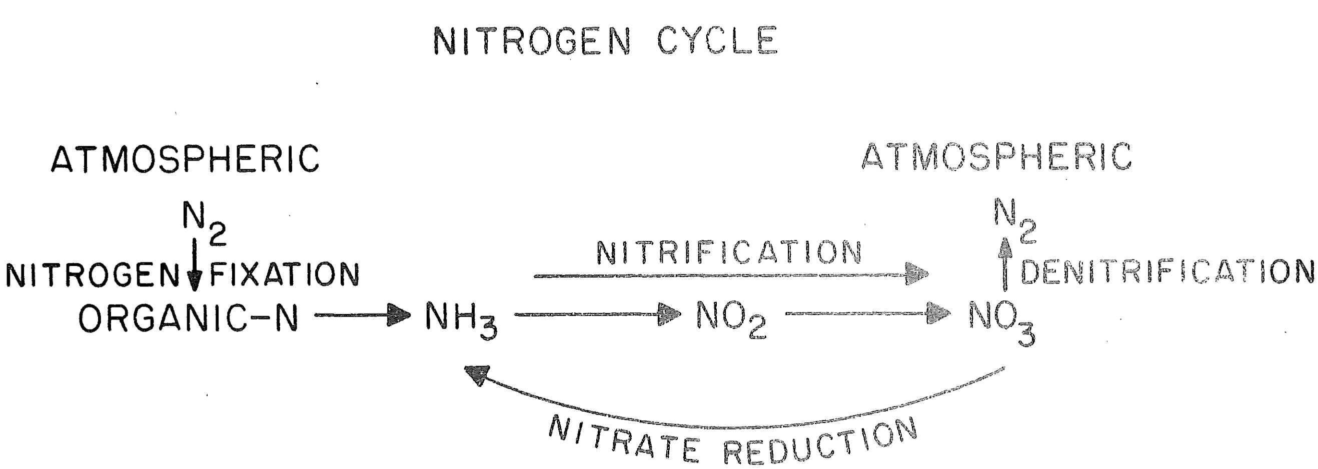

FIGURE 27: GENERALIZED NITROGEN CYCLE.

fixing plants and bacteria. Upon decay, ammonia (NH_3) is formed which is ultimately changed to nitrate (NO_3) by aerobic decomposition. In the absense of oxygen, nitrate can be converted back to ammonia by nitrate reduction or to N_2 gas by denitrification (Hem 1970). Since peatlands are usually anaerobic being saturated or nearly so, most of the nitrogen should be present in the more reduced forms; organic and ammonia. Procedures exist for determining all forms of nitrogen except organic which is derived by substracting ammomia-N from total-N (AWWA 1971).

Total nitrogen in runoff waters from peatlands varies from about 0.3 to 2.0 mg/l and nitrate concentrations vary up to 0.45 mg/l or 25 percent of total. By comparison, peatland waters appear to contain a smaller percentage of total nitrogen as nitrate than do streams leaving watersheds under various land uses for the eastern United States including Minnesota (Table 24).

Table 24. Mean total and nitrate nitrogen concentrations in streams vs. land use in the eastern U.S. (Omernik 1976).

Land Use	Total Nitrogen mg/l	Nitrate mg/l	Nitrate as % of Total-N
Forest	0.85	0.23	27
Mostly forest	0.88	0.35	39
Mixed	1.28	0.68	53
Mostly urban	1.29	1.25	98
Mostly agriculture	1.81	1.05	57
Agriculture	4.17	3.19	76

High nitrate nitrogen simulates growth by algae and aquatic plants with about 0.30 mg/l of nitrogen being needed for algal blooms depending upon phosphorus concentrations. Water that is suitable for fish life generally has less than 4.2 mg/l nitrate-nitrogen. The 1962 USPHS drinking water standards limit nitrate to 45 mg/l (McKee and Wolf 1963).

6. Phosphorus

The sources of phosphorus include: igneous rocks, soil leaching, organic wastes, and decomposition of organic matter. The phosphorus cycle is poorly understood and the common forms are essentially defined by the analytical technique used to obtain them (Figure 28). Generally, H_2PO_4 becomes available by weathering, which can then be immobilized by plants and converted to an organic form. Upon decay, the organic P is mineralized to H_2PO_4.

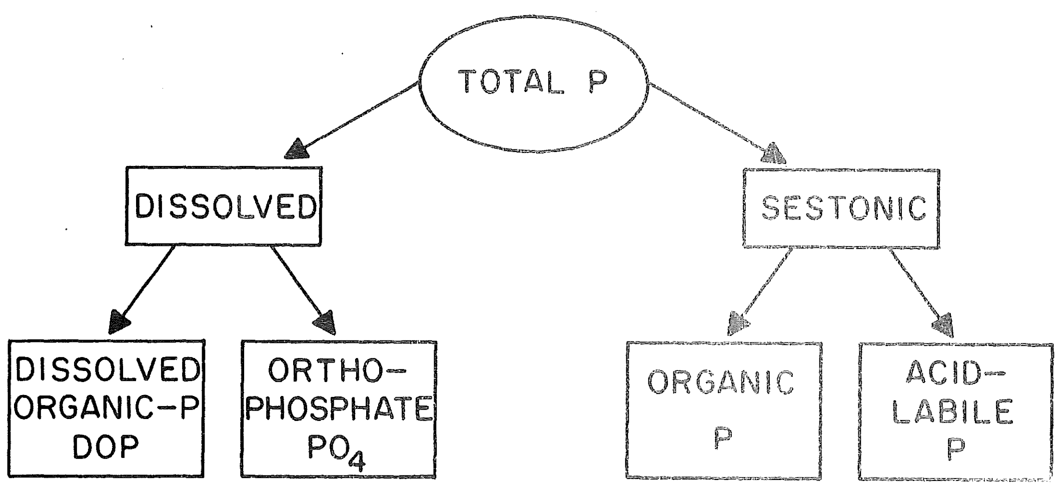

FIGURE 28: THE FORMS OF PHOSPHORUS IN AN AQUATIC ENVIRONMENT.

The various forms of phosphorus are also a function of pH (Figure 29). Between pH 3 and 5 most of the phosphorus is found as H_2PO_4. Ortho-phosphate (PO_4) is actually the final dissociation of

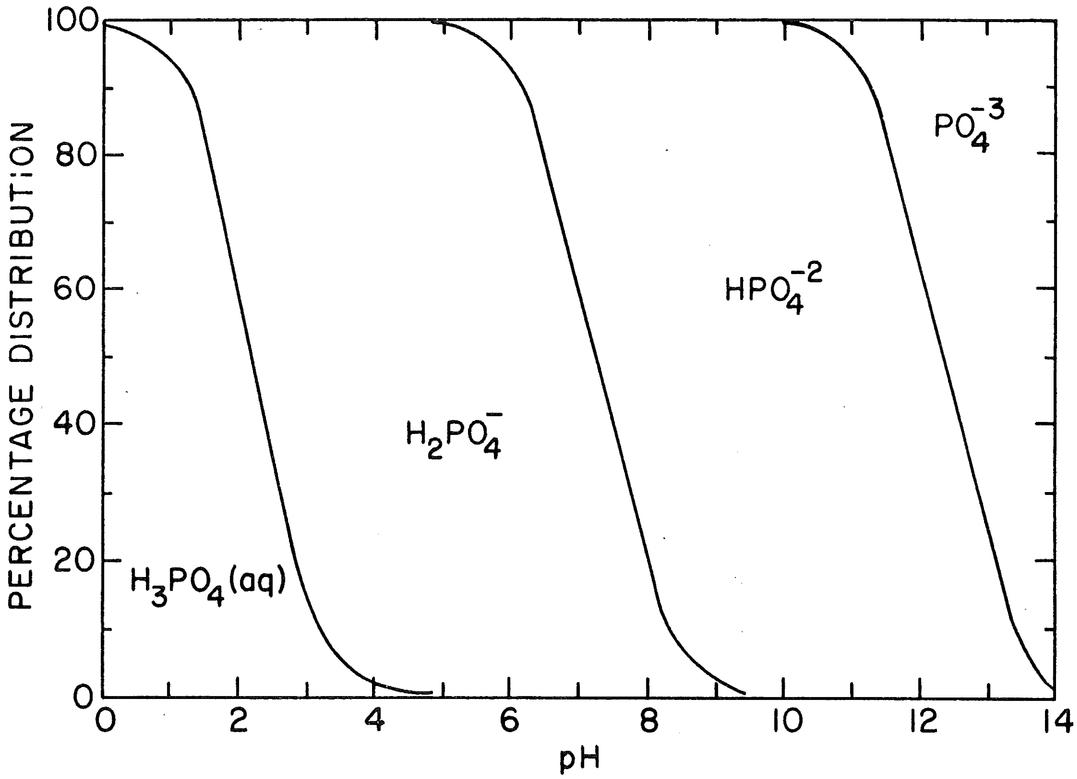

FIGURE 29: PERCENTAGES OF TOTAL-DISSOLVED PHOPHATE SPECIES IN SOLUTION AS A FUNCTION OF pH (FROM HEM, 1970).

phosphoric acid (H_3PO_4). The adsorption of phosphate by metal oxides, especially iron oxide, prevent high concentrations of phosphorus (Hem 1970).

Total phosphorus concentrations in peatland runoff waters vary from 0.01 to 0.40 mg/l and ortho-phosphate varies from 0.01 to 0.35 mg/l. These values are somewhat higher than the average total and ortho-P values for streams vs. land use in the eastern United States (Table 25). Ortho-phosphate remained a relatively constant percentage of total-P ranging from 40 to 43 percent.

Table 25. Mean total and Ortho-P concentrations for streams vs. land use in the eastern U.S. (Omernik 1976).

Land Use	Total-P (mg/l)	Ortho-P (mg/l)
Forest	.014	.006
Mostly forest	.035	.014
Mixed	.040	.017
Mostly urban	.066	.033
Mostly agriculture	.066	.027
Agriculture	.135	.058

Excessive amounts of phosphorus lead to abundant growths of algae and are detremental to fish. Phosphate is rarely toxic, however, and is sometimes beneficial to fish life by increasing food supplies. Generally, excessive algal growth may occur when phosphorus concentrations exceed 0.05 mg/l (McKee and Wolf 1963).

7. Calcium

Calcium is a major constituent of many rock types, especially limestone, and the weathering of these rocks produce soluble calcium. Leaching of mineral soils and glacial deposits also produce soluble calcium in surface and groundwaters. Calcium remains soluble in water as long as CO_2 is present, i.e., at pH's less than 7 to 8. At more alkaline pH's calcium will precipitate as $CaCO_3$. Calcium is one of the major ions contributing to hardness, total dissolved solids, and specific conductivity (Hem 1970).

High calcium concentrations do not appear to be harmful to fish and other aquatic life and actually calcium is known to reduce the toxicity of lead, zinc, and aluminum to fish (McKee and Wolf 1963).

8. Magnesium

Magnesium is the seventh most abundant mineral in igneous rocks and is also abundant in the carbonate rocks; dolomite and limestone. Magnesium is present in water in ionic form and like calcium, its solubility increases with greater CO_2 concentrations or lower pH (Hem 1970).

Magnesium can be toxic to fish if found in concentrations greater than 100-400 mg/l and waters supporting a good fish fauna generally contain less than 14 mg/l magnesium (McKee and Wolf 1963). Most peatlands contain less than 20 mg/l magnesium with fens having higher concentrations than bogs.

9. Sodium

Sodium is abundant in both igneous and sedimentary rocks as well as evaporite sediments and ocean waters. When sodium is leached or otherwise reaches surface and groundwaters it remains in solution since there are no important precipitation reactions (Hem 1970). Sodium in peatland waters has been found to be related to atmospheric sources from sea spray in England and some coastal peatland waters may have sodium concentrations as high as 63 mg/l (Gorham 1956; Gorham and Cragg 1960). Sodium concentrations also appear to be slightly less for the more acid peatlands (Gorham 1955).

Generally, waters supporting good fish fauna have concentrations of sodium plus potassium less than 85 mg/l. Sodium may also have the beneficial effect of reducing the toxicity of aluminum and potassium salts to fish (McKee and Wolf 1963).

10. Potassium

The sources of potassium include igneous rocks, clays in soils, and glacial materials. Usually potassium is less abundant in waters than sodium but this element is essential for plant growth and is cycled by aquatic vegetation, being released to waters upon plant decay (Hem 1970).

Oligotrophic waters generally contain from 0.4 to 1.5 mg/l potassium while eutrophic waters have up to 5.6 mg/l. Peatland waters usually contain less than 4 mg/l of potassium with fens having higher concentrations than bogs. High potassium concentrations are toxic to fish at levels greater than 400 mg/l and to some invertibrates at 700 mg/l. Since potassium is also an essential plant nutrient, plankton growth is simulated by K additions (McKee and Wolf 1963).

11. Manganese

Manganese is found in igneous rocks and is also leached from the soil. Manganese is essential to plant metabolism and is organically circulated becoming soluble upon decay. At pH levels of seven and below, the most predominant form of manganese is MN^{2+}. At higher pH's under oxidizing conditions, MNO_2 may be found which is insoluble and precipitates (Hem 1970). Manganese is seldom found in waters at concentrations greater than 1 mg/l which is not harmful to fish or other aquatic life (McKee and Wolf 1963). The 1962 USPHS drinking water standard for manganese is 0.05 mg/l (U.S. HEW 1969).

12. Iron

The major sources of iron include: igneous and carbonate rocks, sandstones, shales, and many soils. Both ferrous iron (Fe^{2+}) and ferric iron (Fe^{3+}) are found in fresh waters. The behavior of iron in water is related to pH and Eh with ferrous iron (Fe^{2+}) being found soluble under reducing conditions at all acid pH values up to 6 (Figure 30). Ferrous iron is oxidized to ferric iron (Fe^{3+}) in natural waters and is in solution only below a pH of 3 while at greater pH's, Fe^{3+} forms insoluble hydroxides which precipitate. Similiarly, when Fe^{3+} goes into solution, the pH is lowered (Hem 1970; Sparling 1966).

Usually iron is present in concentrations less than 0.10 mg/l but levels as high as 10 mg/l are common (Hem 1970). Peatland waters usually contain less than 5 mg/l (Table 20). Iron is usually not a problem to fish and other aquatic life because it precipitates; however, excessive iron may lower the pH to a toxic level. A concentration of 50 mg/l iron is the upper limit for fish life and waters that have good fish fauna generally contain less than 0.7 mg/l iron.

13. Aluminum

Aluminum is the third most abundant element in the earth's crust and occurs in many rocks, ores and clay. The alumimum ion is insoluble but many of its salts are readily soluble. Below a pH of 4 the cation AL^{3+} predominates and minimum solubility occurs near pH 6 while higher solubilities occur at more acidic or alkaline pH's (Hem 1970). Alumimum concentrations are generaly low in waters

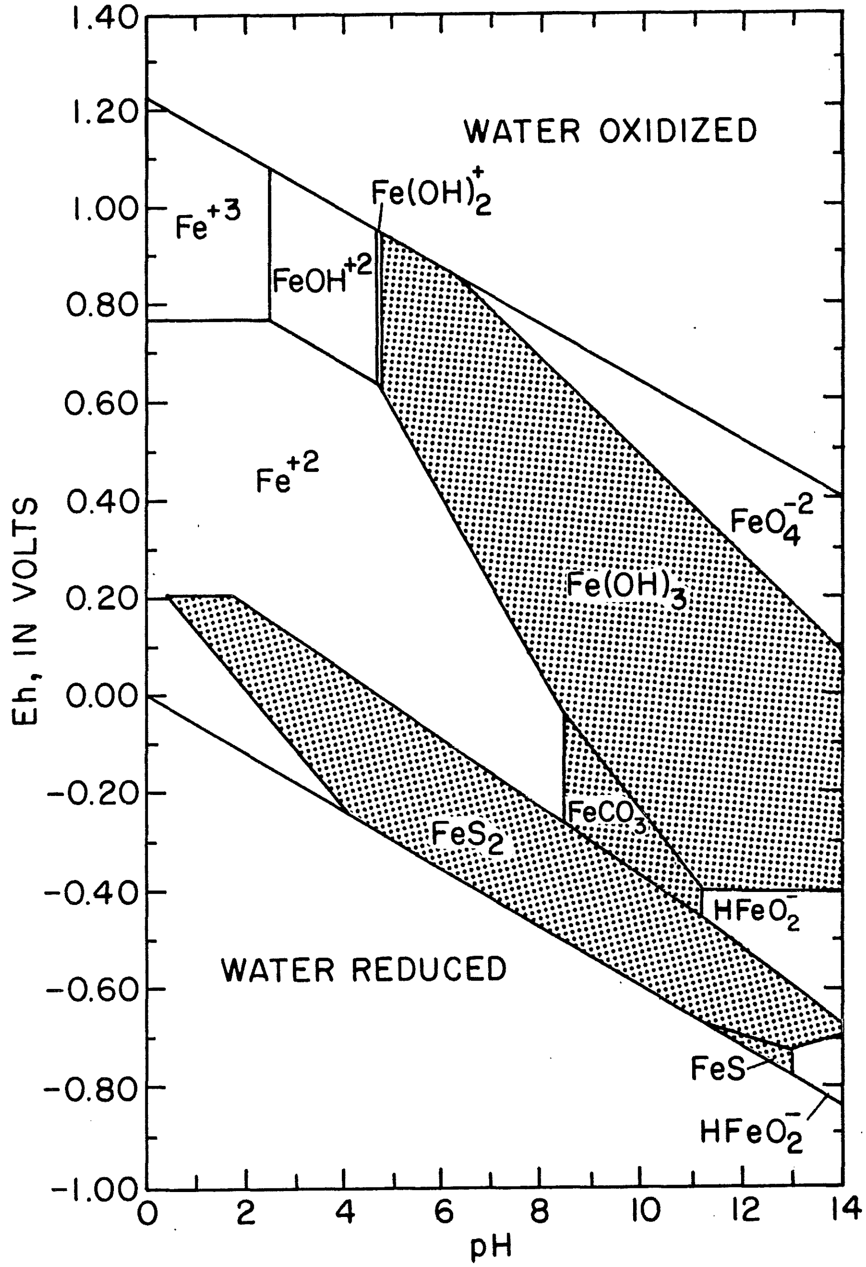

FIGURE 30: FIELDS OF STABILITY FOR SOLID AND DISSOLVED FORMS OF IRON AS A FUNCTION OF Eh AND pH (FROM HEM, 1970).

because it readily precipitates and settles out. Peatland waters generally contain less than 1.5 mg/l aluminum. A concentration of 5 mg/l has been found lethal to trout while aluminum salts have been found to be more toxic with 0.5 mg/l $ALCL_3$ being lethal to fish. Aluminum concentrations are not considered to pose a threat to the public health.

14. Sulfur

Sulfur occurs naturally in water being leached from gypsum and other common igneous and sedimentary rocks. Weathering of these materials with well-aerated water yields oxidized sulfate (SO_4) ions that are carried off in waters. Sulfur, in the form of sulfate, is also found in rainfall at concentrations frequently exceeding 1 mg/l and sometimes greater than 10 mg/l. The various species of sulfur that may be found in water are governed by pH and Eh (Figure 31). The sulfate (SO_4) ion is the most common form in surface waters oxygen is adequate and the pH is greater than 2. At a pH less than 2, HSO_4 is found. Under reducing conditions, the sulfides occur with H_2S being present below pH 7 and HS occurring in alkalyne waters. The sulfide (H_2S) produces the rotten egg smell. Sulfate reduction to sulfide can occur under severe reducing conditions if certain sulfur bacteria and organic matter is present (Hem 1970). Organic sulfur can also be converted to sulfide under reducing conditions (Sparling 1966). A simplified sulfur cycle in water is shown in Figure 32).

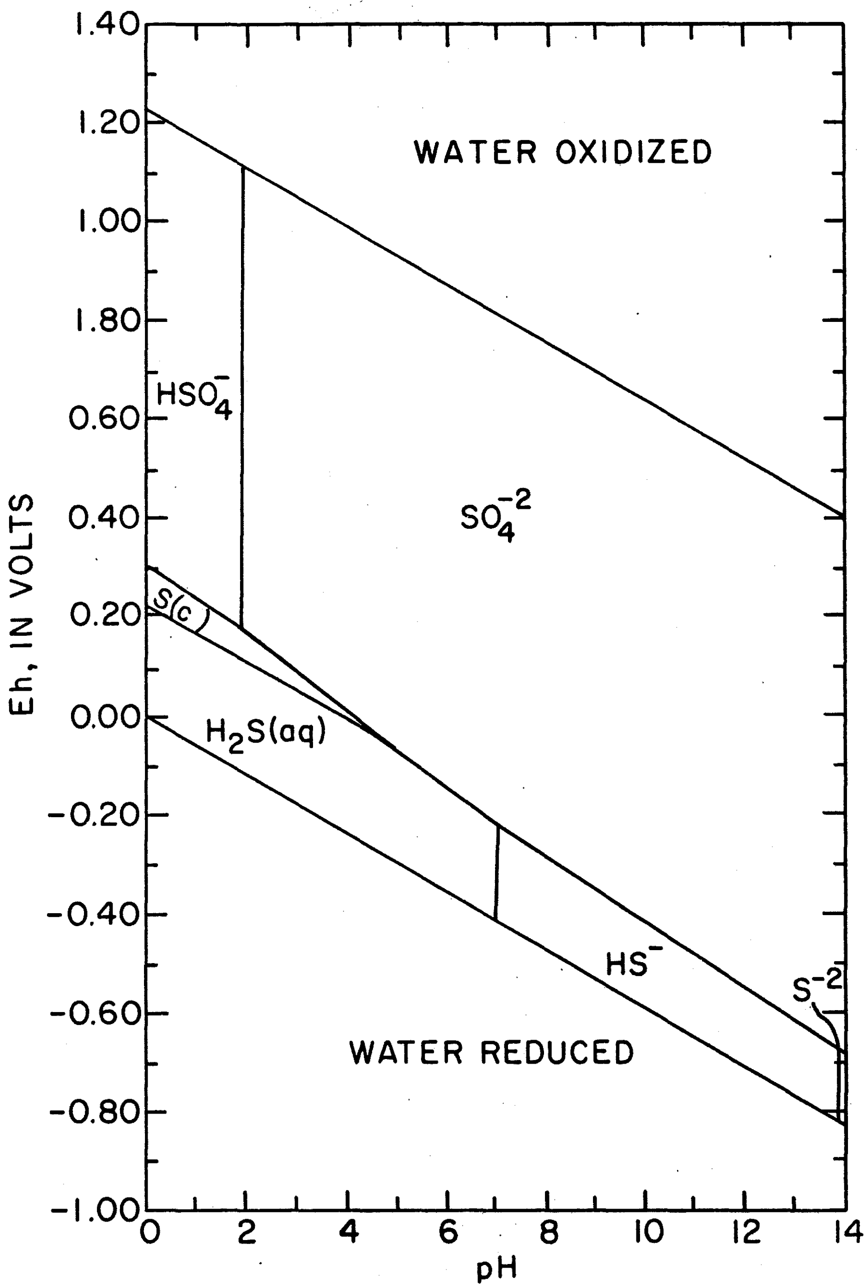

FIGURE 31: FIELDS OF DOMINANCE OF SULFUR SPECIES AS A FUNCTION OF Eh AND pH (FROM HEM, 1970).

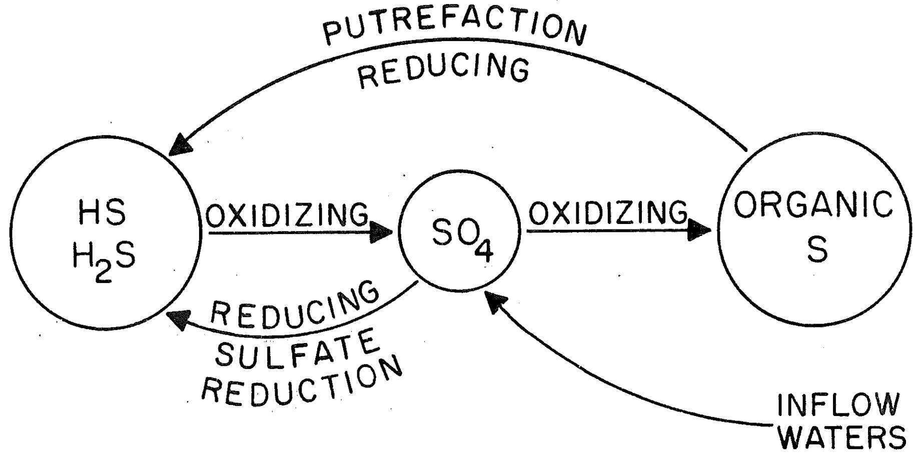

FIGURE 32: SIMPLIFIED SULFUR CYCLE IN AN AQUATIC ENVIRONMENT.

Most peatland waters contain less than 20 mg/l sulfate. Gorham (1956) observed high sulfate concentrations in bog pools in England together with very low pH's. He felt that the anaerobic breakdown of organic sulfur to hydrogen sulfide in the peat with the subsequent oxidation to sulfuric acid in the pools caused the high sulfur and low pH.

The 1962 USPHS drinking water standard for sulfate is 250 mg/l. This standard is not based on taste or physiological reasons but more that high sulfur concentrations act as a laxative. Generally, waters with a desirable fish fauna contain less than 90 mg/l sulfate and waters that have less than 0.5 mg/l sulfate will not support algal growths (McKee and Wolf 1963).

Heavy Metals

1. Copper

Copper is a widely distributed element occurring naturally and an essential plant nutrient that is circulated by biota (Hem 1970). At pH's greater than 7, copper will quickly precipitate as a hydroxide and copper solubility is lower in reducing conditions. Peatland waters in Minnesota have been found to contain from 0 to 0.5 mg/l copper (Table 20). Copper concentrations less than 1 mg/l are not toxic to most fish but concentrations from 0.015 to 3.0 mg/l have been found to be toxic to some fish, crustacea, mollusks, insects, and plankton. Plankton are known to concentrate copper by a factor of 1000 to 5000 times (McKee and Wolf 1963).

2. Nickel

Nickel compounds are found in many minerals and ores. As a pure metal, nickel is insoluble but many nickel salts, such as nitrate and sulfate, are highly soluble. The mean concentration of nickel in North America rivers is 10 mg/l (Hem 1970). At present there is no USPHS drinking water standard for nickel but the USSR has established a limit of 1 mg/l. Nickel is less toxic to fish than copper, zinc or iron but a concentration of 0.8 mg/l was found lethal to the fish stickleback (McKee and Wolf 1963).

3. Zinc

Zinc is common in carbonate and certain igneous rocks and concentrations in water are usually related to mineralized areas (Hem 1970). Zinc salts, such as chloride and sulfate, are highly soluble while carbonate, oxide and sulfide salts are insoluble. Concentrations as high as 50 mg/l have been found in waters leaving zinc mining areas. Minnesota peatland waters have been found to contain less than 0.3 mg/l zinc. The 1962 USPHS drinking water standard for zinc is 5 mg/l based on taste. Zinc is highly toxic to fish especially in soft water where 0.01 to 1 mg/l can be lethal. Copper plus zinc has a synergistic effect on zinc toxicity but only in soft water. Also, lower dissolved oxygen concentrations have been found to increase the toxicity of zinc salts. Since zinc toxicity is greater in soft water, calcium additions have been found to reduce lethal effects (McKee and Wolf 1963).

4. Lead

Lead may be found in limestone or galena and waters draining such areas have been found to contain as much as 0.4 to 0.8 mg/l. Lead usually does not remain long in natural waters but some salts like the choride are soluble while the carbonate and hydroxide salts are insoluble. Lead is a cumulative poison and the 1962 USPHS drinking water standard for lead is 0.05 mg/l. Concentrations as low as 0.1 mg/l are lethal to fish and like zinc, lead is more toxic in soft water or low dissolved oxygen. Lead is also toxic to aerobic bacteria to concentrations of 1 mg/l and bacterial decomposition is inhibited at levels from 0.1 to 0.5 mg/l. Calcium concentrations greater than 50 mg/l have been found to reduce lead toxicity (McKee and Wolf 1963).

5. Mercury

Mercury occurs naturally from igneous and sedimentary rocks and from atmospheric mercury which is evaporated from volcanoes. Cultural sources of mercury include: mining and refining, smelter gasses, fuel combustion, and various industries. Mercury can be found in three forms: 1) elemental mercury, 2) inorganic salts, and 3) organic mercury compounds, including methyl mercury. Mercury ions are found as Hg^{o}, mercurous (Hg_2^{2+}) and mercuric (Hg^{2+}). Of these, Hg^{o} is the most reduced form and is easily volatilized. Hg_2^{2+} is the more reduced of the other basic forms and Hg^{2+} is stable in oxidizing conditions, especially at low pH's. The primary mercuric salt is the sulfide (HgS) which is called cinnebar. In reducing conditions, mercury can join reduced sulfide and precipitate as cinnebar. In oxidizing conditions, mercury tends to be volatile. Mercury has also been found to be more soluble in organic water (Hem 1977).

Methyl mercury is the most toxic of mercuric compounds and can be synthesized from other mercury forms by certain anaerobic microbes making all forms of mercury potentially dangerous. Hem (1977) reports that drinking water should not contain more than 2 μg/l (.002 mg/l) mercury. Methyl mercury is magnified in the food chain by the following amounts: plankton (1,000x), fish (1,000 to 10,000x), and invertibrates (10,000x). Concentrations of from .004 to .020 mg/l mercury have been found harmful to fish but levels as high as 0.2 mg/l were found not harmful to some fish like carp and rainbow trout. Mercuric salts have been found to be lethal to phytoplankton in concentrations from 0.9 to 60 mg/l and toxicity was increased by trace amounts of copper (McKee and Wolf 1963).

Organic Constituents

1. Color

Color in water is caused primarily by organic substances from the decomposition of plant material (McKee and Wolf 1963; AWWA 1967; Lamar 1969). Apparent color includes the effect of suspended matter in the sample. Platinum-cobalt units are the common expression of color where one PL/co unit is one mg/l of platinum in water (McKee and Wolf 1963). Color is highly related to organic acid content and humic acids are said to contribute 75 to 85 percent of color while fulvic acids contribute 15 to 25 percent (Shapiro 1964; Steelink 1977). Color is also related to iron concentrations as iron is retained by organic matter (Lamar 1968).

High color is undesirable for numerous industrial uses including: laundries, ice making, dairy, beverages, textiles, and pulp and paper. The 1962 USPHS standards for drinking water limit color to 15 units. The range in color reported for Minnesota peatland waters is about 30 to 650 units with bogs having greater color than fens (Table 20). High color is somewhat detrimental to fish and other aquatic life by inhibiting light penetration (McKee and Wolf 1963). Hasler et al. (1951) treated two brown-colored lakes in Wisconsin with lime and successfully reduced color and improved light penetration.

2. Humates

Humates are dark colored materials that make up a major part of soil organic matter and consist of two types:

a. Humic acid - the part of organic matter that is soluble in dilute base and insoluble in alcohol and acid, and

b. Fulvic acid - the part of organic matter that is soluble in water and remains in solution after neutralization and after all humic acid has been leached out (Steelink 1977).

Both humic and fulvic acids are polymeric, polyetectrolytes of ill defined composition. Humic acids have higher molecular weights (5000 to 100,000) than do fulvic acids (300 to 2000) and contain more aromic carbon atoms. Humic substances have several properties including: metal chelation, base exchange, an affimity for proteins, and a high absorptive capacity. As mentioned in the previous section, humates contribute substantially to color (Steelink 1977).

Humates can absorb 1 to 17 percent of their own weight in metal with the order of stability of these complexes, from most to least stable, being: Pb > Cu > Ni > Co > Zn > Cd > Fe > Mn > Mg. Fulvic acids form tighter bonds than humic. Metal-fulvic acid-phosphate complexes are common and most phosphate in highly colored waters may be held in this manner. Humates also absorb proteins, insecticides and herbicides, and plasticizers and may operate as a concentrator of these trace toxic materials. Humates are also known to stimulate algae and the complexed iron may be important in this response (AWWA 1967; Steelink 1977; Prat 1955, 1968).

The AWWA (1967) has listed several adverse, neutral, and beneficial effects of humates and color:

Adverse

1. Aesthetically undesirable
2. Humic acids cause taste, interfere with chlorination, may induce bacteria
3. Exceed some industrial limits
4. Foul anion-exchange resins
5. Interfere with colorometric analysis of other constituents
6. May limit productivity by absorbing light
7. Interfere with coagulation
8. May make lead soluble and result in corrosion
9. Hold Fe and Mn in solution

Neutral

There is no evidence that humates are directly harmful to humans although they could make lead soluble.

Beneficial

They help maintain a high primary productivity in waters by stimulating algae.

The normal range of humic plus fulvic acid in surface water is 1 to 5 mg/l but may be as high as 10 mg/l (Steelink 1977). Largin (1976) reports humic and fulvic acid concentrations for peatland waters in the USSR ranging from 16 to 45 mg/l for fens and 85 to 140 mg/l for bogs. Within peat profiles, Bremmer *et al*. (1978) reports that fulvic acid decreases and humic acid increases with depth.

Related Properties

1. Suspended Sediment

Suspended sediment normally consists of fine mineral particles, organic detritus, and plankton but the suspended sediment from peatlands is composed mostly of organic matter (McKee and Wolf 1963). There is no drinking water standard for suspended sediment although a limit of 0.1 mg/l has been recommended in California. High suspended sediment is undesirable for several water using industries including: textile pulp and paper, beverages, dairy products, power plants, laundries, and steel making. Suspended sediments harm fish and shellfish by abrasive injury, clogging gills and respiratory passages, and blanketing stream bottoms, thereby killing eggs, young and food organisms and destroying spawning beds. Fish are also indirectly affected by suspended sediments which reduce light penetration and oxygen concentrations. Suspended sediment concentrations less than 60 mg/l are generally not harmful to fish while values greater than 90 mg/l have been harmful (McKee and Wolf 1963). Largin (1976) reported that concentrations of suspended sediment in waters leaving peatlands in the USSR ranged from 8 to 35 mg/l with greater amounts leaving fens.

Since suspended sediment may be common in peatland runoff waters, it is of interest to determine how fast these materials settle out. In order to perform this analysis it is first necessary to determine the particle size distribution for the sediment. An example of particle size analysis for sediment from a peatland in Poland is shown in Figure 33. This analysis indicates at point 1 that 42 percent of the sample is made of particles smaller than 17 μm (0.17 mm). Next, assuming that peat particles settle in accordance with Stoke's law, the time of sedimentation to a certain depth can be determined for the various particle sizes (Figure 34). For example, after six minutes of settling (point Q), all peat particles larger than 17 μm (58 percent of the sample) have settled to a depth of 0.25 meters.

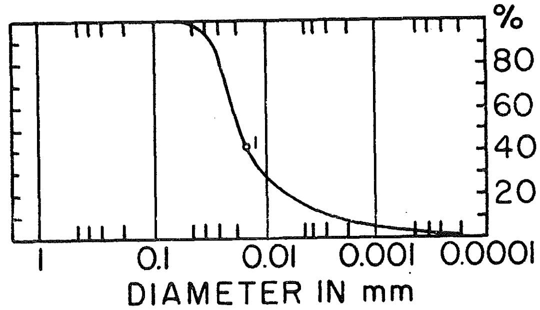

FIGURE 33: PARTICLE SIZE DISTRIBUTION CURVE FOR SEDIMENT FROM A PEATLAND IN POLAND (FROM MARCINEK AND KEDZIORA, 1976).

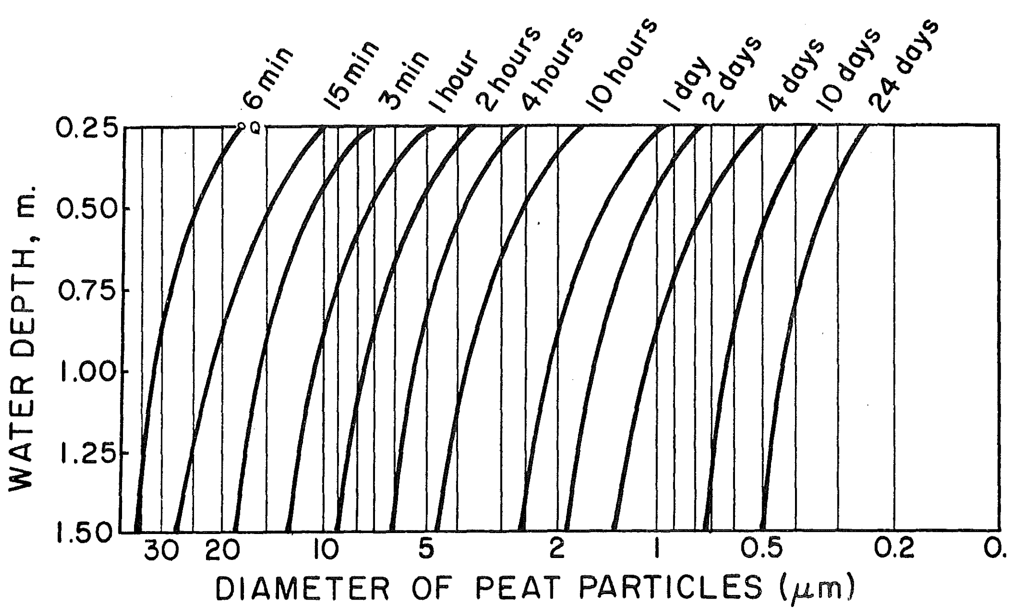

FIGURE 34: PARTICLE SIZE OF PEAT SUSPENSION AS A FUNCTION OF WATER DEPTH AND TIME OF SEDIMENTATION (FROM MARCINEK AND KEDZIORA, 1976).

The concentration of suspended sediment remaining at a given depth can be determined from Figure 35. The concentration remaining would be 0.23 kg/m^3 (point 1) which is equivalent to 230 mg/l.

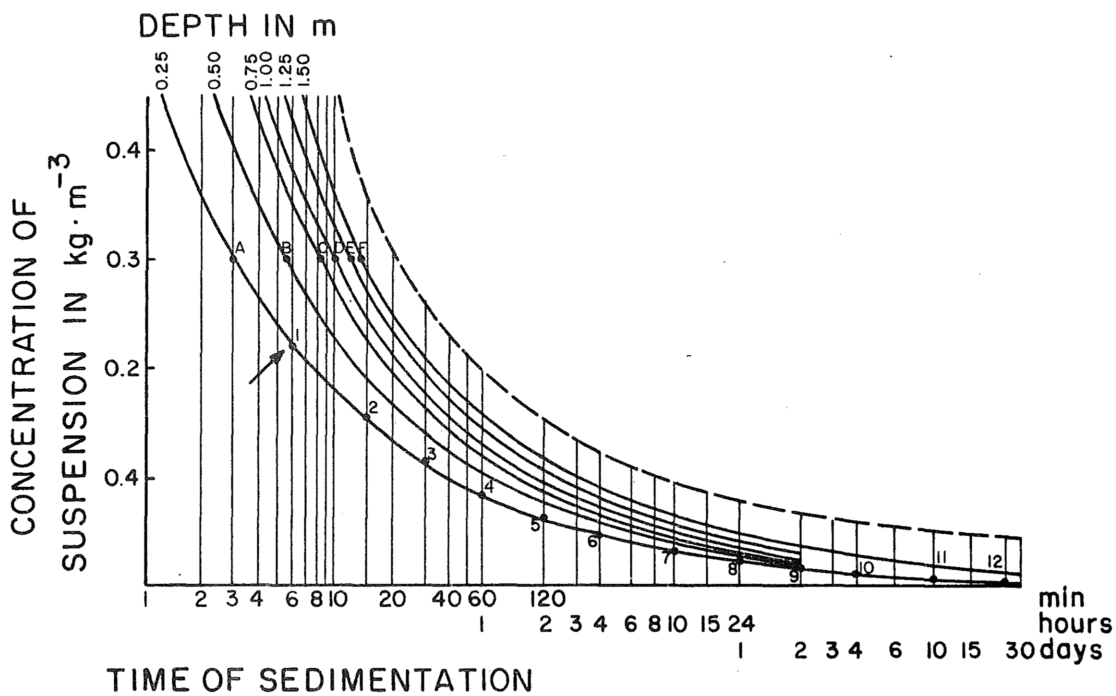

FIGURE 35: CONCENTRATION OF PEAT SUSPENSION AS A FUNCTION OF WATER DEPTH AND TIME OF SEDIMENTATION (FROM MARCINEK AND KEDZIORA, 1976).

This concentration can also be determined by multiplying the initial concentration of 0.535 kg/m^3 times the percent remaining in solution (42%) to equal 0.23 kg/m^3. Based on these three figures about 5 hours of settling time would be needed to settle 90 percent of this material, and the remaining concentration would be 52 mg/l (10% times 0.523 kg/m^3). To settle all the material would require more than 24 days. Once a desired outflow concentration or percent settling is selected, a settling pond can be designed based upon inflow rates of water, initial concentrations, and outflow.

2. Dissolved Oxygen

The major source of oxygen in peatland waters exposed to the air is the atmosphere and, to a lesser extent, photosynthesis by plants. Oxygen is depleted by peat micoorganisms and other oxygen consuming substances. The dissolved oxygen is an indicator of the biochemical condition of the water and is actually a measurement of the balance between oxygen-producing and oxygen-consuming processes. The solubility of oxygen is mainly a function of temperature and pressure with concentrations decreasing with increasing temperature (Hem 1970). Oxygen concentrations affect the solubilities of several constituents, for example iron and manganese which are more soluble at low dissolved oxygen concentrations.

Fish can acclimate to some changes in dissolved oxygen by adjusting respiration. Generally, dissolved oxygen concentrations of 5 mg/l or more are satisfactory for most life stages and activities of freshwater fish (McKim et al. 1975). Sluggish fish, like carp and pike, can withstand lower dissolved oxygen concentrations than lively fish, such as trout. The lethal limit is 6 mg/l for trout in soft water, 3 mg/l for warmwater fish, and 2 mg/l for coarse fish. Low dissolved oxygen concentrations can also enhance the effect of toxic substances (McKee and Wolf 1963).

3. Redox Potential

The redox potential (Eh) is synonomous with the "oxidation potential" and the "oxidation-reduction potential." For oxidizing environments Eh is positive, while in reducing environments Eh is negative. The redox-potential is expressed either in volts or millivolts and for oxygenated water at the air-water surface the Eh would be about 0.35 to 0.55 volts at a pH of 6 to 8 (Hem 1970).

As expected, there is a close correlation between redox-potential and dissolved oxygen with the lowest oxygen being found in waters with the greatest negative redox-potential. High negative Eh's appear to reduce vegetative growth in peatlands (Pierce 1953).

Summary

The significance of water quality constituents in terms of effects on fish and with respect to drinking water and pollution standards is summarized in Table 26. It appears that peatland waters have the potential of exceeding the toxic limits for fish for the constituents: pH, zinc, and dissolved oxygen; exceeding drinking water standards for iron and color; and exceeding the MPCA standards for fisheries and recreation by: pH, color, and dissolved oxygen. Unfortunately, insufficient information exists concerning heavy metals in peatland runoff waters to evaluate their importance. It must also be mentioned that standards are not necessarily the best indicator of water quality. For example, no standards address the importance of humates or suspended sediment.

Table 26. Summary of water quality constituents in terms of effects on fish and pollution standards.

Constituent	Range in[a] Peatlands	Toxic limit for fish	1962 USPHS Drinking Water Stds.	MPCA stds.[b] Fisheries & Recreation
	mg/l			
pH	3.3-7.2	<4.8, >79.2		6.5-8.5
Acidity	24-72			
Alkalinity	26-82			
Spec. Conductivity[c]	20-370	>4000		
Nitrate	<0.45		45	
Total-N	<2.0			
Phosphate	<0.35			
Total-P	<0.40			1.0
Calcium	<100			
Magnesium	< 20	100-400		
Sodium	<3.0			
Potassium	<4.0	>400		
Manganese	<0.2		.05	
Iron	<5.0	> 50	0.3	
Alumimum	<1.5	> 5		
Sulfate	<20		250	
Copper	<0.5	> 3	1.0	.01
Nickel	unk	>0.8		
Zinc	<0.3	.01-1[d]	5.0	
Lead	<.05	>0.1	.05	
Mercury	unk	>.02		
Color	30-650		15	30
Humates	16-140			
Sus. sediment	8-35			
Dissolved oxygen	0.7-7.8	2-6		6,7

[a] From Table 19.
[b] Minnesota Pollution Control Agency.
[c] µmhos/cm
[d] In soft water.

F. EFFECTS OF PEATLAND USES ON WATER QUALITY

Disturbed peatlands are often those that have been drained and sometimes cleared and developed for agriculture, forestry, or peat mining. Both agriculture and forestry can involve fertilization which has additional water quality implications.

1. Agriculture

Most water quality studies on agricultural peatlands have focused on nitrogen, phosphorus, and potassium concentrations in soil solution or discharge waters resulting from fertilization. Generally, drainage, cultivation, and fertilization increases nitrogen, phosphorus, and potassium concentrations in discharge and soil solution waters (Table 27). The magnitude of change varies substantially among locations and in some cases, values are extremely high. To better understand the potential effects of agricultural uses of peatlands on water quality, these studies are examined in greater detail for the major nutrients.

Nitrogen

Many investigators have measured nitrate-nitrogen in agricultural peatland waters (Farnham 1976; Hortenstine and Forbes 1972; and Avnimelech *et al*. 1978). Miller (1974) found that 80 to 91 percent of nitrogen leached in Ontario peat soils was nitrate. Erickson and Ellis (1971) in Michigan stated that nitrate accounts for over half of the total nitrogen loss with the remainder being bound on colloidal organic matter. In Ontario, Nicholls and MacCrimmon (1974) found greater concentrations of ammonia-nitrogen than nitrate in

Table 27. Summary of nitrogen, phosphorus, and potassium concentrations and loadings from drained peatlands used for agriculture.

				Nitrate-Nitrogen		Orthophosphate		Potassium	
Location	Peat Type	Sample Type	Agricultural Treatment	Conc. mg/l	Export Kg/ha/yr	Conc. mg/l	Export Kg/ha/yr	Conc. mg/l	Author, year
Michigan, USA	Muck	Runoff	Culti. & Fert.	.2-2.8	18.7	.01-0.3	1.45	0.9-4.8	Erickson & Ellis, 1971
Florida, USA	Muck	Soil solu.	Uncleared	3-16	-	0.04-.10	-	1.0	Hortenstine & Forbes, 1972
Florida, USA	Muck	Soil solu.	Newly cleared	47	-	0.9	-	12	Hortenstine & Forbes, 1972
Florida, USA	Muck	Soil solu.	Culti. & fert.	50-150	-	1-9	-	15-60	Hortenstine & Forbes, 1972
Ontario, Canada	Muck	Runoff	Uncultivated	.04	<0.1	-	0.34		Nicholls & MacCrimmon, 1974
Ontario, Canada	Muck	Runoff	Culti. & fert.	1.78	4.15	-	1.56		Nicholls & MacCrimmon, 1974
Ontario, Canada	Muck	Soil solu.	Uncultivated	<.07	-	.019	-		Nicholls & MacCrimmon, 1974
Ontario, Canada	Muck	Soil solu.	Culti. & fert.	1.68	-	.031	-		Nicholls & MacCrimmon, 1974
Ontario, Canada		Runoff	Culti. & fert.	-	91-196	-	14-26		Miller, 1974
W. Germany	High bog	Runoff	Culti. & fert.	5-20	-	0.5-2.0	-	5-15	Kuntze & Eggelsmann, 1975
Minnesota, USA	Hemic	Runoff	Uncultivated	.01-4.30	-	.01-.66	-		Farnham, 1976
Minnesota, USA	Hemic	Runoff	Culti. & fert.	.06-1.40	-	.01-.13	-		Farnham, 1976
New York, USA	Muck	Runoff	Culti. & fert.	1-35	39.2-87.5	1-10.0	.6-30.7		Duxbury & Peverly, 1978

soil solution samples but less in runoff waters. Less ammonia than nitrate in runoff was also found in New York by Duxbury and Pererly (1978). Franciszek and Horst (1976) determined the forms of nitrogen in a highly decomposed cultivated West German peat and found that only 2 to 6 percent of the total nitrogen was nitrate while 33 to 50 percent was organic, 1 to 2 percent was ammonium, 22 to 30 percent was insoluble humic-N, and 16 to 27 percent was un-identifiable nitrogen. Higher nitrogen concentrations have also been associated with greater humic acid concentrations (Kuntze and Eggelsmann 1975). It appears that the proportion of nitrate varies in peatland waters and other forms should not be ignored.

Soil solution studies have shown that in uncultivated peatlands, most nitrate production occurs in the upper 30 cm of the peatland while on drained, cleared, and fertilized areas, nitrate was higher at all depths up to 150 cm with concentrations decreasing with depth from the surface. Concentrations of nitrate in soil solution also increased with time after fertilization (Hortenstine and Forbes 1972; Nicholls and MacCrimmon 1974).

Higher concentrations of nitrate in soil solution and runoff waters have been observed during the periods of high rainfall in the summer (Farnham 1976; Nicholls and MacCrimmon 1974). Erickson and Ellis (1971) found that much of the nitrate was flushed from the system in spring before plant growth. Low nitrate concentrations were observed in May and June due to uptake of nitrogen by plants.

The effect of peatland drainage on nitrate-nitrogen release has been intensively studied at the Hula Valley peatland in Israel. Avnimelech (1971) performed a laboratory study of the formation and reduction of nitrate in peat and determined that nitrate production increases with temperature, moisture content up to field capacity, and aeration. Nitrate content was reduced in anaerobic conditions. Later, Raveh and Avnimelech (1973) evaluated both sprinkling and flooding to promote anaerobic conditions and enhance denitrification. Both flooding and sprinkling reduced nitrate concentrations in the surface peat about 80 percent. Sprinkling would be most effective in fall when soil temperatures were high. To control the problem of nitrate leaching, Avnimelech _et al_. (1978) recommended a number of management schemes:

a. Maintain the groundwater level at 1 to 2 meters below the surface and control flooding by gates and large channels.

b. Minimize the depth of the aerated layer, thus reducing nitrification and use sprinkler irrigation after a dry spell to promote rapid denitrification.

c. Use special crops, such as rice which is grown under flooding or forage crops such as alfalfa and Rhodes grass which reduce nitrate accumulation.

Phosphorus

There has been little study of the forms of phosphorus in peatland drainage waters and most researchers have measured ortho-phosphate. Duxbury and Peverly (1978) measured molybdate reactive phosphorus (nominally PO_4) and total dissolved phosphorus and found reactive phosphorus concentrations were greater than total phosphorus, concluding that little organic phosphorus occurred in waters draining from this New York peatland. In soil solution samples, Hortenstine and Forbes (1972) found that inorganic phosphorus was 29 to 62 percent of organic phosphorus with the percent inorganic increasing with depth.

Soil solution studies have shown that cultivation increases inorganic phosphorus (ortho-phosphate) and after fertilization, ortho-phosphate concentrations are much higher in the surface 30 cm. Concentrations decreased with depth but increases have been observed even at the 110 cm depth. It has been concluded that microbial populations respond vigorously to increased oxygen supplies after drainage, resulting in the release of nitrogen and phosphorus (Hortenstine and Forbes 1972; Nicholls and MacCrimmon 1974). Also, Fox and Kamprath (1971) conclude that organic soils have a low capacity to absorb phosphorus and soluble phosphorus fertilizers will readily leach from peat. More adsorption of phosphorus would occur at higher pH's.

Phosphorus concentrations in streamflow have been found to be highest during spring snowmelt and during periods of greater precipitation and discharge (Nicholls and MacCrimmon 1974; Fox and Kamprath 1971; Kuntze and Eggelsmann 1975; and Farnham 1976). Nicholls and MacCrimmon (1974) observed that during the period of greatest discharge, 98 percent of the phosphorus was in the soluble reactive form. In general, it appears that the relative proportions of the various phosphorus forms change with rainfall.

Potassium

Potassium concentrations have not received as much attention as nitrogen and phosphorus but they also have been found to increase in soil solution samples after cultivation and fertilization. Concentrations of potassium also decreased with depth in soil solution (Hortenstine and Forbes 1972). Kuntze and Eggelsmann (1975) state that potassium concentration are expected to be high in peatland waters after fertilization because potassium adsorption in peat soils is small.

Specific Conductivity

Specific conductivity has been found to increase 90 percent in soil solution samples and runoff waters after cultivation and fertilization (Nicholls and MacCrimmon 1974).

2. <u>Forestry</u>

The effects of peatland forest management on water quality have been investigated in terms of both drainage and drainage plus fertilization practices. In Finland, areas drained for 20 to 40 years showed increased pH and conductivity and reduced organic carbon, while total phosphorus and total nitrogen remained the same (Table 28). Drainage in Czechoslovakia increased pH, calcium, sodium, potassium, and manganese and decreased acidity, conductivity, phosphate, and to a lesser extent iron and ammonium (Table 29). Changes were more pronounced when the ditch intercepted mineral soil.

Table 28. Effects of peatland forest drainage and fertilization on water quality in Finland (after Heikurainen et al. 1978).

	Sedge bog		Sedge Pine Swamp	
	Undrained	Drained	Undrained	Drained & Fertilized
Organic C mg/l	22.5	19.3	24.6	21.8
Total-P mg/l	.027	.030	.019	.097
Total-N mg/l	.518	.635	.427	.475
Conductivity ms/cm	31.7	56.0	36.6	35.2
pH	5.80	6.19	4.49	4.98

Table 29. Effects of peatland forest drainage on runoff water quality in Czechloslovakia. Means of 10 values. (Ferda and Novak 1976).

		Drained, ditch btm. in	
	Undrained	Peat	Mineral
pH	3.5	3.9	4.4
Alkalinity mval/l	0	0	0
Acidity mval/l	2.0	1.1	0.5
Conductivity ms/cm	202	111	84
Ca mg/l	2.7	2.8	3.3
Na mg/l	1.1	1.4	1.9
K mg/l	0.8	1.2	1.3
Fe mg/l	4.9	4.3	2.5
Mn mg/l	0.10	0.14	0.22
SO_4 mg/l	19.8	16.0	23.0
CL mg/l	5.2	4.6	3.2
NH_4-N mg/l	0.4	0.3	0.1
NO_2-N mg/l	0	0	0
NO_3-N mg/l	0	0	0
PO_4-P mg/l	0.18	0.04	0.03
SiO_2 mg/l	11.0	11.0	13.0

Heikurainen et al. (1978) reported that ditching increased suspended sediment from pre-drained levels of about zero to 1 kg/ha/day. About 10 weeks after drainage, suspended sediment levels dropped to 8 to 70 kg/ha/day but would rise again during rain storms. Heikurainen (1975) also noted that peat sediments carry phosphorus and reduce light penetration. In separate studies, Heikurainen et al. (1978) observed that organic carbon, which is related to organic acid content, was increased by drainage.

Drainage and fertilization of forested peatlands with phosphorus and potassium in Finland resulted in an increase in total phosphorus in discharge waters (Table 28). On a drained transition type peatland with 70-year old pine in Czechoslovakia fertilization with 600 kg/ha of NPK and 5 tons/ha of $CaCO_3$ also resulted in slight phosphorus increases (Table 30). Most other ions changed very little in response to the fertilizer except that iron was reduced. Liming clearly raised the pH and calcium concentrations and reduced iron concentrations further. The effect of liming on nitrogen mobilization has been studied by Kaunisto (1976) in Finland on sphagnum and woody carex-sphagnum peats. Liming reduced ammonium and nitrate nitrogen in the peat due to increased bacterial activity and the subsequent biological fixation of nitrogen.

3. Mining

Studies on the effects of peat mining on water quality are rare. No difference in pH, percent transmittance and sediment was found among three bogs in New Brunswick, Canada, two of which were being

Table 30. Effects of fertilization and liming on the quality of groundwater in a forested transition peatland in Czechloslovakia. Means of 10 values (Ferda and Novak 1976).

	Treatment		
	O	NPK	NPK & Ca
pH	3.8	3.9	6.0
Alkalinity Mval/l	0	0	0.1
Acidity Mval/l	2.4	2.1	1.4
Ca mg/l	4.0	3.6	16.7
Na mg/l	1.8	1.8	1.9
K mg/l	1.5	1.5	1.7
Fe mg/l	10.6	8.2	6.7
MN mg/l	0.2	0.2	0.1
SO_4 mg/l	17.0	19.0	18.0
CL mg/l	6.2	7.1	7.5
NH_4-N mg/l	3.2	2.0	1.9
NO_2-N mg/l	0	0	0
NO_3-N mg/l	0	0	0
PO_4-P mg/l	0.9	1.2	1.1

mined by the milled peat method, and a raised bog in Maine, USA (Korpizaakke and Pheeney 1970). Little information was provided in this report, although Korpizaakke and Pheeney (1970) recommended that to prevent downstream suspended sediment problems, outlet ditches could be meandered. Also, the final outlet should not connect directly to a stream but the water should be allowed to filter through unditched peat.

The effects of drainage and mining on water quality in the USSR are summarized in Table 31 oligotrophic and eutrophic peatlands. Drainage and mining appeared to increase pH, calcium, magnesium,

Table 31. Effects of drainage and peat mining on water quality for oligotrophic and eutrophic peatlands in the USSR (Largin 1976).

	Oligotrophic (10)		Euthropic (8)	
	Undrained	Drained & Mined	Undrained	Drained & Mined
pH	3.5-5.9	4.0-6.6	5.3-7.1	6.6-7.7
Ca mg/l	1.2-8.0	8-30	12-100	50-88
Mg mg/l	0.1-2.8	0.5-6.0	0.5-10.0	9-18
HCO_3 mg/l	1-18	12-90	50-420	120-320
SO_4 mg/l	2-8	2-24	3-16	8-80
CL mg/l	2-9	0-1.0	1-6	1-5
NO_3-N mg/l	0.1-1.0	0.1-0.5	0.1	0.5-2.0
NO_2-N mg/l	0.1	0	0.1	0.5-1.5
Sus. sediment mg/l	8-20	8-21	15-35	26-35
Humic & Fulvic acids mg/l	85-140	100-230	16-45	15-70

bicarbonate, sulfate, and humic and fulvic acid concentrations for the bogs and increase pH, magnesium, sulfate, nitrate, nitrite, and humic and fulvic acids for the fens. Chloride and nitrate were reduced after mining the bogs while concentrations of calcium and bicarbonate decreased after mining fens. The pH, iron, calcium, magnesium, bicarbonate, and humic and fulvic acid concentrations increased with the age of drainage (Largin et al. 1976). Increased aeration was considered responsible for these increases. Seasonally, summer maximum concentrations were observed for iron, calcium, magnesium, and humic and fulvic acids; fall maximums occurred for bicarbonate; and spring minimums occurred due to snowmelt dilution.

4. Summary

Various peatland uses including: agriculture, forestry, and peat mining appear to modify the quality of both soil solution and discharge waters. The management practices that actually produce changes are: clearing, drainage, cultivation, fertilization, and peat removal.

Agricultural drainage, cultivation, and fertilization appear to increase nitrogen, phosphorus, and potassium concentrations, and specific conductivities in both discharge and soil solution samples. Agricultural practices also seem to modify the relative amounts of the forms of nitrogen in discharge waters.

Peatland forest drainage is likely to increase pH, calcium, sodium, potassium, manganese, suspended sediment, and possibly conductivity while decreasing organic carbon and acidity and possibly iron, ammonium, and phosphate concentratons in discharge waters. Changes are magnified in both directions if ditches intercept mineral soil. Forest fertilization increases phosphorus and decreases iron in peatland waters. Liming increases pH and calcium and reduces iron concentrations further.

Peatland drainage and mining affect the water quality associated with bogs and fens differently. Bog drainage and mining may increase pH, calcium, magnesium, bicarbonate, sulfate, and humic and fulvic acids and decrease chloride concentrations. Fen drainage and mining may increase pH, magnesium, sulfate, humic and fulvic acids, and also nitrate and nitrite concentrations, but may reduce calcium and bicarbonate. As the age of drainage proceeds, pH, iron, calcium, magnesium, bicarbonate, and humic and fulvic acid concentrations may increase. This summary is essentially based only on one study of 18 bogs and fens in one country and much more information in other parts of the world is needed.

III. CONCLUSIONS AND RECOMMENDATIONS

The major findings and conclusions developed from this literature review are contained within the "Executive Summary" found at the beginning of this report. Briefly, it can be stated that the mining of peatlands does have the potential to modify the water resources of Minnesota's peatland both in water quantity and water quality. If forestry and agricultural uses of peatlands are considered as reclamation alternatives or other potential uses, these too can modify water quantity and quality.

Based upon this literature review, a number of gaps in information exist for which additional research is needed on both disturbed and undisturbed peatlands:

1. Groundwater movement
2. Interception by species
3. Infiltration and soil frost relationships
4. Evapotranspiration by various plant species
5. Effects of peat mining on runoff and other components of the hydrologic cycle
6. Chemical content of Minnesota peats, especially heavy metals

7. A water quality characterization of Minnesota's peatlands, both mined and natural
8. Heavy metals in peatland runoff waters
9. The effects of various peat mining methods, for example hydraulic mining, should also be evaluated.

This same information is needed elsewhere in the United States especially in the Lake States, the northeast, the east and southern coasts, the Pacific northwest and Alaska.

Until sufficient information is obtained, it is recommended that:

1. All new and existing peat mining operations should be monitored for water quantity and water quality effects, and
2. Adequate mitigative measures should be employed to prevent adverse effects.

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