PRESENT AND POTENTIAL EFFECTS OF ANTHROPOGENIC ACTIVITIES ON WATERS ASSOCIATED WITH PEATLANDS IN ALBERTA

edited by

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EXECUTIVE SUMMARY

Title

Present and Potential Effects of Anthropogenic Activities on Waters Associated with Peatlands in Alberta

Contents of Report

Purpose

A literature review and problem analysis of the present and potential effects of human activities in peatlands on the quality and quantity of associated waters is presented in this report.

The specific objectives were to describe:

1. The present state of knowledge about waters associated with peatlands in Alberta.
2. The kinds of anthropogenic activities which may have an impact on waters associated with peatlands in Alberta.
3. The potential impact of anthropogenic activities on waters associated with peatlands.
4. The significant deficiencies in information and knowledge with regard to Alberta peatlands, and recommendations for further action, if required.

Premises

The area of peatlands in Alberta is estimated at 12.7 million hectares, accounting for about 19% of the total land area of the province. There has been relatively little use of peatlands in Alberta and in the rest of Canada, although some types of peatland exploitation are increasing. The utilization of peatlands in other parts of the world has been shown to have various effects on the environment, particularly with regard to the downstream quality and quantity of water. There is increasing use of peatlands in Alberta, especially for agriculture, forestry, and horticultural peat moss extraction. Peatlands can also be affected by pollutants, particularly the deposition of acidic and acid-forming substances. Because of the considerable area of peatlands in Alberta, the increasing uses of peatlands for various purposes, and the potential impacts of pollutants, there is concern about the effects on the quality of waters associated with peatlands. Consequently there is a need for an initial review of the nature and extent of these problems in Alberta, and for identification of those areas that require further attention.

Methods

The objectives indicated above were addressed through a review and analysis of pertinent world and local literature. The nature, properties, and functions of peatlands are reviewed in the first few chapters of the report. This is followed by an examination of the kinds and extent of activities which may affect peatlands in Alberta and, based on the literature from other provinces and countries, a review of the actual hydrochemical and hydrological impacts which could occur.

Chapter Summaries

Chapter 1. Introduction

A brief introduction to the report and the project objectives are presented.

Chapter 2. Types, Distribution, and Extent of Peatlands in Alberta

Information about the peatland resource in Alberta, as well as terminology and classification systems for describing peat and peatlands, are presented. The total area of peatlands in Alberta is estimated at 12.7 million ha or about 19% of the total area of the province. Peatlands occur predominantly in boreal regions and in the far north are characterized by subsurface permafrost conditions. A map was produced to depict the areal coverage of peatlands within the physiographic districts of the province. Information about peatland properties within broad ecological regions is also provided on the map. Peatland inventory information is currently available in the form of forest cover, geomorphological, ecological, soils, and other maps. The coverage varies from detailed to exploratory scales of inventory, but information about the hydrology and hydrochemistry of peatlands which can be derived from these is limited.

Chapter 3. Peatland Vegetation and Biogeochemistry

The development of peatlands, with particular reference to Alberta, is discussed in Chapter 3. Peat formation can occur under a variety of vegetational types as the basic requirement for peat accumulation is a cool, wet, anaerobic environment. The kinds of peat-forming communities are influenced by climate, nutrient availability, moisture regime, and shade. A hydrochemical categorization of peatlands based on nutrient and ion levels is as follows: bog, characterized by high acidity (pH 3.5 to 3.9) and low nutrient and ion levels; poor fen, characterized by moderate to strong acidity (pH 4.0 to 5.5) and low nutrient and ion levels; transitional rich fen, characterized by neutral reaction (pH 6.0 to 7.2) and moderate nutrient and ion levels; and, extreme rich fen, characterized by moderate alkalinity (pH 7.3 to 7.8) and high nutrient and ion levels. Swamps and marshes with peat deposits are also relatively rich, being similar in hydrochemistry to extreme rich fens.

Peatlands have large microbial populations that control the extent of decomposition of peat. Substrate quality and environmental conditions both interact to inhibit decomposition, and organic materials thus accumulate over time. Two modes of peat accumulation have been postulated. Terrestrialization is the filling in of water bodies by organic deposits eventually resulting in relatively dry organic terrain. Paludification is the formation of peatlands over previously forested uplands, grasslands, or even bare rock, due to climate or autigenic processes. It literally means 'swamping' of previously relatively dry land. In North America the first peatlands began to develop in depressions created by the retreating ice about 12 000 years ago. An abundance of peatlands developed by paludification appears to be related to climatic changes and subsequent influences on vegetation.

Chapter 4. Chemical and Physical Properties of Peat

The general chemical and physical properties of peat are reviewed in Chapter 4. Peat consists primarily of carbon, the relative percentage of which depends on the type of peat and its stage of decomposition. Carbon contents of peat can be as high as 50 to 55% and, by definition, as low as 17%. Carbon and ash contents are inversely related, ash contents being only a few percent in high carbon peats. The major components of peat are cellulose, hemicellulose, lignin, bitumen, and humic materials. The content of most plant nutrients is higher in fens than in bogs; these include nitrogen, phosphorus, potassium, calcium, magnesium, and sodium. Heavy metals of significance in peat materials are iron, manganese, zinc, copper, lead, chromium, nickel, mercury, and uranium.

The pH of peat ranges from about 3.5 in bogs to about 7.5 in fens. Acidity in peats arises chiefly from carboxyl and phenolic functional groups. The total or reserve acidity is high in many types of peat. Cation exchange and buffering capacities of peats are high per unit weight of peat, but they are low when expressed on a volume basis. The cation exchange capacity and acidity of peats are interrelated. The contents of base cations, which are essential nutrients to organisms, and of aluminum, which can be toxic, are dependent on the relationship between acidity and the exchange complex.

The bulk densities of peat are low; they generally increase with depth and degree of decomposition and decrease with higher fiber contents. Higher bulk densities result in lower porosity, water retention, and hydraulic conductivities. The hydraulic conductivity of peat has a wide range, from 10^{-9} $cm \cdot s^{-1}$ to 8 $cm \cdot s^{-1}$.

Data for Alberta peatlands which have been reported in the literature are presented for many of these chemical and physical properties.

Chapter 5. Hydrology of Peatlands in Alberta

Various water balance equations and their components are discussed in Chapter 5 in an attempt to elucidate the movement of water within peatlands. Evapotranspiration rates are positively correlated with the height of the water table and are therefore higher for peatlands than for mineral soils and higher for fens than bogs. During the growing season, stream discharge from peatlands may be affected because of the high evapotranspiration rates.

The upper 30 to 70 cm active layer (acrotelm) has the highest heat and water exchange within a peatland. This layer can contribute up to 80% of stored water to drainage due to high permeability and high hydraulic conductivity. Movement of stored water is generally horizontal during periods of excess moisture and vertical during drought. Fen water tables fluctuate less than those of bogs because the former are sustained by groundwater. Noticeable fluctuations occur when initially low water table levels are recharged by precipitation. Recharge rate depends on the peat porosity which is inversely correlated with depth. Further, the flat topography and retarding effects of peatland vegetation significantly reduce peak flood flows in a drainage basin.

Drainage alters the hydrologic balance of a peatland as well as its physical structure. The water table is lowered, increasing the zone of aeration and oxidation. This increases decomposition and subsidence and decreases permeability. An increased storage capacity alters storm runoff and annual discharge. These alterations in hydrology are generally accompanied by changes in the water quality.

Chapter 6. Water Chemistry of Natural Peatlands

Information about the chemistry of peatland waters, presented in Chapter 6, is required for predicting and understanding possible impacts of drainage, acid deposition and other human influences on peatlands and associated waters. Information about the water chemistry of natural peatlands in Alberta is limited to a few studies in which pH, electrical conductance, and major ions have been reported. There is little or no information about nutrients, minor or trace elements, and properties of organic matter. However, levels of peat water constituents in Alberta are likely similar to those found in other provinces and countries.

Seasonal variations occur in peatland runoff flow rates and in concentrations of many constituents of waters. High flows produce higher values of pH, dissolved oxygen, and potassium, and lower levels of electrical conductance, acidity, calcium, manganese, iron, aluminum, and sulphates.

The major ions in peatland waters are calcium, magnesium, hydronium, and bicarbonate. These are highly correlated with each other, and with pH and electrical conductance. Electrical conductance and pH are, therefore, useful and easily measured indicators of peatland type.

Peatland waters are characterized by high levels of dissolved organic materials, with bog waters being generally higher in these coloured substances than fen or swamp waters. Functional groups on humic materials participate in the adsorption, exchange, and complexation of hydrogen ions, micronutrients, and trace metals. They thus buffer acidity and bind trace elements thereby reducing toxicity of certain elements. However, they may maintain the mobility of complexed substances and thus cause potential problems downstream.

Interactions of humic materials with other substances is highly complex. Interactions and reactions include: sorption of other organics, degradation by photolysis, precipitation or flocculation in waters of sufficient ionic strength, oxidation-reduction, and complexation of metal ions.

The contributions of peatlands to the chemistry of downstream waters depend on their extent in the watershed. Moreover, the amounts of inorganic and organic solutes in peatland and runoff waters can vary seasonally.

Chapter 7. Peatland and Runoff Water Properties in Relation to Surface Water Quality Guidelines

A review of studies from various countries wherein effects of peatland development on water quality were investigated is presented in Chapter 7. Data were compared with water quality guidelines of Environment Canada and Alberta Environment, and specific parameters adversely affected by development were identified. Constituents of peatland runoff waters in which concentrations exceeded or fell short of water quality guidelines were considered to have potential for noncompliance in the receiving waters as well. Water quality parameters which could be deleteriously affected included the following: oxygen, biochemical oxygen demand, chemical oxygen demand, pH, nitrogen compounds, phosphorus compounds, suspended solids, turbidity, aluminum, iron, manganese, mercury, zinc, colour, and phenolics. Oxygen can be depleted to below guideline levels because of elevated biochemical oxygen demand and chemical oxygen demand resulting from the input of dissolved and particulate organic substances. Increased acidity is considered to result from oxidation of reduced sulphur forms upon drainage of peatlands. However, in many studies increases in pH were noted to occur in situations where waters had come in contact with mineral subsoils in drainage ditches. Accelerated eutrophication of water bodies resulting from input of nitrogen and phosphorus released from developed peatlands has been reported from several countries. The highest levels of these nutrients have been found where heavy fertilization of agricultural crops is practised. In addition to the minor and trace elements of concern noted above, cadmium, lead, and copper may also increase in runoff waters, but results are inconclusive and research is needed to assess potential problems related to these elements.

Chapter 8. Summary of Research Related to Peatland Hydrology and Hydrochemistry in Alberta

Work carried out within the province of Alberta on hydrology and hydrochemistry of peatlands and associated water bodies is summarized in Chapter 8. Investigations which only indirectly pertain to peatlands but which can provide baseline data for further studies are reviewed in addition to research dealing directly with peatlands. There is generally a scarcity of information in Alberta about properties of peatlands and associated water bodies. Provincial and federal water quality monitoring programs provide information about lakes and rivers in the province. Only larger water bodies are monitored and the data are, therefore, useful only in broad, regional assessments of peatland development impacts on water quality as opposed to evaluations of small watershed areas. However, information is available for smaller water bodies in the Athabasca oil sands area and in the Cold Lake region. Some information has been gathered in these regions through environmental impact assessments carried out for oil sands and heavy oil developments. Water quality information has been obtained as part of peatland drainage research projects for forestry purposes. Much of the hydrochemical information for peatland surface waters has been gathered through various peatland ecology studies carried out by the University of Alberta. A large amount of information on peatland surface waters and peat profile composition has also been collected recently in the programs of the Northern Forest Research Centre, Forestry Canada.

Chapter 9. Acid Deposition and its Effects on Peatlands

The nature of acid deposition is described briefly in Chapter 9. The natural acidification of peatlands is reviewed as it is necessary to know the effects of natural processes before any evaluations of anthropogenic acidification effects can be made.

Processes which produce acidity in peatlands include: cation exchange in peatland vegetation, particularly *Sphagnum* species; dissociation of polygalacturonic acids produced by *Sphagnum* species; uptake by nutrients by peatland biota; oxidation and reduction of sulphur and nitrogen compounds; and, production of organic acids by decomposition of vegetation and by excretion from plants. Peatlands, through intrinsic properties, can govern their own evolution from slightly alkaline or circumneutral to more acidic types of peatlands. These changes are reflected in shifts in plant communities over time as revealed, for example, by examination of plant remains in stratigraphic profiles of peatlands.

Anthropogenic sources of acidity can influence peatlands in various ways. There is some evidence that bryophytes, the dominant vegetation of peatlands, may be more vulnerable to acid deposition than vascular plants. Spraying of simulated acid rain on various mosses has shown that detrimental effects can occur at pH 3.5 or lower. Various British and European studies have linked modifications in mire vegetation over time to atmospheric pollutants. Transplants of *Sphagnum* species from non-industrial areas were shown to be unable to thrive in polluted areas. From a biogeochemical perspective, there is evidence that sulphate reduction and nitrate uptake have increased considerably in wetlands of eastern North America since the period before acid deposition. Experimental application of sulphuric and nitric acids to peatlands has shown that sulphates and nitrates are rapidly assimilated by the vegetation and microorganisms. Thus, nutrient uptake and microbial reduction appear to mitigate acidification processes, but considerably more research is required to assess long-term effects of mineral acid deposition.

Whole ecosystem responses to acid deposition have mainly been considered in terms of sensitivity of different peatland types. Poor fens are recognized as being most sensitive to acidification effects because of their very low alkalinities, pH values of about 4.5 to 6.0, and lack of input of any mineral materials. Richer fens have higher buffering capacities than poor fens. Bogs are considered to have low sensitivity because they are already highly acidic.

Chapter 10. Forestry and its Effects on Peatlands and Associated Waters

The potential for peatland forestry in Alberta, and the possible environmental consequences, are discussed in Chapter 10. Regulated drainage generates tree growth by improving aerobic conditions which increase biological activity, nutrient cycling, and improve soil structure. Physical alterations of the peat increase minimum and maximum runoff volumes. Evapotranspiration may increase slightly due to an increase in transpiration and interception. Drainage alters water quality by increasing suspended solids, dissolved organic matter, water colour and nutrient loading. Silvicultural practices also alter water quality. Fertilizer P and K are easily leached and over-application may increase surface acidity. Harvesting timber increases runoff and produces a rise in the water table. Further, in areas of permafrost, aggradation and deep frost cracking may occur due to the loss of an insulating snow cover brought about by windswept conditions. Over-drainage increases structural breakdown to the point where peat may lose its ability to retain water, becoming dry and cracked. Alberta peatland forestry is still in the research stage and results may differ from those found in the literature due to the varied peatland types and climate of the province. An area of about 4 million hectares is considered to have potential for drainage and improvement of forest growth.

Chapter 11. Peatland Agriculture and its Effects on Hydrology and Water Quality

The conversion of peatlands for agricultural use in Alberta is reviewed in Chapter 11. Improved pasture, forage crops, grain crops, and rough grazing and rangeland are the main kinds of agricultural use of peatlands in the province. Some effects of drainage and development found in other countries and provinces include: subsidence and oxygenation of the surface layers, increased levels of particulate and dissolved organics, and, increases in oxidation products such as sulphates in the runoff waters.

The rate of subsidence is influenced by biological oxidation, height of water table, character of organic material, compaction, burning, wind and water erosion, shrinkage and dehydration, and the cropping system used. Nutrient loading in the runoff waters may occur partly because of the mobilization of minerals, but will more likely occur because of fertilization. Further, organic carbon levels and suspended matter will increase during operations which drastically disturb the soil, such as clearing.

The area with potential for agriculture was roughly estimated at about one million hectares. The area currently being farmed is probably no more than 10% of this. Thus, a considerable area of peatlands may yet be developed for farming and some undesirable consequences with regard to water quality are possible.

Chapter 12. Effects of Mining, Linear Developments, and Other Activities on Peatlands and Associated Waters

Other anthropogenic activities which might have an effect on peatlands and associated waters, such as peat moss mining and linear developments, are reviewed in Chapter 11. Methods of peat moss extraction are described; impacts include higher levels of colour, suspended sediment, water temperature, acidity, nitrogen, potassium, iron, aluminum, and sodium in downstream waters. Linear developments such as seismic lines, pipelines, roads, power transmission lines, and drilling well roads and pads can all affect peatlands, the amount of the effect depending on the degree and extent of the disturbance. Effects include disturbance of surface drainage, surface vegetation, and soil thermal regime. Ponding of surface water can degrade permafrost and possibly divert nutrient-rich waters into surrounding nutrient-poor bogs. Major disturbances, such as surface mining, are also considered. The drainage of muskeg in a planned mine site in the Athabasca oil sands region and subsequent impacts on the Muskeg River are given as an example. Impacts included increased levels of turbidity, suspended solids, and nutrients such as nitrogen and phosphorus. However, these had little impact in terms of runoff from the drainage basin as a whole.

Conclusions

It is possible that the quality of waters associated with peatlands in Alberta will be deleteriously affected as a consequence of direct exploitation or of deposition of pollutants. Conversion of peatlands to agricultural or forest production, and the deposition of acidic and acid-forming pollutants, could have the most widespread effects in Alberta. Other anthropogenic influences such as horticultural or fuel peat extraction and linear developments could have relatively localized effects on water quality of downstream lakes and water courses. The degree of impact will be largely dependent on the extent of peatland development; development of a large proportion of a drainage basin can have a significant impact on levels of water constituents, while the impact of development of a small peatland area will be mitigated by relatively higher dilution in downstream waters. Another factor to consider in evaluating impacts is that some constituents in runoff waters from natural peatlands exceed water quality guidelines. There are significant deficiencies in the information base regarding peatlands in Alberta, as well as in general understanding of the forms and dynamics of various peatland water constituents. These deficiencies are addressed in some recommendations for action presented below.

Recommendations

Recommendations for Acquisition of Baseline Data on Peatlands and ssociated Waters

1. Identify peatland drainage basins with current or planned developments that can affect hydrology and hydrochemistry of associated waters.

2. Define the water quality of peatland surface and runoff waters, and examine the impacts of existing developments on water quality.

3. Develop a data base of the distribution, extent, types, and characteristics of peat and peatlands in Alberta.

Recommendations for Research on the Dynamics of Peatland Waters and Their Constituents

4. Determine the forms, interactions, and movement of toxic substances in natural and developed peatlands and their runoff waters.

5. Define the role of organic substances in peatland drainage waters in relation to acidification and fate of trace and nutrient elements.

6. Determine the effects of acidic and acid-forming deposition on peatlands and associated waters.

7. Quantify the hydrologic cycle for peatlands in Alberta.

Recommendations for Development of Tools to Assist Research and Management of Water Resources in Peatland Areas.

8. Develop and apply water quality and hydrologic models for evaluating influences of peatlands on water quality, for determining research needs, and for management purposes.

9. Develop and apply methodology for examining peatland-use impacts and for applying models on a geographic, or spatial basis.

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1. INTRODUCTION

L.W. Turchenek

The area of peatlands in Alberta is estimated at 12.7 million hectares, accounting for about 19% of the total land area of the province. Peatlands occur predominantly in the boreal regions of the province and in the far north are characterized by permafrost conditions in the subsurface. There has been relatively little use of peatlands in Alberta and in the rest of Canada, although some types of peatland exploitation are increasing. The utilization of peatlands in other parts of the world has been shown to have various effects on the environment, particularly with regard to the downstream quality and quantity of water.

The human activities that can affect the nature of peatlands and their receiving waters fall into different categories. The first category is direct exploitation of peatlands. The main activities and developments within this category in Alberta are agriculture, forestry, and horticultural peat moss extraction. In other parts of Canada, extraction of peat for use as fuel can also be included. A feature common to this category is the drainage of peatlands either prior to development, or as part of water table regulation measures required for developments remote from the peatland itself. A second category consists of those activities which do not actually exploit the peatland but nevertheless affect it through physical disturbance. This category includes linear features such as roads, pipelines, and cut-lines running through peatlands. A third category of activities which have an impact on peatlands involves contamination by pollutants, from near or distant sources, such as acid deposition, heavy metal deposition, and dust-fall.

The environmental implications of developing peat resources for various uses are not well understood. A number of reports reviewing potential problems in peatland development have been produced in recent years. However, most of these have been oriented towards the use of peat as fuel for electric power generation. Information about the environmental effects of other kinds of peatland uses must be gleaned from a large volume of world literature, only a part of which is relevant to problems specific to Alberta. It is clear that to adequately evaluate the impacts from development or external influences, the function of peatland ecosystems should first be understood. Thus, this review outlines the nature, properties, and functions of peatlands, examines the kinds and extent of activities affecting peatlands in Alberta, and reviews the actual hydrochemical and hydrological impacts which could occur based on literature from other provinces and countries.

In summary, the specific objectives of this report are to describe, based upon a review and analysis of pertinent world and local literature:

1. The present state of knowledge about waters associated with peatlands in Alberta.
2. The kinds of anthropogenic activities which may have an impact on waters associated with peatlands in Alberta.
3. The potential impact of anthropogenic activities on water associated with peatlands.
4. The significant deficiencies in information and knowledge with regard to Alberta peatlands, and recommendations for further action, if required.

2. TYPES, DISTRIBUTION, AND EXTENT OF PEATLANDS IN ALBERTA

L.W. Turchenek

2.1 INTRODUCTION

Peatlands are abundant in Alberta, accounting for almost one-fifth of the total area of the province. They occur in the central and northern portions of the province, and are highly variable in their properties, ranging from shallow deposits of peat associated with sloughs and potholes in the parkland regions to deep deposits with permafrost in the subarctic regions of the north. This chapter reviews the general nature and properties of peatlands in Alberta, summarizes the peatland inventory information currently available about these resources, and introduces concepts and conventions applied in description and inventory of peatlands. The information presented is complemented by a glossary (Chapter 14) which provides definitions for specific terms.

2.2 TERMINOLOGY AND DEFINITIONS

Peatlands are unbalanced systems in which the rate of production of organic materials by living organisms exceeds the rate of decomposition, usually under conditions of almost continuous saturation by water. The result is an accumulation of an organic deposit which is termed 'peat'. As a peat deposit thickens, the surface vegetation becomes isolated from underlying soils and rocks, and the resulting environmental changes are often accompanied by floristic changes that reflect the altered hydrology and chemistry of the peat surface (Moore and Bellamy 1974). The chemical composition of peat is influenced by the nature of the plants from which it has originated and by the moisture conditions during and following its formation and accumulation.

Various definitions for peat can be found in the literature with variations arising due to origins in different scientific and applied disciplines. For purposes of forest science technology, Ford-Robertson (1971) defined peat as unconsolidated material consisting largely of undecomposed, or only slightly decomposed, organic matter accumulated under conditions of excessive moisture. The definition of the Canadian Society of Soil Science (1976) is similar except that peat is referred to as a soil material. Definitions developed by the International Peat Society, as reported by Stanek and Worley (1983), represent an attempt to standardize certain peat and peatland terms internationally. The definition of peat from this source is as follows:

> **Peat** - material constituting peatlands, exclusive of the live plant cover, consisting largely of organic residues accumulated as a result of incomplete decomposition of dead plant constituents under conditions of excessive moisture (submergence in water and/or waterlogging).

Peat may contain a variable proportion of transported mineral material; it may form in both base-poor and base-rich conditions; it can be either autochthonous peat or allochthonous peat; and it may contain basal layers of coprogenic elements and comminuted plant remains (such as gyttja) or humus gels (such as dy). The physical and chemical properties of peat are influenced by the nature of the plants from which it has originated, by the moisture relations during and following its formation and accumulation, by geomorphological position, and by climatic factors. The moisture content of peat is usually high; the maximum water-holding capacity occurs in *Sphagnum* peat, being over 10 times its dry weight and over 95% of its volume. Most peats have a high organic content (85% and more). However, in general, peat must have an organic matter content of not less than 30% of the dry weight (Stanek and Worley 1983).

The criteria for organic matter content in the definition above (i.e., more than 30% organic matter) are the same as for definitions of Organic soils or Organic soil horizons in the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987), and for the Histosol order in the system of the US Department of Agriculture Soil Conservation Service (1975). The description of Organic soil from Stanek and Worley (1983) is as follows:

> **Organic soil** - term used in the Canadian and U.S. soil classifications for soils that have developed primarily from plant remains and contain >30% of organic matter (17% or more carbon derived from the organic matter). They are classified on the basis of decomposition, type of plant fibers, and presence of mineral, frozen, water or rock layers.

Jeglum et al. (1974) reviewed a considerable amount of literature in selecting terms and concepts that were pertinent and useful to a proposed framework for wetland classification in Ontario. Peatland, organic terrain, wetland, and mire were considered to be broadly similar terms. Wetland is a general term used to designate any poorly drained, uncultivated tract whatever its vegetation cover or soil (Ford-Robertson 1971). Mire is a term roughly equivalent to wetland and is used more commonly in Great Britain and Europe. Wetland is the more common term in Canada and is defined by Zoltai et al. (1975), Tarnocai (1980), and Stanek and Worley (1983) as follows:

> **Wetland** - land having the water table at, near or above the land surface or which is saturated for long enough periods to promote wetland or aquatic processes as indicated by hydric soils, hydrophytic vegetation and various kinds of biological activity which are adapted to the wet environment. Wetlands include peatlands and areas that are influenced by excess water but which, for climatic, edaphic or biotic reasons, produce little or no peat. Shallow open water, generally less than two metres deep, is also included in wetlands.

The concept of wetland includes peatlands, wet mineral soils (i.e., gleysols), shallow open waters less than 2 m deep, and areas modified by water-control structures or by cultivation but which, if allowed to revert, become saturated for long periods and are associated with wet soils and hydrophytic vegetation. Peatland is the commonly used term for terrain or landscapes with extensive peat deposits. Almost all peatlands can be considered as wetlands, an exception being a group of Organic soils called Folisols, which develop mainly in coastal areas through accumulation of forest residues in non-saturated conditions. The definition for peatland from the National Wetlands Working Group (1988) is as follows:

> **Peatland** - a generic term including all types of peat-covered terrain.

Depth of peat, with 20 to 40 cm being a common limit, has been used as a criterion for distinguishing peatlands from mineral soils (Jeglum et al. 1974; Stanek and Worley 1983). In Canada, depth criteria for the Organic order in the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987) are generally applied. This depth is 40 cm or more where peat overlies unconsolidated materials, or 10 cm or more where it overlies rock.

Terms such as bog, swamp, muskeg, and marsh have general layman's usage as well as more restricted technical meanings (Jeglum et al. 1974). Of these, bog, marsh, and swamp are used in a restricted sense and, along with the terms 'fen' and 'shallow water', are used for the main wetland classes in the Canadian Wetland Classification System. The terms are defined in the following section where the wetland classification system used in Canada is described.

The various terms defined above are the major ones used in the body of this report. Discussion will use the specifically defined terms and will avoid those (e.g., muskeg) that have broad but less specifically defined usage. Numerous other terms will be used in this report - definitions for them are provided in the Glossary.

2.3 CLASSIFICATION OF PEATLANDS

2.3.1 Kinds of Classification Systems

Several types of classification systems have been developed for peat and peatlands. Some classifications are based on specific features such as the vegetation, peat material, or water in peatlands. Other classifications combine a number of specific features to describe a vertical column of a peat deposit. Organic soil classifications are the most completely developed of these, although classifications of peat stratigraphy have been developed in some countries. These emphasize the peat material, but vegetation and water properties can also be included. Finally, classifications based on the three-dimensional attributes of peatlands, adding landscape features to the above properties, have been developed. Some of these classifications could be regarded as incorporating a fourth dimension whereby the genesis of peatlands - namely vegetation, landscape, and climate acting over time - is considered. A number of single-factor classifications, soil/stratigraphic classifications, and a general system, the Canadian Wetland Classification System, are discussed below.

It is important not to confuse the various classification systems. They have been developed for different purposes within different scientific disciplines, and different systems describe peatlands systems at different levels of complexity. They can be used alone, as in some sections of this report, or they can be combined within the hierarchical framework of the Canadian Wetland Classification System.

2.3.2 Peat Material Classification

A framework for universal classification of peat was developed by the International Peat Society (Kivinen 1980). The scheme uses three properties for classification: botanical composition, degree of decomposition, and the trophic status (nutrient richness) of the peat.

1.	Botanical composition	- Moss peat
		- Sedge peat
		- Wood peat
2.	Decomposition degree	- Weakly decomposed
		- Moderately decomposed
		- Strongly decomposed
3.	Trophic status	- Oligotrophic
		- Mesotrophic
		- Eutrophic

This system can be applied to most types of peat, but requires further refinement and subdivision according to local conditions and use requirements. The subdivisions of the three main properties indicated above are guidelines only. The categories and terminology used differ in different countries. Classes commonly used in Canada for the three peat properties are outlined below.

2.3.2.1 Botanical composition. Categories of botanical composition used to describe peat materials have been suggested for use in Canada by Tarnocai (1984). The classes are as follows:

1. *Sphagnum* peat
2. Sedge peat
3. Brown moss-sedge peat
4. Woody sedge peat
5. Woody peat
6. Feather moss peat
7. Sedimentary peat
8. Amorphous peat

Broader categories are used in the Canadian System of Soil Classification as described by the Agriculture Canada Expert Committee on Soil Survey (1987) and by Mills et al. (1977). The categories are bog, sedge, forest, and sedimentary peat. In some parts of the country: other types of peat may occur and may be added to the list. For a more detailed categorization, the above may be combined or subdivided. For example, sedimentary peat may occur as gyttja, a nutrient-rich coprogenic sediment, or as dy, an acidic and relatively nutrient-poor deposit of humus gels. Examples of combinations of the above categories are woody *Sphagnum*, woody brown moss-sedge, amorphous woody peat, and so on. The classification can thus be adapted to different peat types occurring in different areas.

2.3.2.2 Degree of decomposition. Decomposition refers to the breakdown of plant matter accomplished biochemically through the action of microorganisms as well as mechanically through the action of wetting-drying and freeze-thaw cycles in peat (Puustjarvi 1977; Kong et al. 1980). From the point of view of the structure of peat, the results of chemical decomposition and physical disintegration are similar and difficult to distinguish. Several methods of determining the degree of decomposition have been developed. Some are practical field methods, while others are for use in a laboratory. Two systems are commonly employed in Canada. One is used in soil classification and has three categories of decomposition - weakly, medium, and strongly decomposed - corresponding to the terms fibric, mesic, and humic, respectively, as described in the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987). The criteria and terminology of the soil classification system in Canada are as follows:

> **Fibric peat** - peat consisting largely of materials that are readily identifiable as to botanical origin. Fibric material has 40% or more of rubbed fiber by volume and is classified in the von Post scale of decomposition as class 1 to class 4.

Fiber is defined as the organic material retained on a 100-mesh sieve (0.15 mm), except for wood fragments that cannot be crushed in the hand and are larger than 2 cm in the smallest dimension. Rubbed fiber is the fiber that remains after rubbing a peat sample about 10 times between the thumb and forefinger or after treatment in a blender or similar mixer.

Mesic peat - material that is at a stage of decomposition intermediate between fibric and humic materials. The material is partly altered both physically and biochemically, and does not meet the requirements of either a fibric or a humic horizon. Mesic material usually is classified in the von Post scale of decomposition as class 5 or 6.

Humic peat - material that is at an advanced stage of decomposition. It has the lowest amount of fiber, the highest bulk density, and the lowest saturated water-holding capacity of the organic materials. It is physically and chemically stable over time, unless it is drained. The rubbed fiber content is less than 10% by volume and the material usually is classified in the von Post scale of decomposition as class 7 or higher.

The second commonly used system for describing peat decomposition is the 10-point system of von Post (1924). In this system, a value of H-1 represents undecomposed peat and H-10 refers to completely amorphous material. This system is applied to wet peats, and is unsuitable for dried materials or for peats with high contents of mineral materials. The method is applied by squeezing a small amount of fresh peat in the palm of the hand. The colour of the water extract and the amount and nature of fiber remaining in the hand are indicators of the degree of decomposition. The nature of the peat and water for each category are summarized in Table 2.1.

2.3.2.3 Trophic status. A threefold classification of nutrient richness of peats - oligotrophic, mesotrophic, eutrophic - is widely recognized. These terms have commonly been used as descriptors of nutrient status in aquatic systems. With respect to peat materials, the terms are used in the sense of capacity of peat to supply plant nutrients to the growing vegetation, or simply as the content of nutrients as determined by analytical procedures. Parameters that could be used include total contents of nutrients, pH, base cation

Table 2.1. The von Post scale of decomposition for peats.[a]

	Decomposition Degree, H	Nature of Squeezed Water	Amount of Peat Extruded Between Fingers	Nature of Residue After Squeezing
	1	clear, colourless	none	unaltered, fibrous
Fibric	2	clear, yellow brown	none	almost unaltered
	3	turbid, slight brown	none	slightly altered, plant remains distinct
	4	turbid, brown	almost none	somewhat mushy, plant remains easily identifiable
Mesic	5	very turbid, dark	very little	very mushy, plant remains difficult to identify
	6	muddy, dark	about 1/3	strongly mushy, plant remains indistinct, scarcely identifiable
	7	very muddy, dark	about 1/2	very soupy, plant remains scarcely identifiable
Humic	8	little free water, very dark and muddy	about 2/3	very soupy, very few identifiable plant remains
	9	no free water	almost all	homogeneous, little or no plant remains
	10	no free water	all	completely amorphous, no plant remains

[a] Adapted from Agriculture Canada Expert Committee on Soil Survey (1987), Crum (1988) and Puustjarvi (1977).

content, and degree of base saturation. The exact limits between the three groups (oligotrophic, mesotrophic, eutrophic) may vary in different regions and according to the analytical methods used (e.g., Pyavchenko 1979; Daniels 1979).

At a broad level of characterization, pH has been found to be strongly correlated with the overall availability of plant nutrients (Jeglum et al. 1974). Both pH of groundwater and pH of surface peat layers have been used as indices relating nutrient status to types of vegetation growing in peatlands (Jeglum 1971). Jeglum's (1971) classification of the nutrient-pH gradient and ranges of pH used are shown in Table 2.2. The moist peat pH values have been found to average 0.5 units lower than water pH values and thus, the limits of the criteria differ for water and peat in Table 2.2.

Nutrient classes have more recently been based on the total calcium content of the surface 50 cm of peat (Zoltai and Johnson 1987). Calcium was the preferred index element because it is the most abundant inorganic constituent in peat and because it influences acidity and nutrient mobility in the soil. The nutrient classes and calcium ranges suggested by Zoltai and Johnson (1987) are as follows:

1. Oligotrophic $< 5\ 000$ mg kg^{-1};
2. Dystrophic $5\ 000 - 10\ 000$ mg kg^{-1};
3. Mesotrophic $10\ 000 - 30\ 000$ mg kg^{-1}; and
4. Macrotrophic $>30\ 000$ mg kg^{-1}.

Table 2.2. Classification of the nutrient-pH gradient.[a]

Nutrient Status	pH of Water	pH of Peat
Very oligotrophic	<4.4	<3.9
Oligotrophic	4.5 to 5.4	4.0 to 4.9
Mesotrophic	5.5 to 6.4	5.0 to 5.9
Eutrophic	6.5 to 7.4	6.5 to 6.9
Very eutrophic	>7.5	>7.0

[a] Source: Jeglum (1971).

The classification of the nutrient-pH gradient of Jeglum (1971) partially corresponds to the reactions classes for peat in the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987). There are two categories- euic, having pH($CaCl_2$) >4.5, and dysic, having pH($CaCl_2$) <4.5. Determination of pH in 0.01 M $CaCl_2$ provides a value that is usually about 0.5 units lower than that determined in moist or saturated peat. Hence, pH($CaCl_2$) of 4.5 is equivalent to about pH 5 in Jeglum's (1971) classification and the terms oligotrophic and dysic are essentially the same. The euic category in the soil classification system is not subdivided, however, and there are no classes equivalent to mesotrophic and eutrophic.

2.3.3 Peat Water Classification

2.3.3.1 Peat water source. Broad categories of water source and water chemistry of peatlands are widely used to describe peatlands. Two terms are commonly used:

Ombrotrophic - receiving ionically poor water through precipitation

Minerotrophic - receiving ionically poor to rich water from runoff and groundwater sources

Note that Ingram (1983) and other workers use the terms 'meteoric' and 'telluric' which are generally equivalent to 'ombrotrophic' and 'minerotrophic' respectively.

Water source combined with basin configuration is the basis of a classification summarized by Damman (1986) and the National Wetlands Working Group (1988) as follows:

Limnogenous - water from ponds or streams and precipitation

Topogenous - water from local runoff, groundwater, and precipitation which collects in depressions

Ombrogenous - water predominantly from precipitation

Soligenous - water from regional overland drainage, groundwater discharge, and precipitation

2.3.3.2 Peat water chemistry. It has long been recognized that the hydrochemistry in different peatlands can be distinctly different and that it could be related to the source of water as well as to the vegetation. Terminology for trophic status of peat materials introduced in the section above applies to water chemistry as well. The classification for both peat materials and peat water was presented in Table 2.2.

Peatland systems are commonly distinguished by reference to their trophic status based on water chemistry. The terms ombrotrophic bog, very poor fen, intermediate fen, and rich fen are based on criteria more or less corresponding to the water chemistry categories presented in Table 2.2. Terms such as these are used in the following chapter in describing the vegetation and environmental gradients in peatlands.

2.3.4 Soil and Stratigraphic Classification

Soils of peatlands, or Organic soils, are classified according to the Canadian System of Soil Classification (Agriculture Canada Expert Committee on Soil Survey 1987). At the highest level of abstraction, the Order, organic soils are defined as containing 17% or more organic carbon (30% organic matter) to certain depths. At the second level of abstraction, the Great Group, organic soils are differentiated mainly on the basis of the degree of decomposition of the organic material in certain tiers (layers) or combinations of tiers. There are four Organic soil Great Groups: Fibrisol, Mesisol, Humisol, and Folisol. The first three are based on predominance in the 40 to 120 cm layer of fibric, mesic, and humic peat, respectively. Folisols, which occur mainly in coastal regions, are well drained Organic soils developed from forest materials of upland origin. At the third level of abstraction, the Subgroup, the soils are differentiated on the basis of the kinds and sequences of horizons. The next level in the system, the Family, is based on characteristics of the surface tier, reaction, soil climate, particle size of the terric layer, limno materials, and depth to lithic contact. At the Series level of the classification, a number of properties can be used to differentiate soils: e.g., botanical origin of parent material, abundance of logs and stumps, calcareousness, bulk density, mineralogy of terric or cumulo layers, and others. An outline of the soil classification system is presented in the Glossary under the heading 'Organic soil order'.

Stratigraphy of peat refers to the composition and sequence of materials in a deposit. The stratigraphy provides a record of peatland genesis and evolution, reflecting variations in botanical origin and in conditions for peat decomposition. For descriptive purposes, classifications based on stratigraphy have been developed, for example, by Olson et al. (1979) in Minnesota. Such classifications are based on peat material properties, sequences of the materials, and on depth. Soil classification systems such as those used in Canada and the United States incorporate elements of peat stratigraphy, but they are inadequate for describing the total peat column because they are based on only the top 1.60 m of the peat column. A stratigraphic classification for general application in description and mapping has not been developed in Canada. In the following chapter, some specific stratigraphic studies are discussed in relation to genesis of peatlands.

2.3.5 Classification of Peatland Vegetation

A general physiognomic classification of peatland vegetation has been developed for use within the Canadian Wetland Classification System as described in the following section. More detailed classifications based on dominance types and floristic composition of peatlands have been developed (e.g., Jeglum et al. 1974; Pollet 1972; Zoltai and Johnson 1987). A general cover type classification based on dominant species was developed for mapping peatlands by remote sensing methods by Turchenek and Palylyk (1990). The classification of Zoltai and Johnson (1987) was developed from investigation of peatlands in the Prairie Provinces. It is suitable for application within the Canadian Wetland Classification System as described in the following section. The classes are outlined in Table 2.3.

2.3.6 Canadian Wetland Classification System

The Canadian Wetland Classification System was devised for classification of both organic and mineral wetlands for application across the country. The classification system was originally developed by a Subcommittee of the National Committee on Forest Land Use established in 1970 (Zoltai et al. 1975). It is a hierarchical system based on ecological parameters that influence the growth and development of wetlands. These parameters are both biotic (flora, fauna, peat) and abiotic (hydrology, water quality, basin morphology, climate, bedrock, soil). The criteria for the classification were that it be compatible with the biophysical approach to land classification, that it include criteria that were important to the greatest number of disciplines, and that it be readily applicable to air-photo interpretation with only minimum ground checking.

The classification system is outlined in the Canadian Wetland Registry (Tarnocai 1980), and has more recently been described by the National Wetlands Working Group (1987, 1988). The system of classification is carried out at three levels. At the highest level, wetland class, the wetlands are classified according to their genesis. At the second level, wetland form, they are classified according to their surface morphology, surface pattern, morphology of the underlying mineral terrain, hydrology, and the type of water. At the third level, wetland type, the wetlands are classified according to the general physiognomy of the vegetation cover.

2.3.6.1 Wetland class. Wetlands are classified according to their genesis at this level, and are characterized by considerable integrity regarding surface morphology, soil type, nutrient and moisture regimes, drainage regime, and vegetation cover (Zoltai et al. 1975). Of the five categories at the wetland class level

Table 2.3. Peatland vegetation classes and characteristic plant species.[a]

Peatland Class	Characteristic Plant Species[b]
OLIGOTROPHIC TREED BOG ("dry form")	*Picea mariana/Ledum groenlandicum/Sphagnum fuscum-Pleurozium schreberi*
("wet form")	*Picea mariana/Ledum groenlandicum-Chamaedaphne calyculata/Sphagnum angustifolium-S. magellanicum*
OLIGOTROPHIC OPEN FEN	*Carex limosa-Scheuchzeria palustris/Sphagnum jensenii-S. majus-S. riparium*
OLIGOTROPHIC SHRUBBY AND TREED FENS (too few sites to adequately characterize these classes)	
DYSTROPHIC OPEN FEN	*Carex lasiocarpa-C. chordorrhiza/Campylium stellatum*[c]
DYSTROPHIC SHRUBBY FEN	*Chamaedaphne calyculata-Andromeda polifolia/Sphagnum angustifolium-S. magellanicum*
DYSTROPHIC TREED FEN	*Larix laricina-Picea mariana/Chamaedaphne calyculata/Sphagnum angustifolium-S. magellanicum*
MESOTROPHIC OPEN FEN ("dry form")	*Carex aquatilis-C. diandra/Drepanocladus aduncus-D. revolvens*
("wet form")	*Carex chordorrhiza-C. lasiocarpa-C. limosa/Scorpidium scorpioides*
MESOTROPHIC SHRUBBY FEN	*Betula pumila/Carex lasiocarpa-C. aquatilis-C. diandra/Tomenthypnum nitens-Drepanocladus aduncus-Campylium stellatum*

continued . . .

Table 2.3. Continued.

Peatland Class	Characteristic Plant Species[b]
MESOTROPHIC TREED FEN (*Sphagnum* cushion type)	*Larix laricina-Picea mariana/Betula pumila-Ledum groenlandicum/Sphagnum warnstrorfii-S. fuscum-S. angustifolium-S. magellanicum*
(*Tomenthypnum/Sphagnum* type)	*Larix laricina-Picea mariana/Betula pumila/Tomenthypnum nitens-Sphagnum warnstorfii*
(*Tomenthypnum* type)	*Larix laricina/Betula pumila/Carex lasiocarpa/Tomenthypnum nitens*
MESOTROPHIC THICKET SWAMP	*tall Salix spp.*[d]*/Carex aquatilis-Calamagrostis inexpansa/Drepanocladus aduncus-Climacium dendroides*
MESOTROPHIC HARDWOOD SWAMP ("birch type") (one site only)	*Betula papyrifera/Salix bebbiana/Calamagrostis canadensis/Climacium dendroides*
("black ash type") (SE Manitoba only)	*Fraxinum nigra/Matteuccia struthiopteris/Plagiomnium ellipticum-Climacium dendroides*
MESOTROPHIC CONIFER SWAMP ("black spruce type")	*Picea mariana/Ledum groenlandicum/Pleurozium schreberi-Hylocomium splendens*
("tamarack type")	*Larix laricina-Picea mariana/Ledum groenlandicum-Betula pumila/Carex leptalea-C. disperma*
MACROTROPHIC OPEN FEN[e]	*Carex chordorrhiza-C. limosa-C. lasiocarpa/Scorpidium scorpioides*
MACROTROPHIC SHRUBBY FEN	*Betula pumila/Tomenthypnum nitens-Drepanocladus revolvens*

continued . . .

Table 2.3. Concluded.

Peatland Class	Characteristic Plant Species[b]
MACROTROPHIC TREED FEN	*Larix laricina-Picea mariana/Betula pumila/Muhlenbergia glomerata/Tomenthypnum nitens*

[a] Source: Zoltai and Johnson (1987).
[b] Species within a given stratum are listed in order of decreasing abundance.
[c] Differentiating species *Sphagnum subsecundum sensu stricto*.
[d] The most common species are: *Salix bebbiana*, *S. lucida*, *S. planifolia*, and *S. pyrifolia*.
[e] Differentiating species from Mesotrophic Open Fen are *Catoscopium nigritum* and *Cinclidium stygium*.

(bog, fen, swamp, marsh, and shallow water), the first three pertain to peatlands. Following are the definitions for all the wetland classes taken from Tarnocai (1980) and from the National Wetlands Working Group (1987, 1988).

Bog - a peatland with a high water table, generally at or near the surface. The bog surface is often raised, or level with the surrounding wetlands, and is virtually unaffected by the nutrient-rich ground waters from the surrounding mineral soils. Hence, the groundwater of the bog is generally acid and low in nutrients. The dominant peat materials are *Sphagnum* and forest peat underlain in some places by fen peat. The associated soils are Fibrisols, Mesisols and Organic Cryosols. The bogs may be treed with black spruce, or treeless, and they are usually covered with *Sphagnum* mosses, feather mosses, and Ericaceous shrubs.

Fen - a fen is a peatland with a high water table that is usually at or above its surface. The waters are mainly nutrient-rich, minerotrophic waters from mineral soils. The dominant peat materials are shallow to deep, well to moderately decomposed sedge and/or brown moss peat. The associated soils are Mesisols, Humisols and Organic Cryosols. The vegetation consists predominantly of sedges, grasses, reeds and brown mosses with some shrub cover and, occasionally, a scanty tree layer.

Marsh - a marsh is a mineral or a peat-filled wetland that is periodically inundated by standing or slowly moving, nutrient-rich waters. Surface water levels may fluctuate seasonally, with declining levels exposing drawdown zones of matted vegetation or mud flats. The substratum usually consists dominantly of mineral material, although occasionally it consists of well decomposed peat deposits. The associated soils are predominantly Gleysols with some Humisols and Mesisols. Marshes characteristically show zonal or mosaic surface patterns composed of pools or channels interspersed with clumps of emergent sedges, grasses, rushes and reeds, bordering grassy meadows and peripheral bands of shrubs or trees. Where open water occurs, a variety of submerged and floating aquatic plants flourish.

Swamp - a swamp is a mineral wetland or a peatland with standing or gently flowing waters in pools and channels. The water table is usually at or near the surface. There is strong water movement from margins or other mineral sources; hence the waters are rich in nutrients. If peat is present, it is mainly well decomposed wood peat underlain at times by sedge peat. The associated soils are Mesisols, Humisols, and Gleysols. The vegetation is characterized by a dense cover of coniferous or deciduous trees or shrubs, herbs, and some mosses.

Shallow water - shallow water is intermittent or permanent, standing or flowing water with relatively large and stable expanses of open water, which are locally known as ponds, pools, sloughs, shallow lakes, bays, lagoons, oxbows, impoundments, reaches, or channels. Shallow waters are distinguished from deep waters by mid-summer water depths of less than two metres, and from other wetlands by summer open water zones occupying 75% or more of the wetland surface area.

2.3.6.2 Wetland form. A large range of peatland forms are found in Canada because of the wide variations in climate and physiography. Subdivision of wetland classes into wetland forms is based mainly on surface morphology of wetlands, including the distribution of surface waters, and the morphology of the underlying mineral terrain and enclosing basin (Zoltai and Pollett 1983). Some types of wetland forms are based on hydrotopographic features, such as lakes or rivers, with which they are associated. Descriptions and keys for identifying the various wetland forms have been developed by the National Wetlands Working Group (1987, 1988). All of the forms cannot be found in any one region of the country since a variety of peatland types has developed as a consequence of the varied climatic regimes in the country. Descriptions according to the National Wetlands Working Group (1987, 1988) of those forms found in Alberta are given below.

Bog Wetland Forms

Basin bog - a bog situated in a basin that has an essentially closed drainage system, receiving water from precipitation and from runoff from the immediate surroundings. The surface of the bog is flat, but the peat is generally deepest at the centre.

Collapse scar bog - a circular or oval-shaped wet depression in a perennially frozen peatland. The collapse scar bog was once part of the perennially frozen peatland, but the permafrost thawed, causing the surface to subside. The depression is poor in nutrients, as it is not connected to the minerotrophic fens in which the palsa or peat plateau occurs.

Flat bog - a bog having a flat, featureless surface. It occurs in broad, poorly defined depressions. The depth of peat is generally uniform.

Northern plateau bog - a raised bog elevated 0.5 to 1 m above the surrounding fen. The surface is generally even, characterized only by small wet depressions. The plateau bog is usually teardrop-shaped, with the pointed end oriented in the downslope direction.

Peat plateau bog - a bog composed of perennially frozen peat, rising abruptly about 1 m from the surrounding unfrozen fen. The surface is relatively flat and even, and often covers very large areas. The peat was originally deposited in a non-permafrost environment and is often associated with collapse scar bogs or fens.

Veneer bog - a bog occurring on gently sloping terrain underlain by generally discontinuous permafrost. Although drainage is predominantly below the surface, overland flow occurs in poorly defined drainage-ways during peak runoff. Peat thickness is usually less than 1.5 m.

Fen Wetland Forms

Channel fen - a fen occurring in a topographically well-defined channel which does not contain a continuously flowing stream. The depth of peat is usually uniform.

Collapse scar fen - a fen with circular or oval depressions, up to 100 m in diameter, occurring in larger fens, marking the subsidence of thawed permafrost peatlands. Dead trees, remnants of the subsided vegetation of permafrost peatlands, are often evident.

Floating fen - a fen occurring adjacent to ponds or lakes, forming a floating mat, underlain by water or fluid, loose peat. The fen surface is less than 0.5 m above the level of the lake and the rooting zone is affected by lake water.

Horizontal fen - a fen with a very gently sloping, featureless surface. This fen occupies broad, often ill-defined depressions, and may be interconnected with other fens. Peat accumulation is generally uniform.

Net fen - a fen with a broad net pattern of low, interconnected peat ridges (strings), enclosing wet hollows or shallow pools. The wetland surface is almost completely level; greater slopes result in the formation of northern ribbed fens.

Northern ribbed fen - a fen with parallel, low peat ridges (strings) alternating with wet hollows or shallow pools, oriented across the major slope at right angles to water movement. The depth of peat exceeds 1 m.

Shore fen - a fen with an anchored surface mat that forms the shore of a pond or lake. The rooting zone is affected by the water of the lake at both normal and flood levels.

Slope fen - a fen occurring mainly on slowly draining, nutrient-enriched seepage slopes. Pools are usually absent, but wet seepage tracks may occur. Peat thickness seldom exceeds 2 m.

Spring fen - a fen nourished by a continuous discharge of groundwater. The surface is marked by pools, drainage tracks, and, occasionally, somewhat elevated 'islands'. The nutrient level of water is highly variable between locations.

Stream fen - a fen located in the main channel or along the banks of permanent or semipermanent streams. This fen is affected by the water of the stream at normal and flood stages.

Swamp Wetland Forms

Basin swamp - a swamp developed in a topographically defined basin where the water is derived locally but may be augmented by drainage from other parts of the watershed. Accumulation of well decomposed peat is shallow (less than 0.5 m) at the edge, and may reach 2 m at the centre.

Flat swamp - a swamp occurring in broad areas of poorly drained lowlands. The outer edges of the swamp usually merge gradually into the upland, without sharp boundaries. Peat build-up is generally thin (less than 0.5 m), but may exceed 2 m.

Floodplain swamp - a swamp occurring in a valley which may be inundated by a seasonally flooding river. Slow draw down after flooding preserves a high water table for most of the growing season. Shallow peat development may be encountered.

Peat margin swamp - a swamp occurring in a relatively narrow (up to 25 m wide) zone between the mineral uplands and the peatland. The high water table is maintained by the peatland, but drainage from the upland adds nutrient-enriched water to the swamp. Peat deposition (less than 1 m) is common.

Shore swamp - a swamp occurring along the shores of permanent ponds or lakes. The high water table is maintained by the water level in the lakes, but seasonal flooding may take place. Peat development is possible.

Spring swamp - a swamp nourished by the discharge of groundwater. The surface is characterized by low hummocks, small pools, and drainage tracks. The amount of dissolved solids in the spring water varies regionally.

Stream swamp - a swamp occurring along the banks of permanent or semipermanent streams. The high water table is maintained by the level of water in the stream. The swamp is seasonally inundated, with subsequent sediment deposition.

2.3.6.3 Wetland type. The terms used to describe wetland types are based on the general physiognomy of the plant cover. There are 16 categories developed for this classification system as indicated by Tarnocai (1980): (1) treed (coniferous and hardwood); (2) shrub (tall, low and mixed); (3) forb; (4) graminoid (grass, reed, tall rush, low rush, and sedge); (5) moss; (6) lichen; (7) aquatic (floating and submerged); (8) non-vegetated.

Other vegetation classifications systems may be used instead of the above. The classification of Zoltai and Johnson (1987) was developed for peatlands in Alberta and the other prairie provinces. These classes were presented in Table 2.3, Section 2.3.5.

2.3.6.4 Wetland variety. Wetland variety is a fourth, more detailed level of classification intended to recognize the special needs of various disciplines such as forestry, engineering and others (Zoltai and Pollett 1983). At this level, smaller, more homogeneous portions of peatlands can be delineated and described with greater accuracy, according to the interest of the investigator. Floristic or phytosociological units can be determined, or differentiation may be based on hydrological characteristics, soil type, and various other peatland properties as described in the preceding sections. Delineations may also be made on the basis of land use interpretations such as forest productivity, agricultural potential or drainability (Zoltai and Pollett 1983; Zoltai et al. 1975).

2.4 SOURCES OF INFORMATION ABOUT PEATLANDS IN ALBERTA

2.4.1 Forest Inventory

2.4.1.1 Broad inventory (Phase I). Forest cover is displayed on maps with information about density, height, and species. The minimum stand size is 65 ha (160 a). Maps originally produced at a scale of 1:63 360 or 1:126 720 have been revised to 1:100 000. Organic terrain is classified and mapped as either 'treed muskeg' or 'marsh bog or muskeg'. The 'marsh bog or muskeg' category probably represents an open fen type of peatland. The 'treed muskeg' category may be either bog or fen. In addition, peatlands may be represented by various black spruce and tamarack cover types. Other than the general cover type, little information about peatland properties can be obtained from these maps.

2.4.1.2 Detailed inventory (Phase II). This inventory covers only part of the province and is similar to broad inventory except that the minimum stand size is 16 ha (40 a). The map scale is 1:31 680. The comments in the previous section regarding peatland information which can be derived also apply to Phase II inventory.

2.4.1.3 Detailed inventory (Phase III). The Phase III inventory provides detailed information about forest density, height, species, age, and site quality at a scale of 1:15 000. The minimum stand size is about 8 ha (20 a). As in Phase I and II maps, symbols are used to designate peatlands. Three categories generally represent peatlands, namely 'coniferous scrub', 'open muskeg', and 'treed muskeg'. Three other categories, 'deciduous scrub', 'brush', and 'burn' may or may not represent peatlands. In addition, black spruce, tamarack, and black spruce-tamarack cover types can occur on peatlands. The Phase III maps do not provide information equivalent to the peatland class level of the classification (i.e., categories of fen, bog, or swamp can only be inferred from the cover types). They do, however, provide accurate information about peatland locations, areas, and boundaries. In addition, forest cover descriptions for some of the peatlands provide some information about vegetation while little if any is provided by most of the other inventories (discussed below).

2.4.2 Soil Survey

Organic soils are classified according to the Canadian System of Soil Classification as described in Section 2.3.4. Most of northern Alberta was surveyed at an exploratory level (scale about 1:750 000) in the 1950s and 1960s. In these surveys, organic soils were indicated as percentages of broad soil areas and were not otherwise differentiated. Reconnaissance-level soil surveys have been conducted for much of central and northwest Alberta (generally at a scale of 1:126 732 and 1:190 080). Peatlands are characterized by high spatial variability of soils. In reconnaissance surveys prior to 1950, there was no separation of peatland soil types, and peatland areas were simply labelled as 'Organic' soils. In the later reconnaissance surveys, organic soils were mapped as complexes in many areas. In most of these surveys, two mapping units were used: the name 'Kenzie' was applied to organic soils derived from *Sphagnum* mosses, and the name 'Eaglesham' was used to designate those derived from sedges. The name 'Fickle' has been used to designate organic soil complexes in the foothills areas (Dumanski et al. 1972). In addition to Kenzie and Eaglesham complexes, the names 'Dizzy' and 'Mikkwa' were used for mapping organic soils with permafrost in northern Alberta (Scheelar and Macyk 1972; Turchenek and Lindsay 1982). Mapping of peat soils in greater detail began with the soil survey of the Sand River sheet (NTS sheet 73L) in east-central Alberta where 10 units were devised to describe different combinations of soils (Kocaoglu 1975). Current soil mapping is carried out at a semi-detailed level (scale of 1:50 000), and organic soil mapping units are usually based on those established in the Sand River sheet.

In addition to information about areas and extent of peatlands indicated on soil maps, most soil survey reports provide some analytical data about peat soils. This is the only source of information about chemical and physical properties of peats in Alberta other than information from site-specific studies published in various journals. The amount of information thus provided is nevertheless scant compared to the volume of data available for mineral soils.

2.4.3 Special Geomorphic (Landform) Maps

The special geomorphic (landform) maps were prepared by the Canada Land Inventory and the Alberta Land Inventory as a physical resource data base for rating forest productivity of land. Parent geological material (including organic material) and topography are displayed on 1:63 360 black-line maps. Organic terrain is mapped as 'Organic' or as 'Veneer Organic', the latter referring to shallow peatlands and to wet mineral soils. Thus, these maps provide information about areal extent of peatlands and little or no information about their properties. The map coverage is for much of northern Alberta and the mapping concept is applied uniformly throughout.

2.4.4 Physical Land Classification Maps

These maps, prepared as a physical resource data base for resource planning and management agencies in the Department of Forestry, Lands and Wildlife and other government and research organizations, display the following features: parent material, surface expression, slope, soil classification, and drainage. Organic soils are differentiated on the maps and information about degree of decomposition and soil classification is provided in accompanying reports. Maps for different project areas vary in scale from 1:15 000 to 1:250 000, and the level of detail of information about peatlands varies accordingly. To date, there is only partial coverage of the province with these maps.

2.4.5 Ecological Land Classification and Evaluation Reports

Information in these maps and reports is an integration of landforms, soils, climate, vegetation, and drainage. In the older reports, called 'Biophysical analysis and evaluation of capability' reports, areal extent of organic soils is displayed but little more information, other than ratings of capability for certain uses, is provided.

In more recent projects, land district delineations are based on dominant vegetation, landform, drainage class, slope, and soils. Peatlands are segregated into fen and bog types, and these may occur in complexes with each other and with mineral landforms. Thus, several Organic soil mapping units are possible. Because several ecological components are described, the level of information presented is considerably greater than that of other peatland information sources. Maps are produced at a scale of 1:100 000, and are available mainly for the agricultural-forest contact areas. Thus, their value is presently limited due to partial provincial coverage.

2.4.6 Surficial Geology Maps

A few surficial geology maps of scale 1:250 000 for northern Alberta display large peatland areas but no additional information about their properties is provided. Thus, these maps can only be used to determine general location and aerial extent of peatlands.

2.4.7 Summary of Peatland Information Sources

From the preceding paragraphs, it is evident that peatlands information for the province varies in level of detail and in coverage. Table 2.4 summarizes the sources of peatland inventory information presently available, the scales of maps, the provincial coverage, and the apparent levels of detail.

2.5 EXTENT OF PEATLANDS IN ALBERTA

An early estimate of the area of peatlands in Alberta made by the Alberta Soil Survey (1967) was about 10×10^6 ha, or about 15% of the province's area. More recent estimates based on soil survey data indicate that the area is 12.67×10^6 ha, accounting for 20% of the land area or about 19% of the total (land and water) area of the province (Tarnocai 1983, 1984). This area accounts for 12% of the total Canadian peatland area. The volume of peat in Alberta is estimated to be $316\ 822 \times 10^6$ m^3 and its dry weight to be $36\ 118 \times 10^6$ t (Tarnocai 1983, 1984).

Holowaychuk and Fessenden (1986) have produced a map of Alberta that indicates the sensitivity of different soil types to acid deposition. The area of soils of the Organic order as derived from their map is 10.5×10^6 ha. In addition, there are 4.4×10^6 ha

Table 2.4. Summary of peatland inventory information available in Alberta.

Type of Inventory	Map Scale	Coverage	Level of Detail
Wetlands Regions of Canada	1:7 500 000	Map of Canada	Low
Peatland Distribution in Alberta	1:2 000 000	Map of Alberta	Low
Soil Surveys	1:760 320	North of 55°, except Yellow Zone	Low
	1:190 080	Mainly White Zone, Some Yellow Zone	Low
	1:126 720	Yellow Zone, Some Green Zone	Low
	Larger Scales	Very Few Areas	Medium
Special Geomorphic (Landform) Maps	1:63 360	Most of Green Zone	Low
Physical Land Classification Maps	Variable	Parts of Green Zone	Low
Ecological Land Classification and Evaluation Maps	1:100 000	Green Zone-White Zone Contact Areas	Low
Surficial Geology Maps	1:250 000	Parts of Green Zone	Low
Forest Inventory Maps - Phase I	1:126 720 to 1:63 360	Green Zone	Low
- Phase II	1:31 680	Green Zone	Low
- Phase III	1:15 000	Green Zone	Medium

of Organic Cryosolic soils, or peat soils with permafrost. The total of these is 14.9 x 10^6 ha of peatlands in Alberta. Due to the generalized nature of this map, significant amounts of mineral soils are included in these calculations, and the figures of Tarnocai (1983, 1984) should be considered to be more reliable. However, the data from Holowaychuk and Fessenden (1986) are of interest in that they indicate that about 30% of the peatlands in Alberta are characterized by permanently frozen subsoil conditions.

2.6 MAP OF DISTRIBUTION AND TYPES OF PEATLANDS IN ALBERTA

A map of the distribution of peatlands in Alberta has been compiled by Hamilton (1975). The map outlines areas of the province in which peatlands predominate, but it provides no information about the kinds of peatlands within the mapped areas. A broad-scale map of Canada providing information about the distribution of wetlands and about zonation based on differences in wetland types has been produced by the Canada Committee on Ecological Land Classification, National Wetlands Working Group (1986). The distribution and types of wetlands in Canada have also been reviewed by Zoltai and Pollett (1983).

It was considered that for the purposes of this review, development of a more detailed map than those referred to above would be helpful for further discussion of peatland resources and human impacts on them. A map of 'Distribution of Peatlands in Alberta' was developed principally by adaptation of the report and map of Preliminary Wildlife Habitat Regions Subregions of Alberta (Pedocan Land Evaluation Ltd. 1984). The Preliminary Wildlife Habitat Regions/Subregions map is a first approximation of a classification of ecological habitat units at a scale of 1:1 000 000 for the purpose of overview evaluations of wildlife resources in the province. The map identifies 12 wildlife habitat regions which are essentially the ecoregions of Alberta identified by Strong and Leggat (1981). The subregions are subdivisions of regions based on physiography, lithology, landforms, surficial materials, and soils. The

Physiographic Map of Alberta (Pettapiece 1986) is the major source for the subregion information. The extended legend for the map of the Preliminary Wildlife Habitat Regions/Subregions of Alberta provides information about the proportion of each subregion that is occupied by aquatic forms of surface cover. Fens, bogs, and undifferentiated peatlands are included among the aquatic forms. Information is also provided about percentages of organic landforms and of organic soils.

Aspects of the maps of 'Distribution of Wetlands in Canada' and of the 'Wetland Regions of Canada' produced by the National Wetlands Working Group (1986) were also adapted for developing the wetland distribution map of Alberta. The legend for percentage cover of peatlands from these maps was adopted for the map of 'Distribution of Peatlands in Alberta'. Adjustments were made for incongruencies between the national wetlands map and the provincial ecoregion map. These pertained mainly to lack of coincidence of the boundaries on the two maps (although they were fairly close for most regions) and to the larger number of ecoregions than of wetland regions mapped in Alberta. The descriptive information about wetland regions from the National Wetlands Working Group (1986) was applied, along with information from other sources, to the ecoregions depicted on the peatland distribution map of Alberta. A descriptive legend for the subdivisions of the ecoregions (i.e., the subregions) is not provided as there is insufficient inventory information for most of northern Alberta to adequately describe these areas. The resulting map and legend of peatland distribution in Alberta is provided in the back pocket of this report.

2.7 SUMMARY

Almost one-fifth of Alberta's area consists of peatlands. Peatland is a generic term applied to all types of peat-covered terrain. Peat is the accumulation of undecomposed or partially decomposed plant remains. Peat materials are generally described and classified in terms three major properties; botanical origin, nutrient status, and degree of decomposition. Peat deposits can be classified according to a soil classification system or according to a more comprehensive stratigraphic classification system. Peatland water is usually categorized in terms of origin; ombrotrophic refers to atmospherically derived water, and minerotrophic refers to water derived from runoff and groundwater. The vegetative cover of peatlands is classified using physiognomic categories or dominance types.

The Canadian Wetland Classification System is a comprehensive, hierarchical system devised for classification of wetlands throughout Canada. The main peatland classes that occur in Alberta are fens, bogs, and swamps. Subclasses of these are based on surface form, basin morphology and other characteristics. Horizontal fens, northern ribbed fens, northern plateau bogs, and peat plateau bogs (raised bogs with permafrost) are probably the most common peatland types in Alberta.

Various types of natural resource inventory data bases indicate the extent and distribution of peatlands in Alberta, but little information is available about peatland properties such as thickness of deposits, types of peat, and the chemical and physical properties of peat. Information about the general nature, extent and distribution of peatlands in Alberta was compiled on a 1:2 million scale map which accompanies this report.

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3. PEATLAND VEGETATION AND BIOGEOCHEMISTRY

B.J. Nicholson

3.1 INTRODUCTION

Peat formation can occur under a variety of vegetational types as the basic requirement for peat accumulation is a cool, wet, anaerobic environment. Under these environmental conditions organic material is not rapidly decomposed. The peat which accumulates is a stratified, in situ formation of undecomposed and slightly humified plant tissue plus amorphous humic materials. Changes to the plant material which occur during peat formation include loss of organic matter from leaching and microbial attack, loss of physical structure, and a change of chemical state (Clymo 1983).

Peatlands are highly variable in their nature and properties, and human activities will consequently affect different peatlands in different ways. The developmental processes that lead to variability among peatlands as well as within individual peat deposits are discussed in this chapter. The extreme variability which exists in peat-forming communities reflects variation in regional and local conditions of climate, nutrient availability, moisture, and shade. These factors are discussed below, followed by discussion of peat-forming materials, modes of accumulation, and successional changes occurring over time as revealed through study of peatland stratigraphy.

3.2 CONTROLS ON PEAT-FORMING VEGETATION

3.2.1 Climate

Climate affects peat-forming communities through species distribution, and can be seen along two gradients. The first is a north-south temperature influence, and the second is an oceanic versus continental moisture influence. For example, there are 21 species of the genus *Sphagnum* presently known in Alberta. These have either widespread or continental tendencies in their distribution patterns. Oceanic and suboceanic species are absent in Alberta (Vitt and Andrus 1977).

Climatic change can have a profound influence on vegetation and peat formation: A change to relatively dry conditions favours enhanced peat decomposition and a change in vegetation communities which give rise to forest peat. Climatic change towards increased wetness can upset the state of equilibrium by enhancing the water supply to surface layers thus leading to increased growth of telmatic and semi-terrestric vegetation which increases peat accumulation.

3.2.2 Nutrient Availability

Nutrient availability is a more local influence and has a major impact on peat-forming vegetation. In a given peatland the surface vegetation is insulated from the underlying mineral soil by a layer of accumulated peat. The floristic composition of a peatland is therefore dependent upon the amount and type of incoming nutrient sources. Minerotrophic peatlands are defined as peatlands which receive waters containing dissolved ions derived from mineral soils (Ca^{2+}, Mg^{2+}) through groundwater discharge or overland flow. Ombrotrophic peatlands are isolated from the surrounding local water table and theoretically receive ionically poor water through atmospheric precipitation only. Oligotrophic and eutrophic are terms used to describe nutrient status. Oligotrophic peatlands are nutrient poor, receiving low amounts of nitrogen and phosphorus. Eutrophic mires are nutrient rich receiving high amounts of nitrogen and phosphorus. The relative proportions of nitrogen, phosphorus, and dissolved ions received by marshes, swamps, fens, and bogs can be ordinated along the two nutrient axes as shown in Figure 3.1.

Bogs are ombrotrophic peatlands, receiving ionically poor water through precipitation, and are nitrogen and phosphorus deficient. They are usually formed under conditions of high precipitation and low evapotranspiration. Bogs can be differentiated from other minerotrophic mires by the predominance of *Sphagnum*

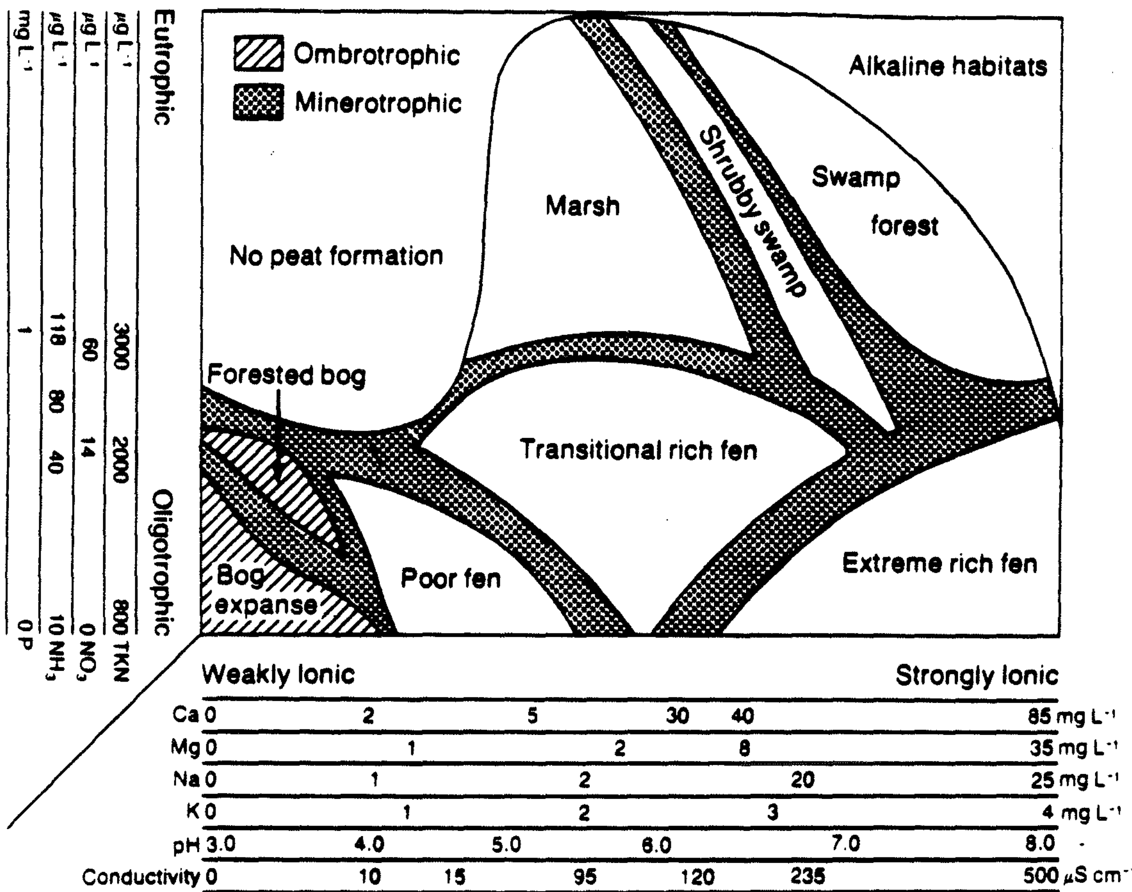

Figure 3.1. Relative macronutrients and dissolved ions received by wetlands. (Source: Vitt 1986; unpublished information).

species, low species diversity or richness, and a lack of *Carex* species (Table 3.1). Zoltai and Pollett (1983) described bogs as *Sphagnum*-dominated systems with a high cover of ericaceous shrubs. Sedges may grow on them and trees may or may not be present.

In Alberta, large ombrotrophic raised bogs do not occur in the classical European sense. Instead ombrotrophic areas develop on restricted plateaus where drainage divides have formed, or develop in complex mires where hydrological isolation has occurred due to flow diversion around upland mineral ridges. These sites are characterized by a dense surface carpet of *Sphagnum fuscum*, *S. magellanicum*, and *S. angustifolium*. Important associates are *Eriophorum vaginatum*, *Rubus chamaemorus*, and the ericaceous shrubs *Ledum groenlandicum* and *Andromeda polifolia*.

In the Canadian arctic and subarctic peatlands, permafrost conditions create hydrological isolation through the expansion and uplifting of frozen peat layers (peat plateaus) and ice cores (palsas). Under these conditions ombrotrophism occurs and these areas are colonized by *Sphagnum*. The insulating properties of *Sphagnum* moss reduce heat loss and preserve the frozen condition. Eventually the site becomes extremely dry and lichens become dominant with a sparse black spruce and shrub cover (Zoltai and Pollett 1983).

In oceanic areas of Canada, where climate is characterized by high precipitation and low evapotranspiration, raised bogs and plateau bogs do develop. Within their ombrotrophic bog plains, vegetation patterning is controlled by position of the water table and forms three structural vegetational units (Damman 1978). These units are:

1. Mudbottoms - depressions where inundation occurs for most of the growing season. This area is dominated by hepatics.
2. Lawns - depressions along the pool margins which are dominated by graminoid species.
3. Hummocks - *Sphagnum* mounds dominated by ericaceous shrubs.

Table 3.1. Water chemistry variation and vegetation characteristics in Alberta peatlands.

	Bog	Poor Fen	Transitional Rich Fen	Extreme Rich Fen
Ca $mg \cdot L^{-1}$	0-2	0-5	0-40	30-85
Mg $mg \cdot L^{-1}$	0-1	0-2	1-8	2-34
Na $mg \cdot L^{-1}$	1-2	1-2	1-20	1-20
K $mg \cdot L^{-1}$	0-1	0-1	0-3	1-4
pH	3.5-3.9	4.0-5.5	(5.5) 6.0-7.2	7.3-7.8
Reduced conductivity $\mu S \cdot cm^{-1}$	0-20	0-95	14-120	235-490
TKN $\mu g \cdot L^{-1}$	900-2000	730-1900	1600-3000	920-1700
NO_3 $\mu g \cdot L^{-1}$	3-14	0-17	0-16	0-62
NH_3 $\mu g \cdot L^{-1}$	10-47	1-37	0-118	0-77
P $mg \cdot L^{-1}$	0-1	0-1	0-1	0-1
Species diversity (no species)				
Sphagnum	3-8	4-10	2-3	0-1
brown mosses	few			many
vascular plants	few (5-10)			many (20-30)

In this situation the peat-forming community is not only determined by climate and nutrient status but is also influenced by moisture gradients.

Fens are minerotrophic and thus receive mineral-enriched water flow. The nutrient status of fens varies from nutrient poor (oligotrophic) through to nutrient rich (eutrophic). Based on water chemistry criteria, fens have been subdivided into the categories of poor fen, intermediate (or transitional rich) fen, and extreme rich fen.

Poor fens develop under conditions of impeded drainage and gentle slopes, and are often associated with drainage divides. Large poor fen peatlands have been found in Alberta, situated on the top of flat plateaus such as Thickwood Hills, Muskeg Mountain, Swan Hills (Vitt et al. 1975), and Stoney Mountain Uplands. Poor fens are the most ionically and nutrient poor of the fens. Under these extreme oligotrophic conditions, *Sphagnum* is dominant. From 4 to 10 species of *Sphagnum* can occur with higher species richness in the Bryopsida and vascular plants (Table 3.1). Zoltai (1976) used surface morphology and not chemistry to differentiate fen types and regarded fens as sedge-dominated with reed and grass associations in local pools. *Sphagnum* is usually subordinate or absent. A shrub cover of low to medium height is usually present (Zoltai and Pollett 1983).

In the Alberta poor fens, lawns are the most prevalent structural unit, and are occupied by *Sphagnum jensenii*, *S. majus*, *S. fallax*, and *S. angustifolium* with *Scheuchzeria*, carices, *Menyanthes* and *Eriophorum*. The deeper pools may contain *Drepanocladus exannulatus*. Vitt et al. (1975) found two distinct hummock associations in the poor fens of the Swan Hills, one dominated by *Picea mariana*, *Sphagnum magellanicum*, and *Ledum groenlandicum*, the second by *Betula glandulosa*, *Tomenthypnum falcifolium*, and *Aulacomnium palustre*. In the Caribou Mountains of north-central Alberta, poor fens are formed in thaw pockets which create oligotrophic pools within the permafrost layer. Extensive floating carpets of *Sphagnum riparium* and *S. jensenii*, with low mounds of *S. angustifolium* and *Carex aquatilis* can be found within these pockets (Horton et al. 1979). Poor fens can also develop as floating mats which surround acidic lakes. Such fens will form around the small sheltered lakes within the granitic shield areas of northwestern Ontario (Vitt and Bayley 1984) and Michigan (Vitt and Slack 1975). In these communities the *Sphagnum* lawns are composed of *S. majus*, *S. fallax*, *S. pulchrum*, *S. papillosum*, and *S. rubellum*, which are more oceanic species. The central pools contain *Sphagnum fallax*, *Scheuchzeria palustris*, and *Carex paupercula*. Hummocks are shrub covered with *Alnus rugosa*, *Myrica gale*, *Salix pedicellaris*, *Carex lasiocarpa*, *C. rostrata*, *C. trisperma*, and *Smilacina trifolia*.

Transitional rich (or intermediate) fens have pH values from 5.5 to about 7.0 and receive moderate nutrient inputs. In Alberta, intermediate fens are characterized by their high *Carex* cover and the appearance of *Drepanocladus vernicosus*, *D. lapponicus*, and *D. aduncus*. *Sphagnum* species are limited (Table 3.1) and are restricted to the species *S. teres*, *S. warnstorfii*, and *S. subsecundum*.

In the Fort McMurray area (D.H. Vitt, D. Horton, and N. Malmer, University of Alberta, 1986, personal communication) intermediate open fens are characterized by an abundance of *Carex chordorrhiza*, *Carex lasiocarpa*, *Drepanocladus vernicosus*, and *Tomenthypnum nitens*. Flarks contain *Drepanocladus lapponicus*. Minerotrophic forested areas contain *Larix laricina*, *Andromeda polifolia*, *Betula glandulifera*, *Tomenthypnum nitens*, and *Aulacomnium palustre* (D.H. Vitt, D. Horton, and N. Malmer, University of Alberta, 1986, personal communication). At Alsike, 90 km southwest of Edmonton, the intermediate fen vegetation consists of *Tomenthypnum nitens*, *Aulacomnium palustre*, *Drepanocladus* spp., *Carex lasiocarpa*, *Potentilla palustre*, *Equisetum fluviatile*, *Menyanthes trifoliata*, *Betula pumila* and *Salix* spp. (Karlin and Bliss 1984).

Rich fens are the most minerotrophic and ionically rich, having calcium ion concentrations greater than 10 $mg \cdot L^{-1}$. They are generally observed to be brown-moss-dominated, of the family

Amblystegiaceae. Few *Sphagnum* species are present (see Table 3.1) but species diversity is the greatest in the Bryopsida and vascular plant categories. The rich fens of Alberta tend to be located in areas of regional discharge where the ground surface locally dips below the piezometric surface resulting in calcium-rich waters due to the calcareous tills and bedrock (Prosser 1982).

In the eastern foothills of the Canadian Rocky Mountains, the rich fens are treed with *Larix laricina* and *Picea mariana*. Many of these are ribbed or net fens which are characterized by low peat ridges (strings) alternating with wet hollows or shallow pools (flarks). The most prominent shrub on the strings is *Betula glandulifera*, occurring with lesser amounts of *Salix candida*, *Salix pedicellaris*, *Andromeda polifolia*, and *Ledum groenlandicum* (Slack et al. 1980). Other string species are *Carex* spp., *Triglochin maritima*, *Muhlenbergia glomerata*, and *Galium labradoricum*. String mosses include *Tomenthypnum nitens*, *Dicranum undulatum*, *Pleurozium schreberi*, and *Aulacomnium palustre*. Flarks are dominated by *Carex limosa*, *Scorpidium scorpioides*, *Drepanocladus revolvens*, *Scirpus caespitosus*, and *Campylium stellatum*. At a minerotrophic rich fen located east of Crimson Lake, about 10 km northwest of Rocky Mountain House, predominant species are *Larix laricina*, *Betula* pumila, and *Tomenthypnum nitens* on the slightly elevated areas (Zoltai and Johnston 1984). The vegetation of the wetter depressions is dominated by *Carex chordorrhiza*, *C. lasiocarpa*, *C. limosa*, *C. livida*, and *Scorpidium scorpioides*. Scattered cover of *Equisetum fluviatile* with *Menyanthes trifoliata* and clumps of *Drepanocladus revolvens* were also found (Zoltai and Johnston 1984).

Peatland types referred to as mixed mires are characterized by vertical variation in substrate chemistry and associated vegetation. The accumulation of peat above the standing water table can isolate the upper hummocks from direct mineral influence, creating ombrotrophic conditions in a minerotrophic environment. This isolation develops because of the filtering of the water as it flows upwards into the peat, by the leaching influence of precipitation, and by the decreased ability of the peat to "pull" the water up by capillarity to the upper hummock layers as the peat accumulates further above the standing water level (Karlin and Bliss 1984).

Heatherdown and Wagner peatlands are two such mixed mires in the Edmonton area. The Heatherdown peatland is located 48 km west of Edmonton. It is slightly sloping, is fed by a calcareous spring and is highly patterned. The Wagner peatland is located 13 km west of Edmonton. It also is situated on a slope fed by calcareous springs. At both mires the flark communities consist of *Scorpidium scorpioides* and *Drepanocladus revolvens*, while *Campylium stellatum* with *Tomenthypnum nitens* and *Aulocomnium palustre* occur on the low hummocks. Upper portions of the hummocks are occupied by *Sphagnum fuscum*, *Pleurozium schreberi*, and *Hylocomium splendens* (Karlin and Bliss 1984). Mixed mires are abundant in northern Alberta, with well developed hummocks of *Sphagnum fuscum* below a dense canopy of *Picea mariana*. Minerotrophic influence is indicated by the presence of *Sphagnum warnstorfii* on the hummock sides, and brown mosses and carices in the hollows (D.H. Vitt, D.G. Horton, and N. Malmer, University of Alberta, 1986, personal communication).

Swamps are wooded wetlands (peatlands and mineral wetlands) where standing to gently flowing water occurs seasonally or persists for long periods on the surface (Zoltai and Pollett 1983). The waters are minerotrophic and are moderately acid to circumneutral (pH 7.0 to 7.4) (Schwintzer 1981). In comparison to bogs and fens where the dominant strata are the field (low shrub) and bryophyte layers, swamps have a well developed tree layer. Jeglum et al. (1974) recognized three groups of swamps in Ontario based on cover. Thicket swamps are shrub covered with *Alnus*, *Betula*, and *Cornus* spp. Hardwood swamps are predominantly of the *Fraxinus nigra - Ulmus americana* (Black ash-American elm) variety. Conifer swamps can contain the following in their tree and shrub layers: *Larix* laricina, *Larix laricina* and *Alnus rugosa*, *Picea mariana*, *Picea mariana* and *Alnus rugosa*, *Picea mariana* and feathermoss, *Picea* *mariana* and *Ledum groenlandicum*, *Picea mariana* and *Sphagnum*, and *Thuja occidentalis*.

In Canada, swamps are prevalent in the Transitional Mid-Boreal Wetland District, Low Boreal, Atlantic Boreal, Pacific Temperate, and Atlantic Oceanic Wetland regions (Zoltai and Pollet 1983). Swamps of *Salix* and *Betula* are common only in the Transitional Mid-Boreal Wetland region of Alberta, which is a small horseshoe-shaped area north of the Edmonton area. Here shrub swamps occupy depressional areas along stream beds and peat accumulation is minimal (Zoltai and Pollett 1983).

Marshes are grassy, wet mineral soil areas, periodically inundated to a depth of 2 m or less with standing or slowly moving waters (Zoltai and Pollett 1983). Marshes generally experience a brief period of desiccation termed 'drawdown'. In the Ontario system of wetland classification, Jeglum et al. (1974) recognized three marsh types. Deep marshes are characterized by stands of tall graminoids, (*Typha* spp., *Carex* spp., *Phragmites* spp.), interspersed with pools and channels. This type occurs in basins and marginal zones around lake ponds or bordering rivers and streams. Other equivalent units have been designated reed or reed swamp (Jeglum et al. 1974). The substrate produced is a mineral muck or well decomposed aquatic type of peat. Shallow marshes have tall emergents with closed canopies. Small pools of open water may exist but the surface becomes exposed during the late summer. Meadows have a closed graminoid cover of sedges and grass with little or no open water. This type is usually flooded annually ranging in duration from a few weeks to the entire summer. Soils are flooded but no significant amount of peat accumulates (Jeglum et al. 1974).

3.2.3 Moisture

Moisture regime is one of the most significant influences upon the vegetation, floristics, and productivity of wetlands. Jeglum (1973) showed that wetlands which differ physiognomically can be characterized by different ranges of depth to groundwater. A conceptual model of the relationships among wetland ecosystem types in Ontario was developed in terms of two major complex factors, namely nutrient and moisture regime (Jeglum and Cowell 1982; Jeglum 1985). The model (Figure 3.2) shows how peatland types change with changes in the two major gradients. The relationships between type of peatland and its nutrient and moisture gradients have been studied in somewhat more detail in Europe. In Finland, the trophic level and especially the ground-level attributes (i.e., hummock, intermediate, flark) have been used to help define peatland types (Eurola and Kaakinen 1979, cited by Jeglum and Cowell 1982).

The influence of moisture on peat-forming communities is also known as the hummock to hollow gradient, which involves species distribution according to height above the water table. Vitt and Slack (1984) measured the niche breadth of *Sphagnum* species in northern Minnesota peatlands and showed that the distributions differed significantly in relation to height above water table. The moisture gradient appeared particularly critical in determining habitat occupation for several species. Karlin and Bliss (1984) found that hummock-to-hollow moisture gradients were further complicated by nutrient gradients in moderately and strongly minerotrophic systems. Hollow and low hummock species varied considerably along a minerotrophic gradient whereas hummock community composition was similar in all peatlands studied.

3.2.4 Shade

Shade also influences species distribution in peatlands. Vitt and Slack (1984) found that *Sphagnum* species distribution in northern Minnesota peatlands could be grouped into three broad categories:

1. Those with no shade: *Sphagnum papillosum*, *S. rubellum*, *S. centrale*, and *S. contortum*.
2. Those with open tree cover: *Sphagnum fuscum*, *S. nemorum*, *S. magellanicum*, and *S. angustifolium*.

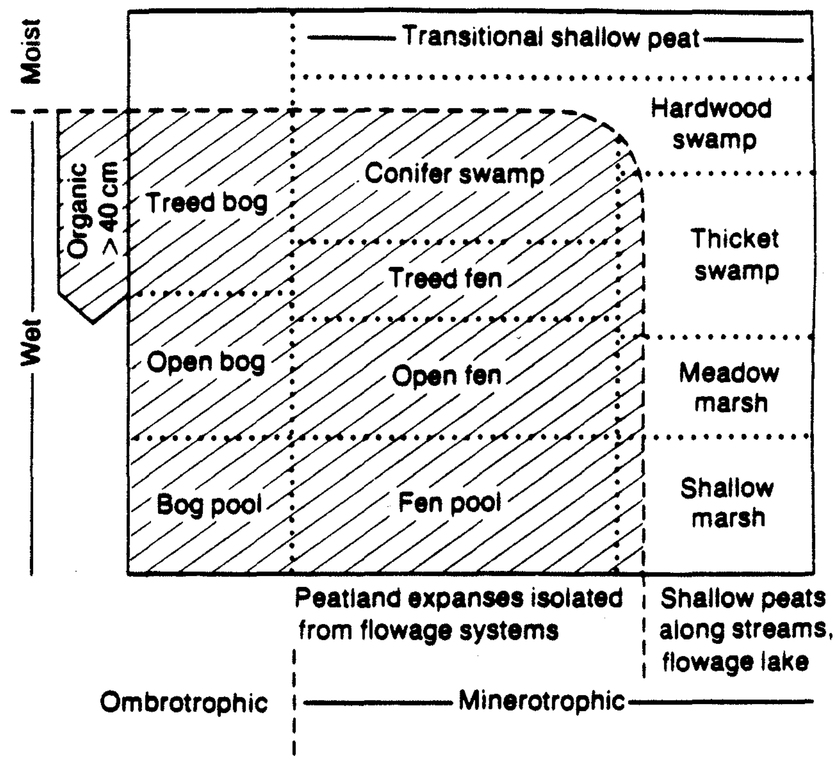

Figure 3.2. A conceptual edaphic model of the main units of wetlands in Ontario. (Source: Jeglum 1985).

3. Those with a closed tree cover: *Sphagnum teres* and *S. warnstorfii*.

In Alberta, however, relationships may not be the same as in Minnesota. *Sphagnum nemorum*, for instance, is restricted to shady micro habitats in Alberta whereas it is more common in the open in the eastern part of the continent. Factors which differ in Alberta are the continental climate and the higher latitudes which may lend relatively more importance to the shade factor.

3.3 PEAT DECOMPOSITION

In the 1920s peat and coal experts considered peatlands to be sterile and that chemical processes gave rise to the accumulation of organic matter. Waksman and Stevens (1928a, 1928b, 1929) published the first microbiological study of peatlands that established the existence of microbial populations in peatlands. Results from other studies (Christensen and Cook 1970; Latter et al. 1967; Sikora and Keeney 1983; Waksman and Stevens 1928a, 1928b, 1929; Waksman and Purvis 1932; Williams and Crawford 1983a) demonstrated that peatlands have large populations of microorganisms, equal to or exceeding those of mineral soils. Therefore, it is not the size of the microbial population that inhibits peat decomposition but the extent of microbial activity. Substrate quality and environmental conditions both interact to inhibit decomposition.

Decomposition in peatlands takes place through three different groups of microorganisms: bacteria, fungi, and actinomycetes. Bacteria are the largest group of microorganisms in peatlands. Latter et al. (1967) estimated bacterial biomass at a *Juncus squarrosus* study site in Great Britain to be 135 $g \cdot m^{-2}$. Bacterial populations are predominantly aerobic Gram-negative rods at the peat surface, changing into facultative anaerobes down the peat profile. Fungi are obligate aerobes and contribute significantly to the decomposition of the surface litter. Latter et al. (1967) found 62 $g \cdot m^{-2}$ of fungal biomass at the litter surface. Actinomycetes are described as being facultative anaerobes, and sensitive to pH of less than 5.0 (Waksman and Purvis 1932). Actinomycetes are responsible for the decomposition of the more resistant fractions of mineral soil and humus; i.e., the lignins, waxes, cutins, and certain hemicelluloses. Actinomycetes play a more important role in the decomposition of the surface peat of fens where the pH exceeds 5.0.

One of the strongest arguments for substrate quality controlling peat decomposition is preferential preservation. *Sphagnum* and *Eriophorum* remain as fossils in the peat profile when other species such as *Rubus* and *Smilicina* can be commonly found growing only on the peat surface. Considerable differences in decomposition rates occur between various mire species. On a blanket bog in the English Pennines, weight losses were recorded at 45% for *Narthecium* and *Rubus*, 10% for *Calluna* stems, and much smaller for *Sphagnum fuscum* and *Drepanocladus* spp. (Dickinson 1983). Substrate quality is markedly influenced by physical and chemical properties. Waksman and Stevens (1928a, 1928b, 1929) found that acid treatment of *Sphagnum* peats, which removed cellulose and hemicellulose, increased microbial activity, while Latter et al. (1967) found that low microbial populations on bare exposed peat were due to the resistant chemical nature of the peat, all of the soluble and readily decomposable materials having been previously degraded.

Lack of nutrients such as nitrate and phosphorus in the peat substrate has also been suggested as a controlling factor in decomposition (Marinucci et al. 1983). Nutrient studies involving nitrogen additions have had mixed results. Some researchers found that decomposition increased with nitrogen addition while others found that only a combined carbon and nitrogen supplement showed any substantial increase in microbial activity (Martin and Holding 1978). In Alberta, addition of phosphorus or lime to peat has been shown to increase decomposition to a greater extent than nitrogen addition (Kong et al. 1980). Nutrient availability under anaerobic conditions affects the decomposition rates of cellulose, oils, and certain fats. Amino acids, starches, sugars, and proteins decompose rapidly while lignins, waxes, cutins, and certain hemicelluloses do not decompose at all (Waksman and Stevens 1928a).

It is generally found that microbial activity decreases under conditions of high acidity. In cultures from British peatlands, bacteria were found to grow best at pH greater than 5.5, lower values restricting their ability to metabolize nutrients. Only 1% of the aerobic isolates grew at pH 5.0 as compared to over 80% at pH greater than 5.5 (Collins et al. 1978). However, Williams and Crawford (1983b) found that a pH as low as 3.0 had no significant effect on either microbial communities or pure cultures. In fact mineralization processes were found to proceed more rapidly at a pH of 4.0 than 7.0.

The rate of mineralization of peat by microbial organisms is governed by the rate of biochemical reactions. Most biochemical reactions are considered to occur at an average respiratory quotient of $Q_{10} = 2$. Results presented in a study by Williams and Crawford (1983b), showed that 37% of ^{14}C-labelled *Sphagnum* was converted to CO_2 at 30°C while only 4% was converted at 4°C. While this study does show that low temperatures can contribute to peat formation, hence the prevalence of peatlands in cool climates, it cannot account for peat accumulation that occurs in the tropics.

Fewer than 10% of the microbes found in peatlands are strict aerobes (Collins et al. 1978). Low oxygen conditions have been found to reduce microbial populations, but significant reductions occur under low oxygen conditions combined with low nutrient levels. However, the largest percentage of the bacterial population at the surface of mires is aerobic (Collins et al. 1978). Anaerobic bacteria are found in smaller population sizes and tend to decrease with depth (Stout 1971). Fungi are aerobic decomposers and contribute significantly to cellulose decomposition in the surface layers. Fungal colonization has been found by Latter and Graag (1967) and Dickinson and Maggs (1974) to be significantly reduced 10 cm below the peat surface. Actinomycetes are considered to be facultative anaerobes but few studies of their activity in peats have been undertaken. They are an important group of decomposers as they attack cellulose and are capable of attacking organic complexes of peat (Collins et al. 1978).

In summary, decomposition of peat is not limited by the size of the microbial population, but rather by the extent of microbial activity. Substrate quality and environmental conditions both interact to inhibit decomposition. Substrate quality can account for preferential species preservation within the peat profile and account to some extent for differences in decomposition rates among different types of peatlands. Environmental factors control changes in decomposition within the peat profile as fresh litter placed at some depth will decompose at a fraction of the surface rate. Factors that influence substrate quality include limited levels of nutrients such as nitrogen, phosphorus, and calcium, and the physical structure of the plant material. Environmental factors involve lack of oxygen, low pH, and low temperature.

3.4 MODES OF PEAT ACCUMULATION

3.4.1 Terrestrialization

Terrestrialization is the formation of a peatland by filling of a water body with organic remains, usually by gradual extension of peat-forming communities from the shoreline toward the middle of a lake. This is the most widely known and documented method of peatland development. Terrestrialization begins in shallow lakes and ponds, and probably occurs immediately following deglaciation. Runoff waters rich in silts and clays fill the depressional basins. These nutrient-rich waters provide ideal conditions for the growth of algae, diatoms, bacteria, and zooplankton. Dead microorganisms settle to the basin bottom beginning the sedimentation process (Everett 1983). Shallow pond weeds grow wherever light penetrates to the bottom, and they contribute to the deposition of organic debris. Microbial respiration depletes oxygen levels in the water creating anoxic conditions, further hindering microbial decomposition. *Carex*, *Typha*, *Scirpus*, and ericaceous shrubs become established around the periphery of the lakes, forming a support structure for the colonization of aquatic mosses. A floating mat develops. With the development of a floating mat, peat accumulation begins to occur in three distinct strata (Kratz and DeWitt 1986). Figure 3.3 depicts a generalized concept of terrestrialization while Figure 3.4 outlines peat formation in more detail as described by Kratz and DeWitt (1986) in the Wisconsin peatlands. Figure 3.4 shows that the first stratum develops as an upper layer of peat formed in the floating or grounded mat. It consists of fibrous, poorly decomposed peat, with an organic matter content of 92 to 98%. The peat is highly structured with interconnecting plant remains. The second stratum forms as a middle layer of peat dropped from the side or bottom of the floating mat. It is a layer of debris peat which is fibrous, moderately decomposed, and has an organic matter content of 75 to 85%. The peat consists of unstructured *Sphagnum* microphylls and other small unidentifiable pieces of organic material. The bottom stratum is formed from lake sediments which are well decomposed and have 40 to 70% organic matter. It consists of a dark brown, amorphous substance in which diatom frustules are common. Upon complete enclosure of the lake basin with bryophytes and vascular plants, a fen is formed. The fen vegetation will persist as long as waterlogged conditions exist at the peat surface (Everett 1983). However, bryophyte species, particularly *Sphagnum* species, have the capacity to modify their own microclimates thus changing the environmental conditions (Glime et al. 1982). Under the proper climatic and hydrologic conditions *Sphagnum* mounds will form coalescing and covering the fen in a broad domed raised bog (Everett 1983).

A classic example of terrestrialization in North America can be found in the Myrtle Lake peatland of northern Minnesota. Myrtle Lake is located within the former lake basin of glacial Lake Agassiz (Heinselman 1970), which receded at 11 740 B.P. Organic sedimentation began in Myrtle Lake with aquatic peat deposits. At

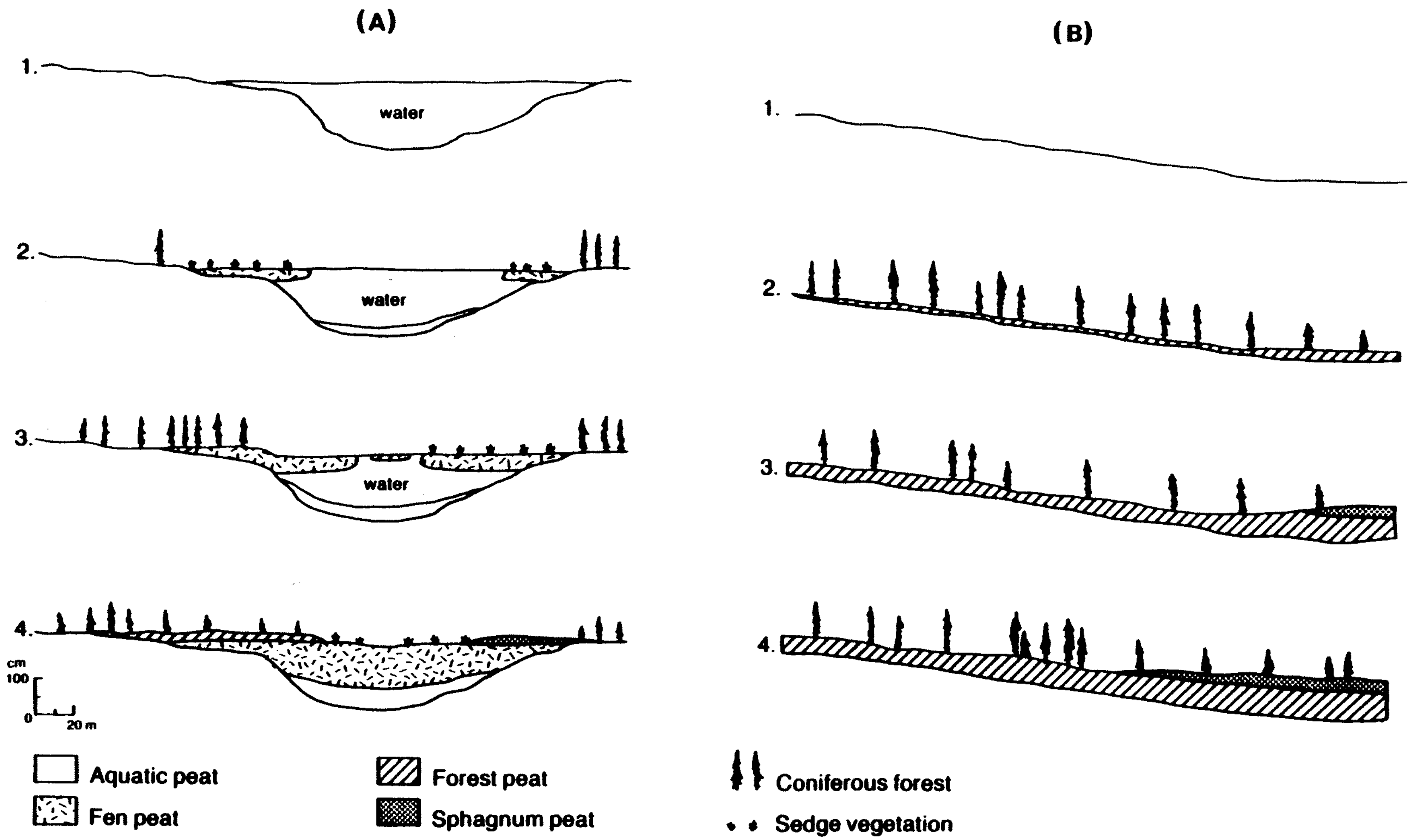

Figure 3.3. Schematic diagrams showing the probable stages of the (A) terrestrialization and (B) paludification processes. (Adapted from Tarnocai 1978).

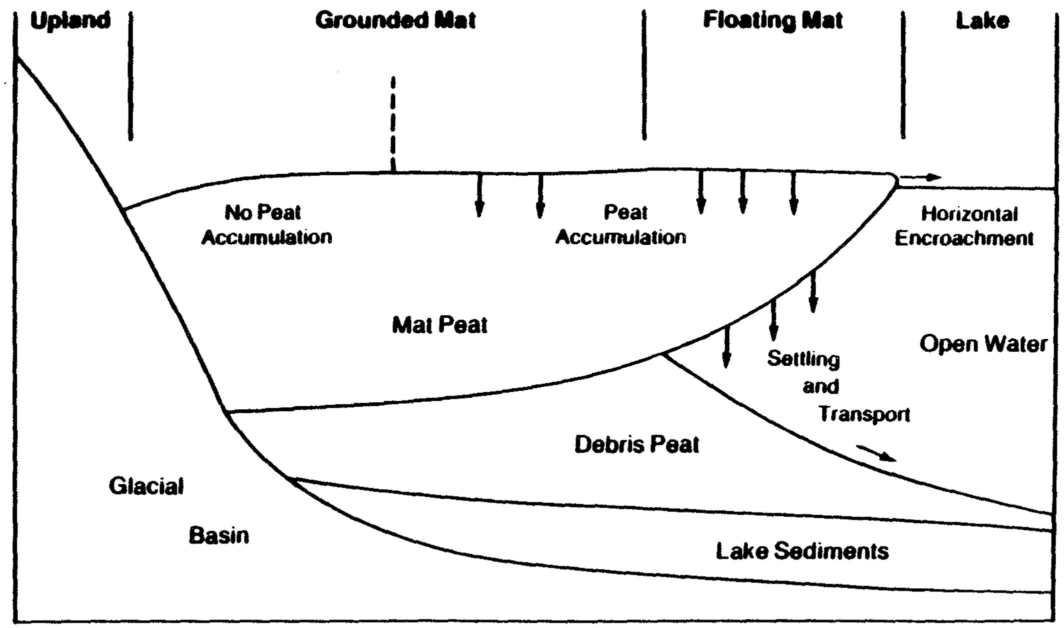

Figure 3.4. Conceptual model of the way small peatlands form around lakes in north temperate latitudes. (Source: Kratz and DeWitt 1986).

8000 B.P. the postglacial, warm dry interval began causing fens, marshes, and shrub carrs to invade the lake basin. This event is marked by a basal layer of sedge peats that covered nearly half of the area. As precipitation increased towards the close of the warm dry period, conifers invaded the fens and carrs forming swamp forest peats (minerotrophic). About 3100 B.P. a water table divide occurred north of Myrtle Lake. With the occurrence of a water table divide, the area became ombrotrophic and mineral depletion developed. This resulted in the formation of an ombrotrophic raised *Sphagnum* bog.

Examples of terrestrialization exist in Alberta; however, they do not tend to cover the complete successional cycle from open lake to raised bog. Macrofossil analysis of peat cores from Mariana Lakes, 100 km south of Fort McMurray, has been used to document peatland development from the lake sediment stage to a poor fen stage (Nicholson 1987). The macrofossil history of Mariana Lakes is depicted in Figure 3.5. Peat formation here began 8180 B.P. with finely stratified limnic sediments. Following the limnic stage a floating mat of *Drepanocladus* spp. formed around the lake. The floating mat allowed the establishment of monocots, ericaceous shrubs, and *Sphagnum* spp., and the site developed into a poor fen. Poor fen (monocot, ericaceous shrub, and *Sphagnum*) peat has been accumulating at Mariana for 3400 years.

Peatlands that formed by terrestrialization in the Edmonton area have been described by Osvald (1970). Sites near Lakes Boag and Fedorah had bottom sediments consisting of neckron muds overlying clays. At the Lake Boag site, these were overlain by *Carex* peats, and the topmost strata consisted of *Sphagnum* peats. Two stratigraphic profiles from two sites at Lake Boag are depicted in Figure 3.6. Osvald (1970) interpreted these peat profiles as having begun development in a shallow lake which was overgrown by a sedge fen, and then succeeded into a bog. At Fedorah, the shallow lake was quickly inundated by a sedge fen, rich in leaf mosses. The peat-forming community here has since remained as a stable edge-brown moss association.

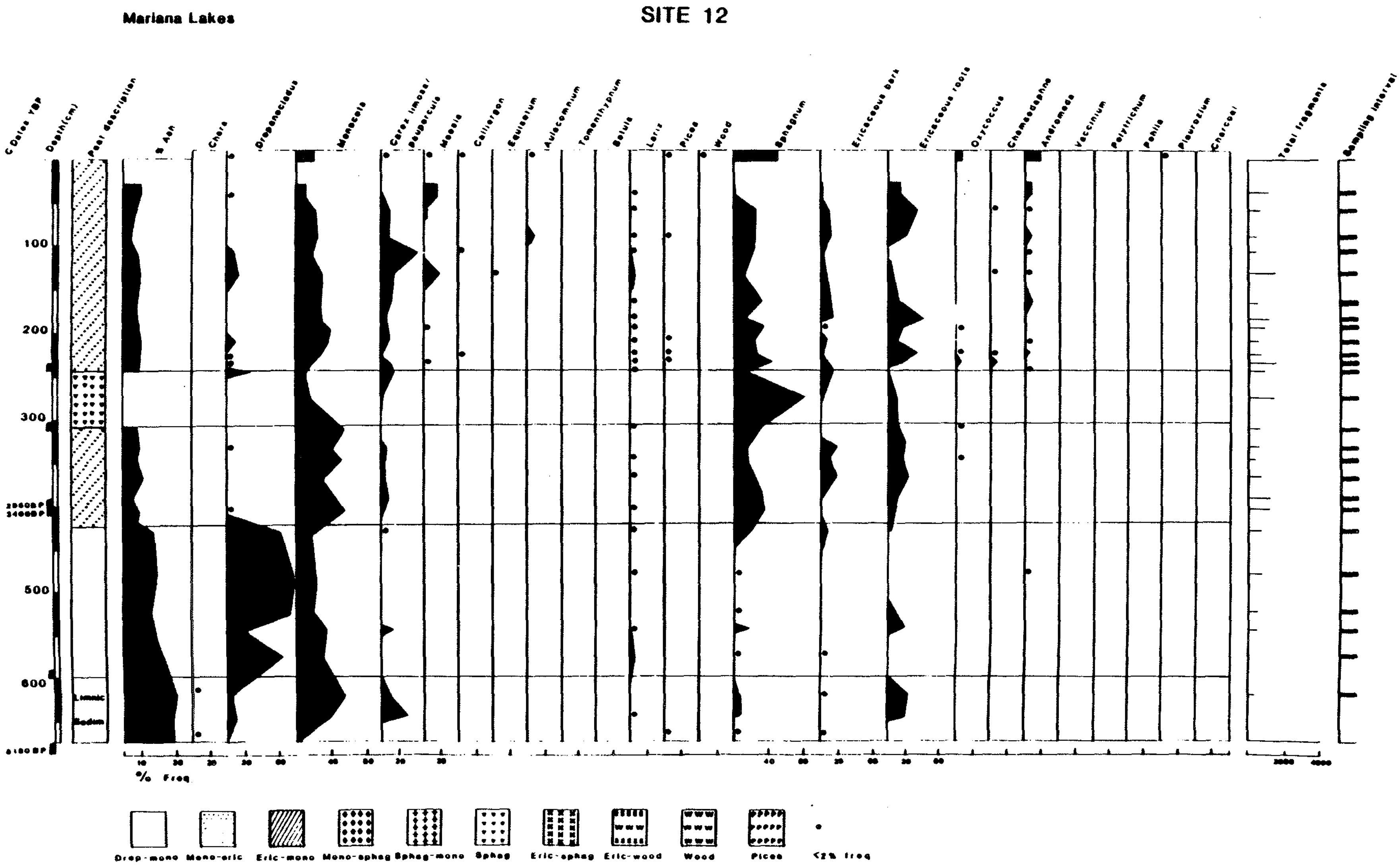

Figure 3.5 Macrofossil analysis of peat cores, Mariana Lakes, Alberta. (Source: Nicholson 1987).

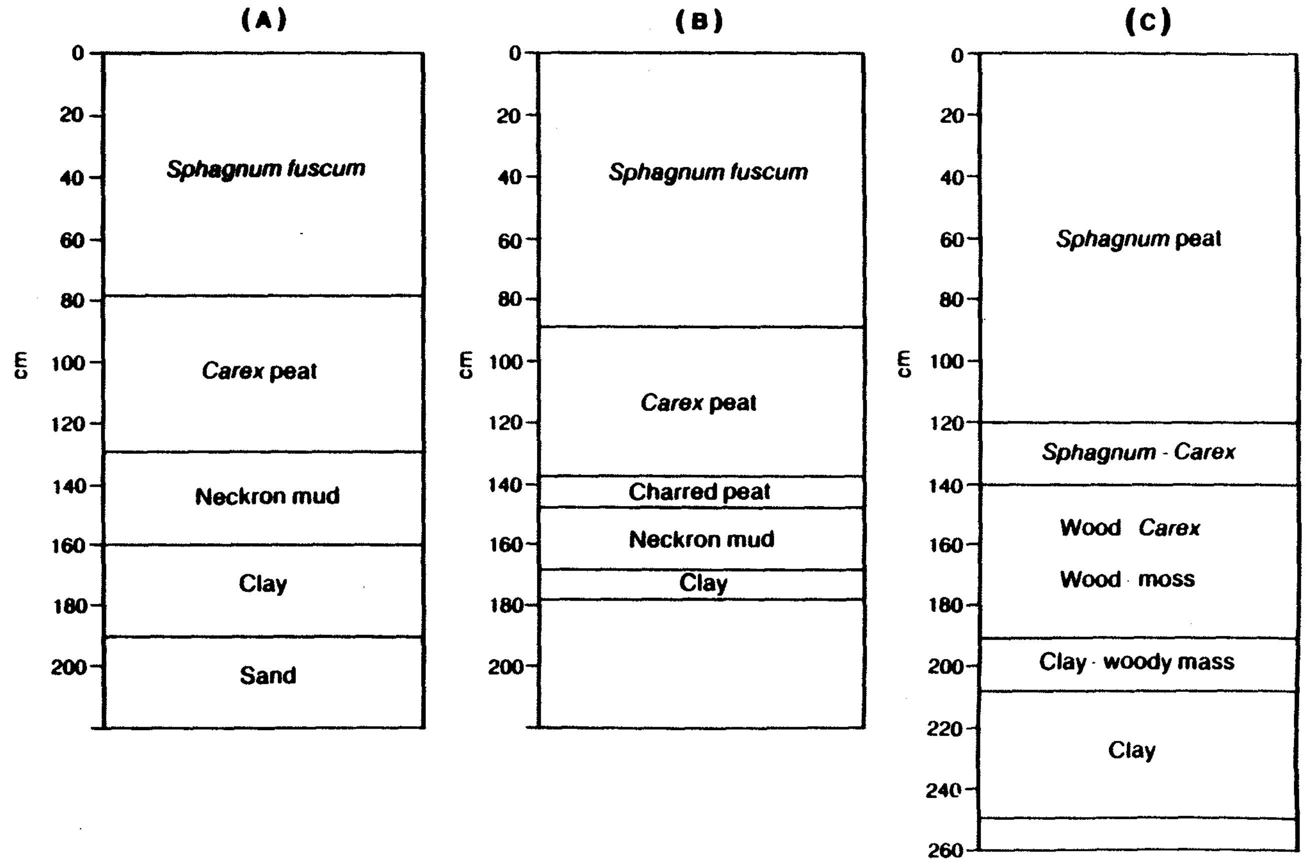

Figure 3.6. Stratigraphic profiles of bogs developed by terrestrialization at two sites near Lake Boag (A and B) and by paludification (C) in the Edmonton area. (Adapted form Osvald 1970).

3.4.2 Paludification

Paludification refers to swamping, or the formation of peatlands over previously forested land, grassland, or even bare rock due to climatic or allogenic processes. It is responsible for most of the peat formation in the world (Sjors 1983), particularly in cool oceanic regions and in the boreal and subarctic zones (Sjors 1963). Paludified sites are affected by increasingly impaired drainage. The general direction of development is toward greater wetness (Sjors 1963). Climatic change is normally thought to promote peat growth on mineral soil. According to Finnish sources extensive paludification occurred at about 6500 B.P. and 4800 B.P. In Canada, Nichols (1969) undertook a survey of basal peats in shallow ombrogenous bogs. He found that the basal dates fell into three groups: 3480 B.P., 2360 B.P., and 700 B.P. Previously documented radiocarbon dates in palynological and stratigraphic studies indicate that periods of climate deterioration to colder conditions occurred in Canada at 3500 B.P., 2400 B.P., and 700 B.P. (Nichols 1969). A conceptual model of paludification is depicted in Figure 3.3. The peat stratigraphy of a *Sphagnum* bog located 60 km west of Edmonton provides a local example of the paludification process (Figure 3.6). A moist depression became covered with a woody moss peat. Early peat development was interrupted by floods, which brought erosional clay onto the peat surface. This peatland development then progressed through a wood-*Carex* and *Sphagnum-Carex* stage, indicating increased moisture and oligotrophy, to a *Sphagnum*-dominated stage (Osvald 1970).

Zoltai and Johnson (1985) documented the paludification of a mineral rise in a graminoid dominated fen near Rocky Mountain House, Alberta. They found that peat accumulation in the fen overwhelmed a nearby thicket swamp, allowing the development of a densely forested bog.

Peat profiles from Mariana Lakes suggest that peat formation here occurred in two stages (Figure 3.7). Initial peat formation began at about 8180 B.P. in the lake basins and culminated

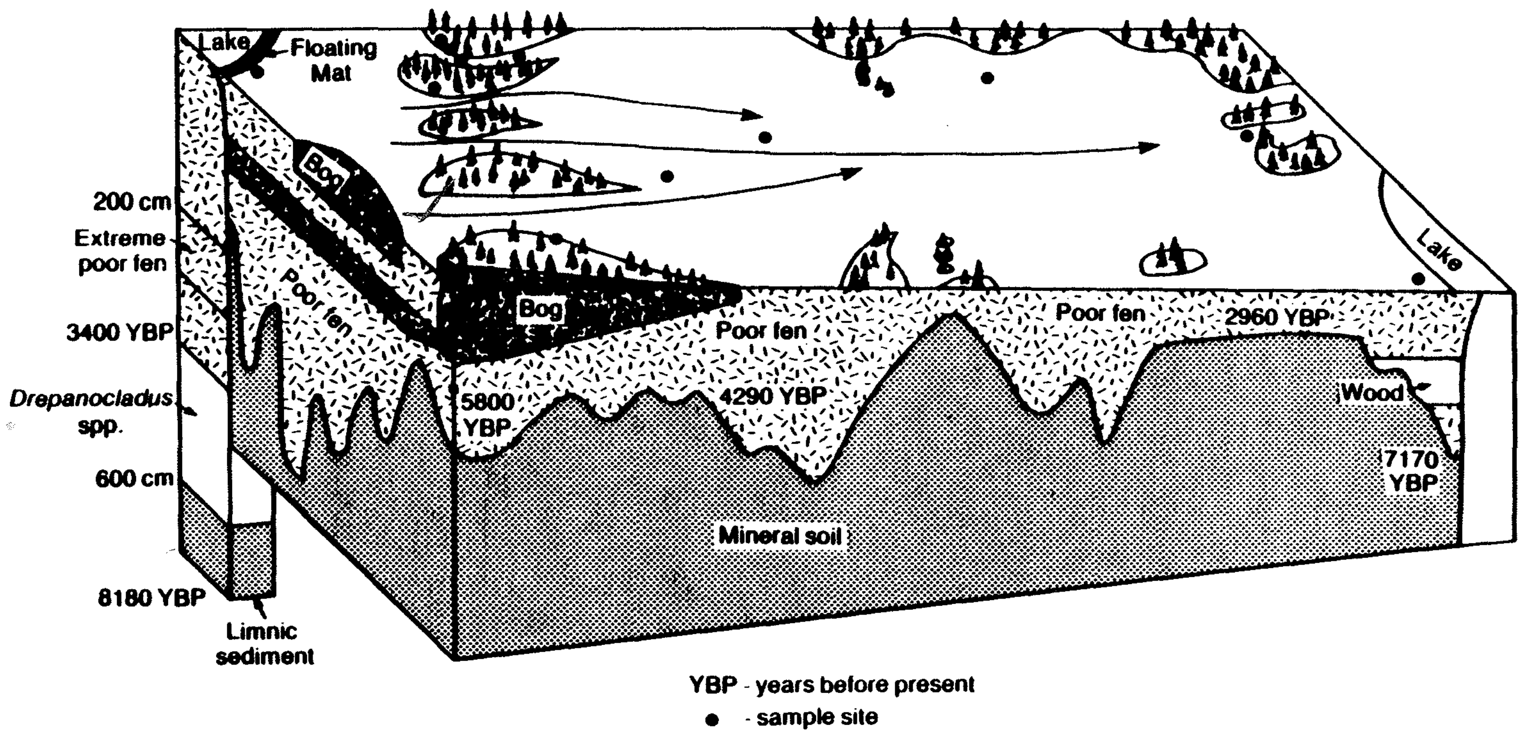

Figure 3.7. Composite of peat macrophyte stratigraphy at Mariana Lakes, Alberta. (Source: Nicholson 1987).

with a *Sphagnum*-dominated poor fen. Paludification of the low-lying areas occurred much later at 4290 and 2960 B.P.

In the Attawapiskat River area of the Hudson Bay Lowlands, the rising low shores of the Tyrell Sea were first occupied by intertidal salt marshes, and quickly invaded by a swampy forest. The well-decomposed forest humus made the soil less permeable and paludification began. A wet rich fen (*Drepanocladus*, *Meesia*) developed over the swampy forest and was followed by a more oligotrophic *Carex* spp. fen. This succeeded into a shrub fen community, and culminated with a *Sphagnum*- ericad bog (Sjors 1963).

Research at the Red Lake peatlands in north-central Minnesota has revealed the complete post glacial history of the area. Glacial Lake Agassiz covered the entire Red Lake area 12 000 years ago (Griffin 1977) and was drained at approximately 11 500 B.P. Pollen records from the vicinity of Red Lake indicate that the depressions were originally occupied by lakes from 11 500 to 10 500 B.P. From 10 500 to 9000 B.P. the lakes drained and a boreal forest grew upon the former lake beds. During the postglacial warming trend, a mixed mesophytic forest covered the area, evolving into prairie vegetation from 8000 to 4000 B.P. In the Lake Agassiz plain, a well developed A_1 soil horizon formed under the prairie vegetation. Paludification began after 4000 B.P. with the development of aquatic and marsh environments. Macrofossils found in the basal peat of Red Lake include: *Typha*, *Lemna*, *Eleocharis*, *Scirpus*, *Sagittaria*, and Gramineae (Griffin 1975,1977). In the peat cores an abrupt change occurred when the pond-swamp environments gave way to sedge meadows with *Salix* shrubs. Under increasing paludification, the Red Lake peatlands succeeded into a minerotrophic rich fen with *Calliergon*, *Scorpidium*, and *Campylium* (Griffin 1977; Gorham and Janssens 1985; Janssens 1987). A more oligotrophic S*phagnum*-dominated, poor fen stage ensued followed by an ombrotrophic *Sphagnum* bog.

3.5 PEAT-FORMING VEGETATION GROUPS

Through peat stratigraphy and macrofossil analysis it is possible to study successional changes occurring over long periods of time in peatlands. The peat classification systems used today are based on grouping peat types according to vegetation gradients (Svensson 1986). Identified plant remains are assigned to species groups, which are used to distinguish vegetation units in the present mire vegetation. From these groups it is possible to reconstruct the vegetation history of the peatland.

Reconstructions are difficult because of preferential decay, over-representation, and differential decay rates. Certain mire species such as *Rubus chamaemorus*, *Smilacina trifolia*, and lichens are not preserved in peat. Higher nutrient concentrations have been reported in *Rubus* leaves (Coulson and Butterfield 1978) and may account for their rapid decomposition. Environmental factors (O_2, moisture, temperature) control the rate at which mire plants enter into the anaerobic acrotelm. Therefore the vascular plants and mosses growing highest on the hummocks are often the absent species (Svensson 1986). *Sphagnum* species are often over-represented in peat samples due to the fact that *Sphagnum* branch and stem leaves do not remain attached to the stem and easily disperse. Highly humified basal peats often contain very decomposed *Sphagnum* leaves with large amounts of ericaceous shrub roots and bark. It appears that the relationship between the amount of decomposition and the percentage of ericaceous material is due to the loss of decomposed *Sphagnum*. The penetration of peat horizons by vascular plant roots from above further complicates reconstructions. Vascular plants can become incorporated into a peat type to which they have no direct affinity (Svensson 1986).

3.5.1 Peat Groups

In North America, Tallis (1983) has identified 7 peat groups from 36 peat profiles distributed equally among the New England states, the lake states, and the north Pacific coast.

The same has been done by Walker (1970, cited by Tallis 1983) for 40 sites in the British Isles. Table 3.2 is a summary of their findings.

Canadian stratigraphic studies are rare. Two studies which have presented peat classification are those of Zoltai and Johnson (1985) and Ovenden (1985). Table 3.3 is a summary of their peat classification studies.

In comparing these four examples of peat classification a few observations can be made. The first is that the classification schemes involve a wide variety of peat communities, not all of which are represented at any given site. The second is that each classification scheme differs because of the level of definition used to determine each peat type. With the increasing number of detailed

Table 3.2. Summary of North American and British peat groups.[a]

North America	Britain
1. Lake mud	1. Lake mud
2. Sedimentary peat - fine organic accumulation below floating mat	2. Swamp mud - from Reed Swamp community
3. *Phragmites* peat	3. Fen peat - accumulated below sedge-tussock and herbaceous fen community
4. Moss peat - remains of hypnoid mosses	4. Wood peat - below swamp and fen carr
5. Sedge peat - floating raft of vegetation	5. Bog peat - composed of *Sphagnum* peat
6. Forest peat - *Sphagnum* with abundant woody remains	
7. *Sphagnum* peat - *Sphagnum* remains without wood	

[a] Adapted from Tallis 1983.

Table 3.3. Summary of Canadian stratigraphic peat groups.

Zoltai and Johnson (1985)			Ovenden (1985)	
	Peat Type	Inferred Fossil Plant Community		
1.	Pool (with marl)	Pool with sedge, moss	1.	Lake sediments
2.	Pool (without marl)	Pool with sedge	2.	Marsh-grass, *Salix*, *Equisetum*
3.	Woody Fen	Tamarack, shrub, sedge	3.	Wet thicket or wet sedge-meadow willow, sedge, grass
4.	Shrubby Fen	Shrub, sedge, *Scorpidium*	4.	Wet thicket-*Andromeda*, *Sphagnum*, *Chamaedaphne*, sedge, grass
5.	Shrubby Fen	Shrub, sedge, *Drepanocladus*	5.	*Carex aquatilis* marsh
6.	Graminoid Fen	Sedge, *Scorpidium*	6.	Amblystegiaceae carpet
7.	Graminoid Fen	Sedge, *Drepanocladus*	7.	*Sphagnum* carpet
8.	Thicket swamp	Willow, sedge, moss	8.	Tussock tundra and shallow peatland
9.	Forested Bog	Black spruce, feathermoss	9.	*Sphagnum* - Cuspidata
10.	Treed Bog	Black spruce, *Sphagnum*	10.	*Picea*, charcoal, *Rubus*, *Sphagnum* - Acutifolia, *Sphagnum* - Cuspidata
11.	Shrub Rich Wooded Bog	Black spruce, heath, *Sphagnum*		

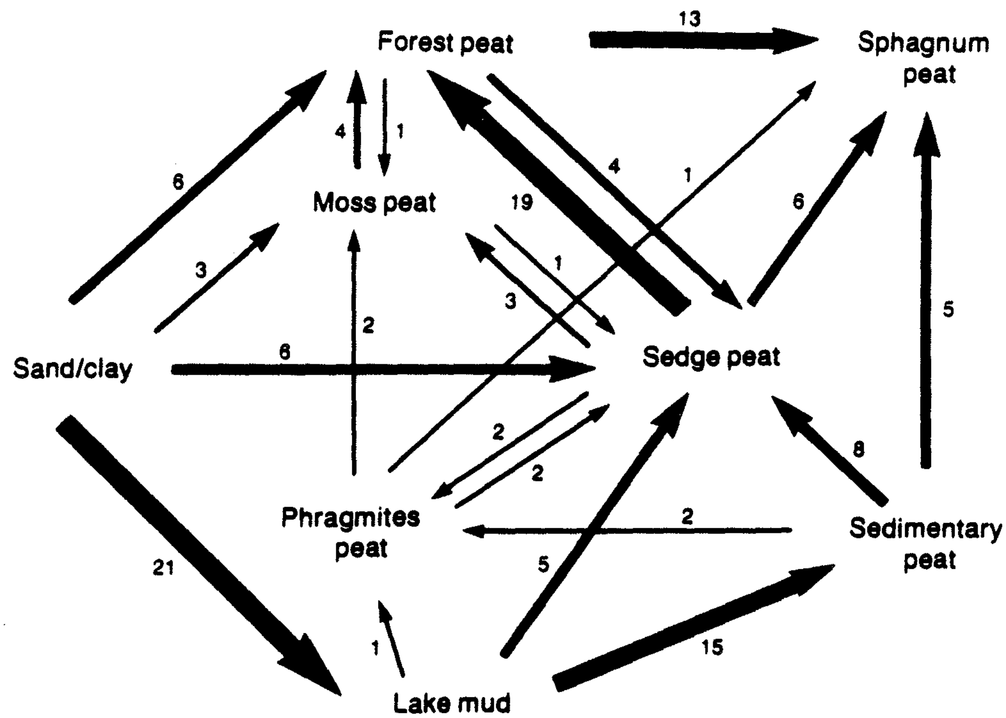

Figure 3.8. Stratigraphic sequences in peat profiles from North America. The arrows connect pairs of superimposed strata in published profile discriptions; the number against each arrow gives the number of recorded instances of that particular transition. (Source: Tallis 1983).

stratigraphic studies that are being undertaken in both North America and Europe the full range of peat types corresponding to common wetland communities should be elucidated.

3.6 PATTERNS OF VEGETATIONAL CHANGE IN PEATLANDS

Figure 3.8 summarizes the stratigraphical sequences recorded by Tallis (1983) from the 36 peat profiles across North America. This diagram reinforces the belief that there are many directions of succession or development in wetlands. Vegetational change in peatlands occurs through two processes. Autogenic changes are initiated by wetlands themselves. Such processes include the shallowing of the water basin with the accumulation of organic debris, the ability of Amblystegiaceae and *Sphagnum* mosses to acidify their surroundings and create higher patches of terrain which are removed from the water table and are open to colonization, and water channelling (Tallis 1983). Water channelling is a process where water flowing through a mire system becomes channelled into definite water courses. Between the flow paths are areas of peat which are affected by mobile waters only during periods of excessive inflow. Accordingly, there is a slow progression towards ombrotrophic conditions in these isolated areas of peat (Tallis 1983).

Allogenic changes are brought about by external factors such as climate and hydrology. The appearance of recurrent surfaces in Europe, in which weakly humified ombrotrophic peats are underlain by highly humified peats containing tree stumps, has been thought by researchers to be due to climatic changes (Tallis 1983). The boundary between the two peats is marked by a thin layer of *Scheuchzeria-Sphagnum cuspidatum* peat. The stratigraphic sequence has been interpreted to be the result of a climatic change from a prolonged period of dryness to an increase in wetness (Tallis 1983). The most notable example of allogenic hydrological changes occurs with stream capture. In North America, stream capture was responsible for the development of ombrotrophic peat conditions in the Lake Agassiz peatlands (Heinselman 1970). A flat valley fen developed in a broad plateau that had a slight northwest drainage. Headwater erosion of the Little Fork River into the northeast section of the peatland created a two-way drainage divide in the centre of the fen. Groundwaters no longer passed through the peatland and ombrotrophic conditions ensued.

3.7 EXAMPLES OF PEAT STRATIGRAPHY IN ALBERTA

Macrofossil analyses of three peat cores taken by Zoltai and Johnson (1985) near Rocky Mountain House are depicted in Figure 3.9. The peat cores form a transect which runs from a mineral-enriched water track into an adjacent bog island. The peat stratigraphy establishes that paludification occurred in the water track and up the adjacent rise.

Prior to 6600 B.P. a graminoid-dominated fen was established in a depression which was part of a flowing groundwater system. On the edges of the fen and on the slight rise (island) thicket swamps were growing (willow, grasses, mosses). As peat accumulated in the fen it overwhelmed the nearby thicket swamp, creating wetter conditions. A densely forested bog developed on the rise, and kept pace with peat accumulation in the fen, laterally expanding into the fen.

At Mariana Lakes, (Figure 3.7) peatland development began with terrestrialization of the lake basins. A floating mat of *Drepanocladus* formed followed by a poor fen. At the end of the climatic optimum, paludification of the low-lying areas occurred (5800, 4290, 2960 B.P.). In the upper (older) reaches of the site, increasing peat depths created more oligotrophic conditions and the area became *Sphagnum*-dominated. Climatic events increased hydrological flows onto the site, channelling more minerotrophic water in the water tracks. In the hydrologically sheltered areas such as the teardrop-shaped islands, the peats did not become inundated with mineral-enriched waters and remained as topographically high ombrotrophic islands. Areas not hydrologically

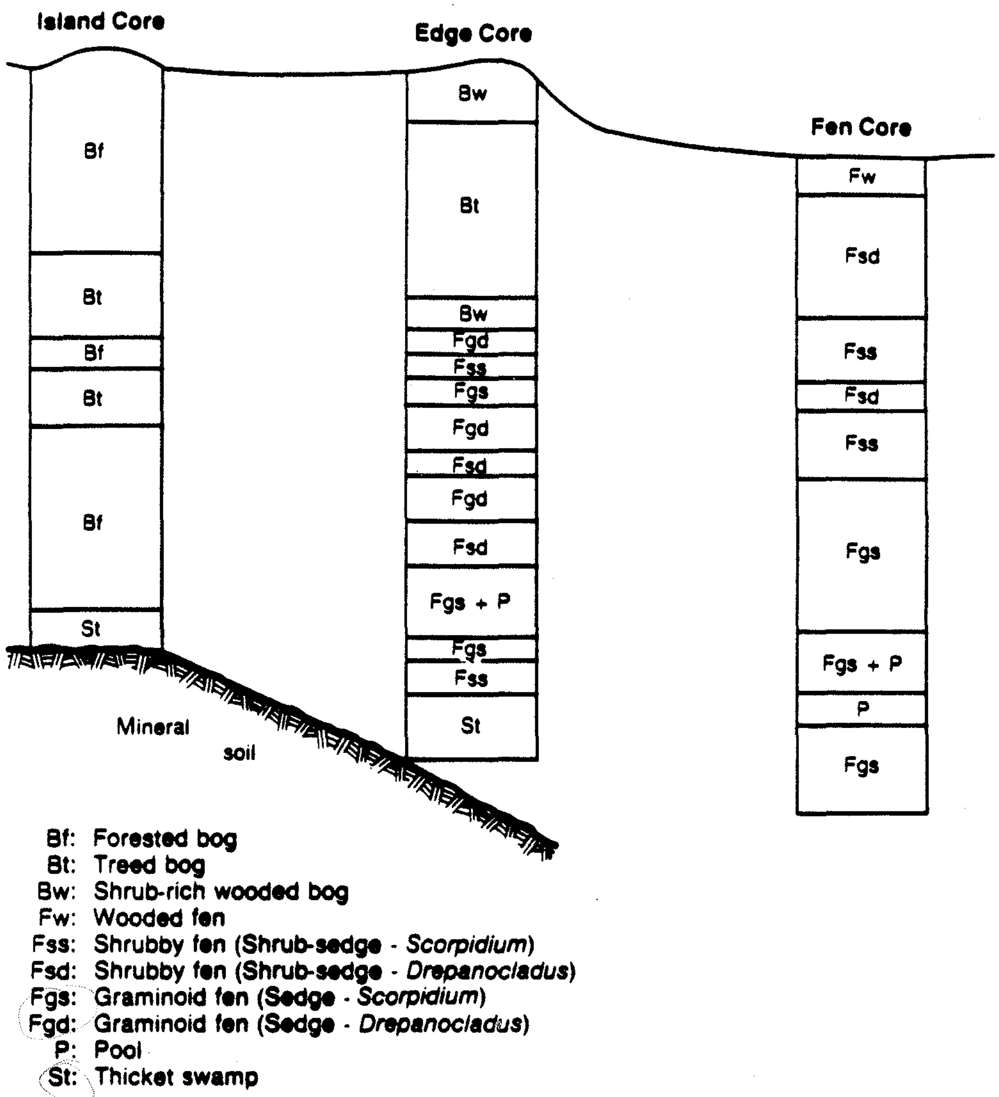

Figure 3.9. Macrofossil analysis of three peat cores near Rocky Mountain House, Alberta. (Adapted from Zoltai and Johnson 1985).

sheltered, such as the water tracks and lower reaches of the site, received mineral-enriched waters and reverted back or remained as poor fen.

3.8 SUMMARY

Peatland formation is the result of many interactive factors, both biotic and abiotic. Factors that control the growth and composition of peat-forming vegetation include climate, nutrient availability, moisture conditions, and shade. Under cool, wet anaerobic conditions, vegetation remains are not easily decomposed by microorganisms, and peat accumulates as a result. Terrestrialization, or infilling of water bodies, and paludification, or swamping of previously 'dry' land, are two main modes of peat accumulation. In North America, the first peatlands began to develop in depressions created by the retreating ice at approximately 12 000 B.P. Terrestrialization processes gradually reduced nutrient enrichment, and succession progressed towards more oligotrophic conditions. Peatland development by paludification occurred extensively. This appears to be related to climatic conditions, as were recurrent surfaces in the peat profiles in Europe. Thus, peatland development is controlled by allogenic processes, or processes brought about by external factors such as climate and hydrology, or autogenic changes, or processes initiated by the wetlands themselves. Autogenic processes move a peatland ecosystem towards shallowing of water in basins by accumulation of organic debris, towards acidification, and towards channelling of water. The specific type of peatland which develops, be it bog, fen, swamp, or marsh, reflects the regional and local conditions of climate, hydrology, water chemistry, and basin topography.

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4. CHEMICAL AND PHYSICAL PROPERTIES OF PEAT

L.W. Turchenek

4.1 INTRODUCTION

The quality of peatland drainage waters is influenced by the nature of the input waters as well as by the chemistry of the peat itself. Likewise, hydrologic functions of peatlands are influenced by the physical properties of the peat. Human activities can alter the chemical and physical attributes of peatlands and thereby change the quantity and quality of drainage waters. Some properties which are important for an understanding of the changes which can occur are described in this chapter.

4.2 ORGANIC CHEMISTRY OF PEAT

The organic chemistry of peat has been thoroughly reviewed by Fuchsman (1980), Walmsley (1977), Clymo (1983), and others. The major peat components are cellulose, hemicellulose, lignin, bitumen, and humic materials. The relative concentrations of these are dependent on factors such as botanical origin, degree of humification, depth, and age (Clymo 1983). As decomposition progresses, cellulose, hemicellulose, and lignins break down and proportions of bitumens and humates increase (Smith et al. 1958). Such differences with degree of decomposition have been demonstrated for Alberta peats using solid-state ^{13}C nuclear magnetic resonance spectroscopy by Abboud and Turchenek (1985).

Humic materials in peats consist of humic acids, fulvic acids, and humins. These materials are important in terms of cation exchange capacities, their acidic nature, and their exchange and chelating properties. In general, the more decomposed the peat, the more the exchange capacity is derived from humic materials (Puustjarvi 1977). Because of these properties, humic substances are particularly important in influencing the fate of external acid inputs and the nature and dynamics of trace elements in peats and peat waters.

4.3 CARBON

The organic carbon content is generally similar for different botanical types of peat which are weakly or moderately decomposed. The carbon contents diminish in more humified peats. Data for some peat profiles from Alberta are presented in Table 4.1. The values range from 45 to 57% in fibric and mesic peats. Humic and some mesic peats from basal layers in these peat deposits have high ash contents resulting from concentration due to humification of the organic matter, addition of mineral matter from external sources, or a combination of both. Peats are arbitrarily defined as materials containing more than 17% organic carbon by weight. Materials with less carbon than this, such as the basal layer labelled as a mineral A horizon in Table 4.1 (II Ahg in the Poor Fen), are called mucks in other classification systems (Landva et al. 1983).

4.4 ASH

The inorganic components of peat are commonly estimated collectively through determination of the ash content. The principal constituent of the ash is usually either calcium or silicon. The silicon is derived from extraneous sources such as sediments being washed in from upland areas, dust blown in by wind, or deposition as entrained particles in precipitation. The ash content can vary from about 1% to the limit, by definition, of 70% (Table 4.1). The principal reason for variation in ash contents is decomposition of peat since mineral elements accumulate as the organic components of peat are mineralized. Along with degree of decomposition, the ash content in a peat profile generally increases with depth. In the absence of elemental analyses, the ash content can be a useful indicator of the amount of mineral elements which can possibly be

Table 4.1. Characteristics of some profiles representative of peat deposits in the Athabasca area, Alberta.[a]

Soil Horizon	Depth cm	Type of Peat	Ash %	C %	N %	C:N	pH Field	pH Lab[c]	Exchange Cations[b] ($cmol_c \cdot kg^{-1}$) Ca^{2+}	Mg^{2+}	K^+	Na^+	Al^{3+}	H^+	Total	Base Sat %
		Very Oligotrophic Peatland - Bog														
Of1	0-15	fibric, sphagnum	1.1	54.5	0.56	97	4.1	3.1	8.4	5.2	1.8	0.1	0.9	146.9	163.3	9
Of2	15-40	fibric, sphagnum	6.8	52.2	0.99	53	4.1	3.4	22.2	6.7	1.0	0.1	0.4	114.8	145.2	21
Of3	40-110	fibric, wood-sphagnum	5.5	56.4	1.49	38	4.7	4.2	54.2	12.2	0.1	0.1	0.4	105.2	172.2	39
Of4	110-155	fibric/mesic, sphagnum	4.2	56.6	2.38	24	4.7	4.8	59.4	10.2	0.1	0.1	0.4	92.6	162.8	43
Om	155-170	mesic, moss-sedge	11.6	52.3	-	-	5.3	5.2	64.2	8.0	0.1	0.1	4.4	96.2	173.0	42
IICg	170-180	sandy clay loam	-	0.8	-	-	-	6.4	-	-	-	-	-	-	-	-
		Oligotrophic Peatland - Poor Fen														
Of1	0-28	fibric, wood-sphagnum	2.6	53.0	0.99	54	-	4.8	53.6	17.9	2.7	0.1	0.8	70.9	140.6	51
Om1	28-119	mesic, wood-moss-sedge	7.1	54.3	2.07	26	5.3	5.0	80.0	15.4	0.1	0.1	0.7	91.3	187.6	51
Om2	119-145	mesic, wood-moss-sedge	8.0	53.8	2.26	24	5.5	5.1	96.4	16.3	0.1	0.1	2.1	112.1	227.1	50
Om3	145-190	mesic, wood-moss-sedge	8.4	53.4	2.46	22	5.6	5.0	88.4	14.0	0.1	0.2	1.2	115.6	219.5	47
Om4	190-217	mesic, wood-moss-sedge	18.4	48.3	-	-	5.7	5.2	90.2	13.0	0.1	0.1	1.9	106.6	211.9	49
Oh1	217-234	humic, wood-moss-sedge	62.4	22.2	-	-	-	5.3	-	-	-	-	-	-	-	-
Oh2	234-244	humic, wood-moss-sedge	29.4	32.9	1.80	18	-	5.3	-	-	-	-	-	-	-	-
IIAhg	244-250	silt loam	-	7.2	-	-	-	5.3	-	-	-	-	-	-	-	-
		Mesotrophic Peatland - Intermediate Fen														
Om1	0-25	mesic, moss-sedge	8.0	53.9	2.98	18	5.9	5.8	114.3	22.2	0.4	0.4	0.8	50.7	188.8	73
Om2	25-50	mesic, moss-sedge	6.4	54.1	2.99	18	6.2	5.8	102.2	20.3	0.1	0.4	0.9	47.7	171.6	72
Om3	50-75	mesic, moss-sedge	7.4	54.7	2.98	18	6.2	5.7	113.4	19.1	0.1	0.1	1.2	66.5	200.4	66
Om4	75-130	mesic, moss-sedge	7.3	54.2	3.13	17	5.9	5.7	101.4	17.5	0.1	0.1	1.5	71.1	191.7	62
Om5	130-140	mesic, wood-moss-sedge	16.0	50.5	1.25	40	6.2	5.6	146.0	22.5	0.1	0.2	3.7	99.3	271.8	62
IICg	140-145	silt loam	-	0.5	0.04	13	-	6.3	-	-	-	-	-	-	-	-
		Eutrophic Peatland - Rich Fen														
Om1	0-30	mesic, wood-moss-sedge	8.7	47.9	2.80	17	6.7	6.3	172.4	22.0	0.8	1.2	1.3	52.9	250.6	78
Om2	30-55	mesic, wood-moss-sedge	11.2	50.7	2.52	20	6.7	6.0	148.0	16.6	0.1	1.6	1.2	60.1	227.6	73
Om3	55-145	mesic, moss-sedge-wood	22.9	45.1	2.38	19	6.7	6.0	177.0	14.3	0.1	3.8	2.7	59.4	257.3	76
Om4	145-160	mesic, sedge	49.9	27.3	1.88	14	6.7	6.4	106.7	8.2	0.1	2.8	1.2	28.8	147.8	80
Oco	160-200	humic, sedimentary	68.2	21.5	-	-	7.0	7.4	-	-	-	-	-	-	-	-

[a] Adapted from the original tables in Turchenek et al. (1984).
[b] Determined with 0.5 N $BaCl_2$-0.05 N triethanolamine at pH 8.
[c] Determined with 0.01 M $CaCl_2$.

released to runoff waters as a consequence of development or other external influences in peatlands.

4.5 ELEMENTAL COMPOSITION OF PEATS

There is generally wide variation in elemental contents of peats. The principal constituents other than carbon, hydrogen, oxygen, and nitrogen are calcium and magnesium. Lucas (1982) has compiled a summary of elemental contents of undisturbed organic soils principally from the United States (Table 4.2). The data likely represent levels in Canadian peatlands as well. Table 4.2 shows ranges of elemental contents for peats as well as typical average percentages for eutrophic and oligotrophic peats.

Except for total C and N contents, very few data are available for Alberta peats. In an early study, Newton (1936) reported some elemental contents of peat samples used for studies of crop growth on organic soils (Table 4.3).

Newton (1936) did not provide information about the actual depths that samples were taken from. The data for N content are similar to data for other Alberta peats (e.g., Table 4.1). Using the reported pH to differentiate oligotrophic and eutrophic peats, P, K, and Ca values are generally similar to the typical percentages of Lucas (1982) shown in Table 4.2. Other data on contents of the major elements can be found in Zoltai et al. (1988).

Factors that influence the chemical composition of peats include botanical origin, the degree of decomposition, and the mineral content of the water associated with formation of peat deposits. The chemical composition is also influenced by sedimentary material deposited by wind or water. Different elements have different vertical distributions in peat profiles due to the above factors as well as factors related to reactivity and mobility of various chemical forms of the elements as a peatland develops.

The distribution of elements in peat profiles has been reported for Scandanavian peatlands by Damman (1978), Sillanpaa (1972), and others. Sillanpaa described the distribution of 13 trace

Table 4.2. Approximate ranges and average in percentages of some elements occurring in undeveloped organic soils.[a]

Element		Percent Range (Oven Dry Basis)	% Typical Average	
			Eutrophic Peats	Oligotrophic Peats
Aluminum	Al	0.01 - 5.0	0.5	0.1
Barium	Ba	0.0006 - 0.3	0.005	
Boron	B	0.00001 - 0.1	0.01	0.0001
Calcium	Ca	0.01 - 6.0	2.0	0.3
Carbon	C	12.0 - 60.0	48.0	52.0
Chlorine	Cl	0.001 - 5.0	0.10	0.01
Cobalt	Co	< 0.0003	0.0001	0.00003
Copper	Cu	0.0003 - 0.01	0.001	0.0005
Hydrogen	H	2.0 - 6.0	5.0	5.2
Iron	Fe	0.02 - 3.0	0.5	0.1
Lead	Pb	0.00 - 0.04	0.005	0.001
Magnesium	Mg	0.01 - 1.5	0.3	0.06
Manganese	Mn	0.0001 - 0.08	0.02	0.003
Molybdenum	Mo	0.00001 - 0.005	0.001	0.0001
Nickel	Ni	0.0001 - 0.03	0.001	0.0005
Nitrogen	N	0.3 - 4.0	2.5	1.0
Oxygen	O	30.0 - 40.0	32.0	35.0
Phosphorus	P	0.01 - 0.5	0.07	0.04
Potassium	K	0.001 - 0.8	0.1	0.04
Silicon	Si	0.1 - 30.0	5.0	0.5
Sodium	Na	0.02 - 5.0	0.05	0.01
Sulphur	S	0.004 - 4.0	0.5	0.1
Zinc	Zn	0.001 - 0.4	0.05	0.005

[a] Adapted from the original table in Lucas (1982)

Table 4.3. Composition of some peat samples from central Alberta.[a]

Peat Sample		Ash %	N %	P %	K %	Ca %	pH
Carnwood	1st depth	4.1	1.07	0.06	0.19	0.64	4.0
Winterburn	1st depth	7.6	1.02	0.05	0.05	2.45	7.0-7.5
	2nd depth	10.4	2.54	0.08	0.04	2.35	7.0-7.5
	3rd depth	37.5	1.75	0.12	0.24	0.76	6.5
Spruce Grove	1st depth	14.0	0.46	0.04	0.34	0.85	4.0
	2nd depth	20.1	0.79	0.06	0.14	3.29	7.0-7.5
Stony Plain	1st depth	21.1	0.89	0.06	0.07	3.39	7.0-7.5
	2nd depth	8.1	1.74	0.09	0.04	2.48	5.5

[a] Adapted from the original table in Newton (1936).

elements in two Finnish bogs in which profiles consisted of *Sphagnum* peat underlain by *Carex*-dominated peat. The general slopes of the elemental distribution profiles were found to be the same for all elements. There was a notable concentration in the surface peat horizons, followed by a decrease which reached a minimum in mid-profile, and a strong increase in the peat/mineral transition zone. This pattern is especially pronounced for the elements Zn, Pb, Mn, Sr, Sn, and V, but not as much for Al, Co, Cr, Cu, Fe, Mo, and Ni. Damman (1978) recognized different patterns of concentration changes with depth in Swedish bogs as follows:

1. Elements that show highest concentrations near the surface and rapidly decrease within the upper 10 to 15 cm (Na and K);
2. Elements that show relatively high concentrations throughout the upper 35 cm of peat and decrease to much lower concentrations further down (Ca, P, Pb, and Mn);
3. Elements that reach maximum concentrations, several times that of living *Sphagnum*, between 20 and 35 cm, and that occur at very low concentrations deeper in the peat (Fe, Al, and Zn); and,

4. Elements that show irregular changes with depth (Mg and N).

Damman (1978) showed that while the distributions of elements in a bog are closely related to water table location and fluctuation, and that K and N (and probably P) are conserved in *Sphagnum* plant material, many elements are removed to a considerable extent before the peat becomes permanently anaerobic.

Distributions of 21 elements in profiles from 5 oligotrophic and mesotrophic peatlands in Quebec and Ontario have been reported by Levesque et al. (1980). Data for bogs in Newfoundland, Ontario, and British Columbia were also reported by Washburn and Gillis Associates Ltd. (1983). There is currently little information about elemental distributions in Alberta peatlands. Zoltai and Johnson (1985) reported the Ca, Mg, K, Na, Al, Fe, and S contents in three profiles sampled from a peatland in the Rocky Mountain House area. The profiles were taken along a gradient consisting of a wooded fen, a bog island within the fen, and the interface of the bog and fen. The bog was slightly higher than the surrounding fen and showed lower levels of most elements in the upper horizon. Below the fen level, the chemical characteristics of the bog peat were similar to those of the fen peat. The data for the eutrophic fen peats provide useful comparison with other reports in the literature which have concentrated on bog peats. Levels of Ca and Mg showed uniformity down the profiles, but increased in basal layers, some of which consisted of marl. Levels of K were high in surface peats. Levels of Na, Al, and Fe showed no definite patterns. However, levels of Al and Na in particular were affected by layers of volcanic ash in the peat deposits. Sulphur content was about four times lower in surface bog peat than in fen peat. Below the fen surface, S levels increased with depth.

Acquisition of baseline data for peat deposits in Alberta will be important for evaluations of pollution potential of nutrient and trace elements in developed peatlands. Little information is currently available, although an extensive sampling and analysis program has been carried out by Forestry Canada and data have recently been published by Zoltai et al. (1988). Information is also available from recent work on the ecology and paleoecology of Alberta peatlands by Chee and Vitt (1990), Nicholson (1989), and Kubiw (1987).

Knowledge of the dynamics and forms of different elements in peatlands, as well as the total quantities, is required in assessing impacts of human activities on peatlands and their associated waters. A thorough biogeochemical review is beyond the scope of this report but some aspects will be discussed in following chapters. Clarke-Whistler et al. (1985) have reviewed levels, forms, and dynamics of some elements and, in addition, have reviewed transformations and mobility of elements through the use of conceptual models. Clymo (1983) has also reviewed elemental contents of peats and some aspects of their biogeochemistry.

4.6 ACIDITY

4.6.1 Nature of Acidity in Peat

The nature of acidity in peats has been reviewed by Walmsley (1977), Clymo (1983) and Gorham et al. (1985). Materials such as soils and peats are considered to be weakly acidic because their deprotonation characteristics are similar to those of weak acids. Weak acids do not deprotonate readily, and their more tightly bound protons are released only to solutions more alkaline than those required to release the protons from strong acids (Bohn et al. 1985). However, acidity in mineral soils changes slowly with time upon addition of base, while true weak acids react almost instantaneously with OH^-. This results from slow liberation of Al^{3+} from clay minerals in soils upon addition of base. The soil acidity released also depends on the salt concentration of the bulk solution. Bohn et al. (1985) suggested that the term 'weak acid' be restricted to those compounds whose acidity is independent of time of reaction and salt concentration. Soil organic materials satisfy this definition better than inorganic soil clays.

From the above review of fundamental acid chemistry, it would appear that peats would act as true weak acids. However, many peats may have high mineral contents and may thus exhibit characteristics of soil clays as well as those of true weak acids. Interactions of organic compounds with Al may also be significant in peat chemical behaviour.

Two categories of acidity can be distinguished in weak-acid systems. Total acidity, a capacity factor, is the acidity which reacts when the solution is neutralized with a strong base such as NaOH. The degree of acidity, an intensity factor, is related to the amount of H^+ in solution and is best represented by the pH. The H^+ in solution may constitute only a small proportion of the total amount of hydrogen available to react with bases.

Concepts of types of acidity in soils may be applicable to peats as well. Total acidity, referred to above, is considered to consist of exchangeable and non-exchangeable acidity. Total acidity refers to the combined total content of H^+ plus the undissociated forms of acidity in the soil. It is quantitatively equivalent to the amount of base required to bring the soil to a predetermined pH value, usually 7.0, 8.0, or 8.2 (Bache 1980).

Exchangeable acidity is the portion of total acidity that can be extracted by a concentrated, unbuffered solution of neutral salt such as KCl or NH_4Cl. Since the salt solution is unbuffered it acquires the pH of the soil or peat so the acidity extracted represents the portion of total acidity which can be extracted at the pH of the soil. In a peat-water sample it would represent those H^+ ions which are already dissociated plus those which readily dissociate upon addition of the salt solution. Non-exchangeable acidity is the total acidity minus exchangeable acidity. It constitutes the major portion of total acidity and exists in soil mainly in undissociated forms (Bache 1979).

Measurements of pH values of solutions are accomplished through the use of colour changes in pH indicator dyes or potentiometric measurement using glass and calomel electrodes (Bache 1979). Peat pH measurements reported in the literature vary depending on the actual material measured and on the method. The glass electrode is most commonly used for accurate pH measurements. It can be applied in determining pH of (1) water from surface pools or pits, (2) fresh peat material, by inserting the electrode into peat in situ or extracted from a pit, (3) water expressed from peat samples, (4) fresh (moist) or dried peat samples immersed in distilled water, and (5) fresh or dried peat samples immersed and equilibrated in salt solutions. Salt solutions may be diluted, such as 0.01N $CaCl_2$, or they may be strong, such as concentrated KCl. Measurements of pH made by pressing a glass electrode into damp peat are usually lower than those of peat water and are difficult to interpret (Clymo 1983). The pH of peat determined in this way has been found to be about 0.5 pH units lower than that of pH of water taken from reference sites (Jeglum 1971). Measurements of pH of peat in salt solutions can be useful for comparative purposes because they essentially measure total exchangeable H^+ (Clymo 1983).

4.6.2 Sources of Acidity

Various mechanisms of H^+ production and consumption in ecosystems are summarized in Table 4.4. Dissolution of CO_2 in bog waters was initially considered to be the main cause of acidity (Villeret 1951, cited by Gorham et al. 1985). Gorham (1956) disproved this and postulated instead that oxidation of organic sulphur compounds in peat to H_2SO_4 was the main source of acidity. Ramaut (1955a, 1955b, cited in Gorham et al. 1985) suggested that living Sphagnum mosses produced soluble organic acids. This hypothesis has recently been revived by Kilham (1982). Gorham and Cragg (1960) suggested that cation exchange of H^+ adsorbed to peat particles for metal cations in atmospheric precipitation might be a significant source of acidity. Clymo (1964) furthered the

Table 4.4. Summary of proton producing and consuming processes.[a]

Proton-producing processes	Example	Proton-consuming processes
(1) Atmospheric input	H^+ (aq(rain/drain water)) = H^+ (aq(soil solution))	(1) Drainage
(2) Assimilation of cations	$M^+ + R_{OH} = R_{OM} + H^+$ M^+ + H.Ligand = M.Ligand + H^+ $NH_4^+ + R.OH = R.NH_2 + H_2O + H^+$	(2) Mineralization of cations
(3) Mineralization of anions	$R.H_2PO_4 + H_2O = R.OH + H_2PO_4^- + H^+$ $R.SO_4 + 2H_2O = R.(OH)_2 + SO_4^{2-} + 2H^+$ $R.NO_3 + H_2O = R.OH + NO_3^- + H^+$ $R.NH_2 + 2O_2 = R.OH + NO_3^- + H^+$ $R.SH + 3/2\ H_2O + 7/4\ O_2 = R.OH + SO_4^{2-} + 2H^+$	(3) Assimilation of anions
(4) Dissociation of acids	$H_2O = OH^- + H^+$ $CO_2 + H_2O = HCO_3^- + H^+$ $R.OH = R.O^- + H^+$ H_n.Anion (aq) = $Anion^{n-}$ (aq) + nH^+	(4) Protonation of anions
(5) Oxidations	$H_2S + 2O_2 = SO_4^{2-} + 2H^+$ $SO_2 + 1/2\ O_2 + H_2O = SO_4^{2-} + 2H^+$ $NH_4^+ + 2O_2 = NO_3^- + H_2O + 2H^+$ $NOx + 1/4\ (5-2x)O_2 + 1/2\ H_2O = NO_3^- + H^+$ $N_2 + 5/2\ O_2 + H_2O = 2NO_3^- + 2H^+$ $Fe^{2+} + 1/4\ O_2 + 5/2\ H_2O = Fe(OH)_3 + 2H^+$ $FeS + 9/2\ O_2 + 5/2\ H_2O = Fe(OH)_3 + SO_4^{2-} + 2H^+$ $FeS_2 + 15/4\ O_2 + 7/2\ H_2O = Fe(OH)_3 + 2SO_4^{2-} + 4H^+$	(5) Reductions
(6) Reverse weathering of cations	M^{n+} (aq) + $1/2nH_2O = 1/2nM_2/nO(s) + nH^+$ (aq)	(6) Weathering of metal oxide components
(7) Weathering of anionic components	$^*NO_{(m+n)}(S) + mH_2O = NO_{(2m+n)}^{2m-} + 2mH^+(aq)$ $Ca(H_2PO_4)_2 = CaHPO_4 + H_2PO_4^- + H^+$	(7) Reverse weathering of anions
(8) Cation/hydronium exchange	M^{n+} + nH.Exch = M.Exch + nH^+	(8) Hydronium/cation exchange
(9) Anion exchange (desorption)	SO_4. Exch + $2H_2O$ = Exch.$(OH)_2 + SO_4^{2-} + 2H^+$	(9) Anion adsorption

[a] Adapted from the original tables in van Breeman et al. (1983, 1984)

ion-exchange theory of bog-water acidity by showing that *Sphagnum* mosses could metabolically generate new exchange sites by synthesizing polygalacturonic acids in their cell walls. In more recent work, Hemond (1980) reported that the low pH of bogs is maintained predominantly by dissociation of weak organic acids produced by decomposition of peatland vegetation. Gorham and Detenbeck (1986) similarly concluded that due to the analytical methods used, inorganic acidity arising from organic S and H_2S oxidation was overestimated in earlier studies and that weak acids now appeared to be the predominant source of acidity in peatlands. Data from Thoreau's Bog at Concord, Massacheusetts strongly implicate fulvic substances as being the major component of coloured, dissolved organic materials responsible for acidity (McKnight et al. 1985).

4.6.3 pH of Peats in Alberta

Values of pH for organic deposits in Western Canada range from about 3.0 to 8.0 (Walmsley 1977). Values reported in Alberta range from 2.8 to 7.8 (Turchenek et al. 1984; Turchenek and Lindsay 1982). The low pH values, less than about 4.0, are characteristic of bogs (see Table 3.1, Chapter 3). Data for central and northeast Alberta found in the references cited above indicate that a considerable proportion of organic soils have pH values in the 5.0 to 6.0 range in the surface soil horizon. These are typical of poor and transitional fens. Values of pH which are near neutrality or higher are characteristic of rich fens and swamps. Rich fens occur commonly in Alberta while swamps appear to be of limited occurrence.

Acidity can change greatly from the top to the bottom of a peat profile. Peat profiles in bogs typically increase downwards from pH 3.0 to 4.0 in the surface layers to about 6.0, (7.0 in some cases) in the bottom layers adjacent to mineral soil. In some bogs the pH remains highly acidic throughout the profile, ranging from about pH 3.0 in the top layers to about 4.0 to 4.5 in the bottom layers. The pH of poor and transitional fens (about 5.0 to 6.0) and of rich fens (about 6.0 to 7.5) is commonly uniform throughout a profile or may increase gradually with depth. However, many fens have low pH (about 4.0 to 5.0) in both peat and pool water in the surface 30 cm or so and change through a narrow transitional zone to somewhat higher pH. Thus, peatlands with poor or intermediate fen characteristics in their surface horizon can overlie materials with rich fen characteristics. In some situations, there is an abrupt increase in pH at a depth where fen peat overlies sedimentary peat which is calcareous due to inclusions of marl or mollusk shells. Very few profiles from swamps and marshes in Alberta have been sampled and analyzed and it is therefore difficult to generalize about their chemical properties. The pH distributions in some profiles representative of peat deposits in central Alberta (Athabasca area) are shown in Table 4.1.

4.7 CATION EXCHANGE PROPERTIES

4.7.1 Cation Exchange Capacity

Peat particulates and colloids, like soils, are characterized by two basic properties that make them chemically active: a large surface area and a surface electric charge. The surface electric charge must be balanced by an equal quantity of oppositely charged counter-ions to satisfy the requirement of electroneutrality. Ion exchange refers to the exchange between the ions balancing the surface change of particles and the ions in solution. Mineral soils are characterized predominantly by permanent charge due to clay minerals and partly by pH-dependent charge from hydrous oxides and organic matter. Charges on hydroxylated surfaces of oxide and hydrous-oxide minerals can be appreciable in some soils. The magnitude of these is variable as it is dependent on pH as opposed to the constant magnitude of charges associated with layer silicate clay minerals. Variable charge is also due to phenolic and carboxylic groups of organic matter in soils (Bache 1979; Gast 1979). The surface electric charge of peats arises mainly from pH-dependent dissociation of the hydroxyl groups on these phenols and carboxylic acids. However, hydrous oxides and clay minerals would contribute some of the charge in humified peats and in mucks. Clymo (1983) has indicated that the term 'cation exchange' is used in a broad sense with peats since processes such as dissociation and chelation are included as well as cation exchange in the strict sense.

The cation exchange capacity (CEC) of peat is a measure of its ability to adsorb cations in exchangeable forms. It corresponds to the negative charge, and is expressed in centimoles of positive charge per kilogram of soil or peat (i.e., cmol(+) kg^{-1}). Due to the variable charge nature of organic materials, the negative charge, and hence the CEC, increases as the pH rises. Helling et al. (1964, cited by Lucas 1982) showed CEC changing with pH as follows (units in cmol(+) kg^{-1}): 73 at pH 3.5, 127 at pH 5.0, 131 at pH 6.0, 163 at pH 7.0, and 215 at pH 8.0. A pH of 8.2 is a norm for CEC and total acidity measurement (Bache 1979). However, various extractants at any chosen pH level can be used to measure CEC. Ammonium acetate buffered at pH 7.0 is another commonly used extractant but, as seen above, the CEC determined would be significantly lower than that determined at pH 8.2. If a neutral salt is used for extraction, the solution attains the pH of the sample and thus estimates the actual surface charge, or exchange capacity of the soil or peat in situ. Thus, it is important to ascertain the CEC determination method in reviewing any CEC data for peats reported in the literature.

The CEC of peats is dependent on the botanical origin of the material. Sphagnum has a relatively high CEC while that of Eriphorum and Carex species is quite low (Clymo, 1983; Puustjarvi 1977). Peats of different degrees of decomposition can have quite similar CEC values. However, data for some Alberta peats tend to show that relatively undecomposed Sphagnum peats have lower CEC values than do moderately to well humified moss-sedge and wood-moss-sedge peats (Table 4.1).

The ion exchange properties and pH values of organic soils in Alberta have been summarized by Holowaychuk and Fessenden (1987) and are presented in Table 4.5. Some data for surface water

Table 4.5. Soil reaction[a] (pH) and cation exchange properties of Organic and Organic Cryosolic soils.

Layer (cm)	$pH(H_2O)$	Cation Exchange Capacity $cmol(+)kg^{-1}$	$cmol(+)L^{-1}$	% Base Saturation	Pool Water pH	Ca + Mg $(cmol(+)L^{-1})$
Organic Soils - Eutrophic (Fens and Swamps)						
0-40	6.0-8.0	120-160	8-10	70-100	6.5-8.0	2-5
40-120	6.0-7.5	120-175	10-18	65-100		
>120	6.0-7.5	120-175	10-18	65-100		
Organic Soils - Mesotrophic (Fens)						
0-40	4.5-6.0	120-160	2-8	25-70	5.0-6.5	0.5-2.5
40-120	5.0-6.5	120-175	4-12	30-70		
>120	5.5-6.5	120-175	6-15	40-75		
Organic Soils - Oligotrophic (Bogs)						
0-40	3.5-5.5	120-160	0-2	10-25	3.5-5.5	>0.5
40-12	03.5-5.5	120-175	2-6	15-40		
>120	4.5-6.0	120-175	4-12	25-70		
Organic Cryosols - Oligotrophic (Bogs)						
0-40	3.5-5.5	125-200	0-6	5-50	3.5-5.0	<0.5
40-80	3.5-5.5	140-200	0-7	5-60		
>80	4.5-6.0	140-200	2-12	15-90		

[a] Adapted from the original table in Holowaychuk (1986).

chemistry are also presented. Examples for specific peatland sites are provided in Table 4.1. The values for CEC are similar in different peatlands and at different depths. The base saturation percentage and contents of Ca and Mg in pool water are directly related to pH of peat or of pool water. Although values for CEC and for content of exchangeable bases in peats are considerably higher than those for mineral soils when compared on a weight basis, they are very low when calculated on a volumetric basis. This is shown in Table 4.5 where CEC data for 1-L volumes are given. On a volume basis, the CEC of peats is similar to that of sandy soils. This has implications, for example, in determining the acid neutralizing capacity of peats in relation to atmospheric inputs of acidity.

4.7.2 Base Saturation

The base saturation is the percentage of the total cation exchange capacity accounted for by the basic cations (Ca^{2+}, Mg^{2+}, K^+, Na^+, NH_4^+). It is clear from the previous discussion of pH dependence of CEC that the pH attained in a measurement method must be specified. For the same peat sample, the percent base saturation will be somewhat lower at pH 8.2 than if measured at a lower pH. The exchangeable cations can be regarded as being available to plants. Thus, the base saturation percentage, as well as the actual value of the ions, can be a useful index of the fertility of a peat soil.

Comparison of the different peatland types in Table 4.5 shows how the contents of exchangeable bases and the base saturation percentages increase from oligotrophic to eutrophic peatlands. There is also a direct relationship between the pH of peat and its base saturation; hence pH is commonly used as an easily measurable index of trophic status.

4.8 REDOX

Chemical reactions involving oxidation and reduction are referred to as redox reactions. In such reactions, an electron donor (the reducing agent) is oxidized and the electron acceptor (the oxidizing agent) is reduced. The intensity of redox reactions is a function of the electrical potential in the reacting system. The potential is measured in volts with that of the standard hydrogen electrode ($2H^+ + 2e^- = H_2$) taken as zero. Oxidizing systems are expressed as having positive potentials, and reducing systems as having negative potentials. A classification of redox potential ranges in soils is as follows (Bohn et al. 1985):

1. Aerated soils, +700 to +400 mv;
2. Moderately reduced soils, +400 to +100 mv;
3. Reduced soils, +100 to -100 mv; and
4. Highly reduced soils, -100 to -300 mv.

Some data for redox potentials in organic soils have been reviewed by Sikora and Keeney (1983). Reducing conditions generally prevail below about 10 cm from a peat surface. Redox potentials have been shown to vary with temperature and they change with water level fluctuation in peatlands. Different chemical processes occur in different ranges of redox potential in reduced systems. Possible systems operating in flooded environments, along with the microorganisms involved, have been summarized by Sikora and Keeney (1983) as shown in Table 4.6. The reactions involved in some of these processes were previously indicated in Table 4.4.

Table 4.6. Possible systems operating in flooded environments as related to redox potential.[a]

System	Redox Potential Range (mv)[b]	Microorganisms Involved
Oxygen disappearance	+500 to +350	aerobes
Nitrate disappearance	+350 to +100	
Mn^{2+} formation	below +400	facultative anaerobes
Fe^{2+} formation	below +400	
Sulphide formation	0 to -150	
Hydrogen, methane formation	below -150	obligate anaerobes

[a] Source: Sikora and Keeney (1983).
[b] Corrected to pH 7 [Eh_7 = Eh - 0.059 (pH - 7)v].

Alkalinity or acidity changes in surface waters resulting from additions of NO_3^- or SO_4^{2-} have been discussed by Schnoor and Stumm (1985). Depending on the redox potential, the acidifying influences of these anions can be reversed by subsequent denitrification or SO_4^{2-} reduction. Lowering water tables in peatlands will change the redox status in the surface peatland thus leading to various chemical transformations. The oxidation of sulphide, in particular, could have a strong acidifying effect on runoff waters. Such transformations are reviewed in greater detail in subsequent chapters.

4.9 BULK DENSITY

Bulk density is the weight of a unit volume of peat. The volume with which bulk density is concerned is that of the solid particles and the pore space. Bulk density is measured by obtaining a sample or known volume and determining the weight after drying.

Bulk density depends on the amount of compaction, botanical composition, mineral content, and degree of decomposition of peat. Values of bulk density range from 0.02 to 0.34 $Mg \cdot m^{-3}$ on a dry basis and from 0.4 to 1.2 $Mg \cdot m^{-3}$ on a wet-volume basis (Walmsley 1977). Values of 0.07 to 0.25 $Mg \cdot m^{-3}$ have been reported by Turchenek et al. (1984) for peats from central Alberta. The bulk density of peat is strongly correlated with ash and fiber contents, and it is, therefore, an indicator of the degree of decomposition as well.

4.10 FIBER CONTENT

An inverse relationship between the content of particles or fibers larger than a particular size and the stage of decomposition of peat forms the basis for classification of peats. A value of 0.15 mm has been arbitrarily chosen to differentiate fiber from smaller particulate and colloidal material. The fiber content is usually determined by wet sieving, either on a fresh peat sample or on a sample that is rubbed or processed in a blender. The rubbing is required to more accurately estimate the degree of decomposition of samples which may be highly humified but which contain fibers that retain their original structure. The Canadian System of Soil Classification uses the rubbed fiber levels to differentiate organic soils. Fiber contents in the entire range of 0 to 100% can be found in Alberta peats. However, contents in the fibric (40 to 100%) and mesic (10 to 40%) ranges are most common while humic peats (<10% rubbed fiber) are relatively uncommon.

Peats have also been fractionated into a number of particle size classes from <75 μm to >2 mm by sieve analysis. Particle size distributions determined in this manner have been found to vary with degree of decomposition and botanical origin of the peat (Levesque et al. 1980). These determinations are also useful in distinguishing between degree of degradation, a physical decomposition feature, and the degree of humification, a chemical decomposition feature.

4.11 POROSITY

The total porosity of a soil or peat is the volume of pore space expressed as a percentage of the volume of pores plus solids. The pores are occupied by air and water, or almost totally by water in the saturated zone of peat deposits. The average porosity of peat is about 92% (Walmsley 1977). Although peats of different degrees of decomposition and botanical origin have similar porosities, their pore size distributions can vary widely. Relatively undecomposed moss readily yields water to drainage because of a predominance of large pores. In decomposed peats, water is held mostly in small pores which are not easily drained. Porosity is thus directly related to water retention and transmission properties. Other aspects of porosity in peats are discussed by Walmsley (1977) and Clymo (1983).

4.12 WATER RETENTION PROPERTIES

Markedly different water holding capacities among peats are related to differences in bulk density, fiber content, and porosity. These in turn are dependent on factors such as degree of decomposition, botanical origin, compaction, and mineral soil content. Water retention or water capacity data for peat can be reported on a weight or volumetric basis. On an oven dry weight basis, values over 3000% have been reported (Walmsley 1977). Typical values reported by Lucas (1982) for various organic soils are:

1. *Sphagnum* peat 1000 to 2000%;
2. Reed-sedge peat 500 to 700%;
3. Wood-sedge peat 300 to 400%;
4. Well decomposed peat 200 to 250%; and
5. Cultivated peats 100 to 250%.

Similar data have been reported for peats in Alberta. For example, peats in the Athabasca oil sands area were found to have water contents ranging from 259 to 2673% (Table 4:7).

Water retention properties and their implications in peatland management for various uses are discussed in greater detail by Walmsley (1977), Clymo (1983), Lucas (1982), and Boelter and Verry (1977). Data for these properties are required for studies of impacts of peatland use on quantity and quality of associated surface waters. For example, lowering of the water table in well decomposed peat deposits would result in less water being drained, and at a lower rate, than if carried out in peatlands with somewhat more fibric peats. It follows that changes in redox potential and other

Table 4.7. Field water content of peats with different degrees of decomposition from the Athabasca area, Alberta.[a]

Degree of Decomposition		No. of Samples	Water Content (%)		
Class	Fiber Content		Range	Mean	CV[b]
Fibric	>70%	14	820-2673	1904	23
Fibric-mesic	41 to 70%	15	598-1907	1271	31
Mesic	11 to 40%	35	611-1330	927	22
Humic	<10%	4	259-522	390	28

[a] Adapted from the original tables in Turchenek and Lindsay (1982).
[b] CV = coefficient of variation (%).

chemical parameters would likewise differ considerably between the two kinds of drained peatlands.

4.13 HYDRAULIC CONDUCTIVITY

The hydraulic conductivity of a material is defined as the velocity of flow across a unit area in response to a hydraulic gradient. Hydraulic conductivity expresses the dynamic aspects of water movement in peat or soil. The term is sometimes confused with permeability which is generally a qualitative expression of how well a material transmits water.

Hydraulic conductivity is an important physical property of peats since it determines the amount of infiltration or runoff of incoming waters, influences groundwater movement, and may indirectly affect regulation of plant root aeration and nutrient availability. Differences in saturated hydraulic conductivity are due primarily to differences in pore size distribution. The total porosity and the size of pores decrease with increase in degree of decomposition. The continuity of pores also changes as peats decompose and become more compacted during the course of peat accumulation. The arrangement of particles also influences vertical and horizontal components of hydraulic conductivity.

Various methods have been used for the determination of hydraulic conductivity. Two common field methods are the auger-hole method and the seepage-tube, or piezometer method, (Rycroft et al. 1975a, 1975b). In the first method, hydraulic conductivity is measured in an unlined well by charting the water level recovery after water is removed. Measurement is carried out similarly in the seepage-tube method, except that a lined well with a cavity at its bottom is used. Thus, these methods measure horizontal hydraulic conductivity and are especially useful where the conductivity may change through the profile. A laboratory method for hydraulic conductivity determinations involves measuring water movement through a soil core and applying Darcy's Law (Boelter 1965). Results of hydraulic conductivity measurements differ depending on the method used (Rycroft et al. 1975a, 1975b). It is therefore necessary to assure that the same method is being used in comparing data.

There is a wide range in hydraulic conductivity among different types of peat, from about 0.08 $m \cdot s^{-1}$ in highly fibric surface *Sphagnum* peats to about 10^{-5} $m \cdot s^{-1}$ in highly humified peats. Walmsley (1977) and Clarke-Whistler et al. (1985) have reviewed various aspects of hydraulic conductivity and provide data for some Canadian peatlands.

In a study by Kong et al. (1980), hydraulic conductivity was found to range from 0.003×10^{-5} to 0.94×10^{-5} $m \cdot s^{-1}$ (Table 4.8). In each of four profiles examined, the hydraulic conductivity was highest in the surface tier and decreased with depth, apparently due to greater peat decomposition at depth.

Data for hydraulic conductivity have recently been obtained for a peatland in the Lesser Slave Lake region of Alberta (Toth and Gillard 1988; Sherstabetoff 1987). The values, presented in Table 4.9, range from 0.13 to 6.30×10^{-5} $m \cdot s^{-1}$. Except for the topmost layer, these data also show a decrease with depth. The data of both Kong et al. (1980) and Sherstabetoff (1987) were obtained by the seepage-tube, or piezometer method, and thus reflect horizontal hydraulic conductivities.

4.14 SUMMARY

Only a few of the chemical and physical properties of peat relevant to understanding of environmental impacts have been addressed in this section. Further information can be obtained from the references cited, particularly Clymo (1983) and Walmsley (1977). An important feature of these peat properties is that many of them are interrelated. Walmsley (1977) and Clausen and Brooks (1980) examined the interrelationships among peat properties and noted that bulk density and fiber content are correlated with the largest number of other physical properties. Fiber and bulk density are, therefore, among the most important properties to measure and a large number of other characteristics can be derived from these. Of the chemical

Table 4.8. Hydraulic conductivity in some Alberta peats.[a]

Peatland Type	Soil Type	Depth (m)	Hydraulic Conductivity[b] ($m \cdot s^{-1} \times 10^{-5}$)
		Mildred Lake	
Bog	Fibrisol	0.45 to 0.65	0.94
		0.70 to 0.90	0.37
		1.05 to 1.25	00.15
Fen	Mesisol	0.30 to 0.45	0.31[c]
		0.45 to 0.75	0.05[c]
		0.75 to 1.20	0.03[c]
Fen	Mesisol	0.40 to 0.60	0.10
		0.60 to 0.80	0.07
		0.80 to 1.00	0.008
		Evansburg	
Bog	Fibrisol	0.30 to 0.40	0.19
		0.50 to 0.75	0.09
		0.80 to 1.00	0.003

[a] Adapted from the original tables in Kong et al. (1980).
[b] Mean value of four measurements.
[c] Mean value of five measurements.

Table 4.9. Hydraulic conductivities measured at the Saulteaux River peatland drainage research site near Lesser Slave Lake, Alberta.[a,b]

Depth (m)	No. of Sites	Hydraulic Conductivity ($m \cdot s^{-1} \times 10^{-5}$) Range	Mean	CV[c]
~0.8	6	0.37-1.70	1.04	53
1.0	24	0.49-6.30	1.78	81
2.0	19	0.33-2.80	1.34	54
2.9	7	0.13-1.80	0.77	93

[a] Adapted from the original tables in Sherstabetoff (1987).
[b] The research area has poor and intermediate fen components; the peat is mesic at most measurement sites and depths.
[c] CV = coefficient of variation.

parameters, pH is related to important properties such as cation exchange capacity, nutrient status, and C and N content of the peat. Where strong relationships among properties can be established, the number of properties measured can be reduced. This can result in significant reduction in amount of effort and resources required to characterize peatlands for land use interpretation or environmental assessment.

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5. HYDROLOGY OF PEATLANDS IN ALBERTA

L.D. Andriashek

5.1 INTRODUCTION

A number of recent review papers address the hydrologic concerns of wetlands and provide a summary of the current knowledge of peatland hydrology in North America (Winter 1981; Carter 1986; LaBaugh 1986). Problems and sources of error in monitoring components of the peatland water budget are highlighted in these papers. Goode et al. (1977) commented that up until 1977, there was practically no research done on water balance of peatlands in Canada. Only a few studies of peatland hydrology conducted since that time in Canada, and in particular Alberta, were found for this literature review .

Complex peatlands, developed on large glacial lake beds and other glacial deposits with low relief, constitute a large proportion of peatlands in North America, yet most hydrologic studies have focused on peatlands within small lake-filled basins that are completely surrounded by uplands (Verry and Boelter 1975; Clausen and Brooks 1983a). The literature reviewed and summarized in this chapter is based primarily on those American studies, as well as from some work in Europe and the USSR. The findings are therefore only partly applicable to our understanding of the hydrologic processes of peatlands in Alberta, where lake-filled (terrestrialized), built-up (paludified) and complexes of these peatlands occur.

5.2 PHYSIOGRAPHIC AND HYDROGEOLOGIC SETTING OF PEATLANDS

An excess of slow-moving water on the land surface is a major factor in peatland development (Boelter and Verry 1977; Ivanov 1981). Excess moisture prevents aeration and leads to a deficiency of oxygen in the soil. This reduces oxidation and enables plant remains to accumulate. With time, the plant remains become compressed and denser under their own weight, and slow decomposition produces peat composed of very fine humic and colloidal particles. This organic material is characterized by having a high capacity to retain water such that volumetric water contents as high as 88 to 97% are attained (Ivanov 1981).

Peatland development can occur on any part of the earth's surface where an equilibrium between moisture influx and efflux results in saturated soil or emergence of groundwater at the soil surface (Ivanov 1981). Where excess moisture conditions exist due to the combined effects of climate, relief, soil, and hydrogeologic setting, peatlands can develop on flat or even convex and sloping surfaces such as those of mountains or uplands. Peat development is not necessarily confined to concave hollows or depressions on the earth's surface, as is commonly believed.

A number of classification schemes have been proposed to divide wetlands and peatlands into hydrogeologic units (Boelter and Verry 1977; Novitzki 1978). Novitzki divided wetlands into four classes based on their physiographic and hydrogeologic setting:

1. Surface-water depression wetlands;
2. Surface-water slope wetlands;
3. Groundwater depression wetlands; and
4. Groundwater slope wetlands.

Surface-water depression wetlands occur where precipitation and overland flow collect in a depression underlain by relatively impermeable sediment (Figure 5.1). Examples of such wetlands can be found in hummocky, knob-and-kettle topography of stagnant ice moraines. In this system water leaves the depression either by evaporation or by slow seepage into the ground. The bottom of the depression is typically above the water table. Ponds, marshes, and seasonally flooded basins are included in this wetland class.

Surface-water slope wetlands are found along margins of lakes and streams that are subject to occasional flooding (Figure 5.2). These wetlands drain readily when flood stages fall. Marshes, swamps, and floodplain forests are included in this class.

Groundwater-depression wetlands develop where hollows on

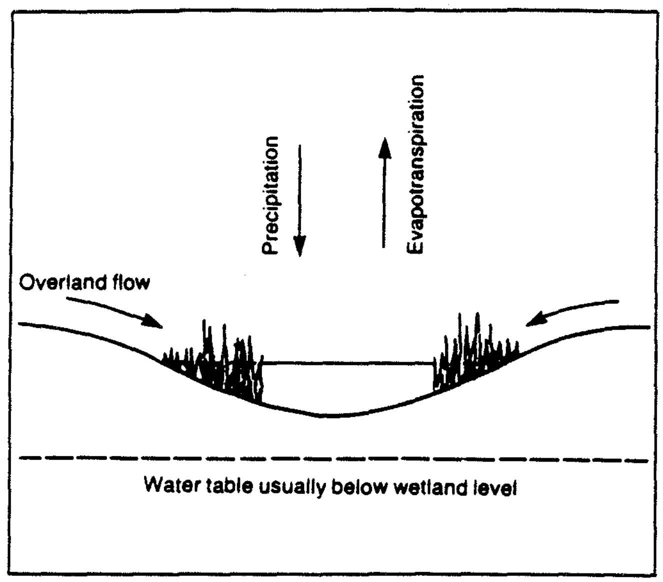

Figure 5.1. Hydrologic characteristics of surface water depression wetlands. (Adapted from Novitzki 1978).

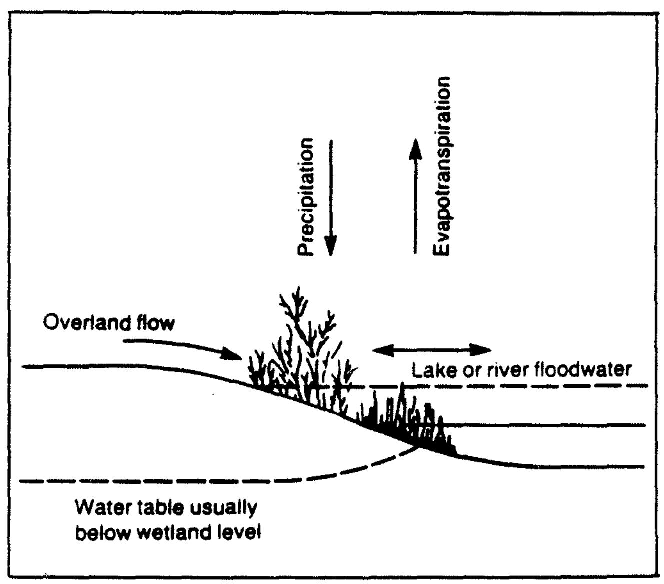

Figure 5.2. Hydrologic characteristics of surface water slope wetlands. (Adapted from Novitski 1978).

the ground surface intercept the groundwater table (Figure 5.3). The depression receives input from precipitation, overland flow, and groundwater flow, but it has no surface drainage away from the site. Precipitation and overland inflow are usually sporadic, whereas groundwater inflow, though it may not be a significant contributor, may be continuous during dry periods. An example of such a setting is a flat, sandy outwash plain where the groundwater table may be relatively high. Marshes, sedge meadow, and fens develop in such systems, and if the peat cover is sufficiently thick to isolate the vegetation from the groundwater, bogs will develop (Figure 5.4).

Groundwater-slope wetlands develop where groundwater discharges as springs or seeps at bottoms of hills or hill slopes (Figure 5.5). In this system groundwater continually recharges the wetland and surplus water drains downslope away from the site. Groundwater can contribute as much as 50% to the water budget of the wetland. The type of wetland that develops is a function of the amount of groundwater influx, the ease of drainage from the site, and the chemistry of the water.

5.3 HYDROLOGIC CLASSIFICATION OF PEATLANDS

Peatlands can be classified by the nature of water input into the system. Recharge to peatlands can be from two sources; meteoric recharge, which is water from the precipitation, fog, or dew, or telluric recharge, which is water from the ground (Dooge 1975; Boelter and Verry 1977; Ingram 1983). These sources differ not only in origin, but in chemistry, seasonality, and their ability to be measured. Based on these differences, peatlands in Alberta can be divided into two hydrologic classes: bogs and fens. Bog peatlands occupy areas of divergent or radiating drainage patterns, such as water table divides. Surfaces of bogs reflect this in that they are convex in cross-section in one or more directions. Similarly, water tables within bogs are also convex or mounded, and because of this prevent inflow of nutrients from mineral soils. Direct precipitation is the only source of recharge water to the bog and it must exceed

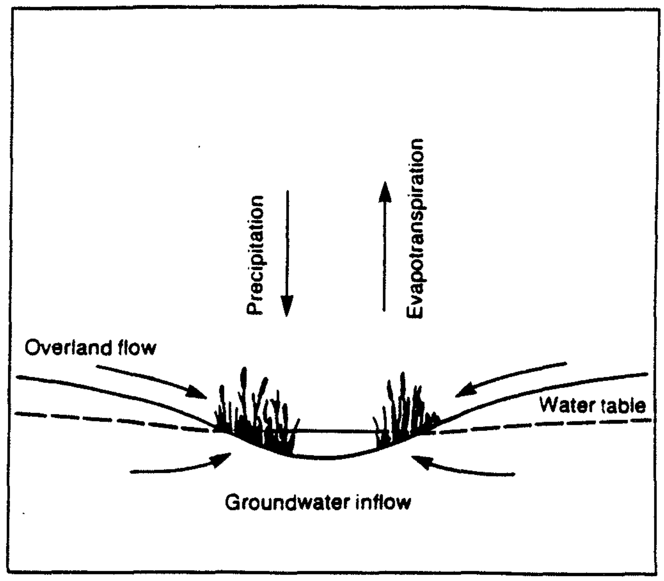

Figure 5.3. Hydrologic characteristics of groundwater depression wetlands. (Adapted from Novitski 1978).

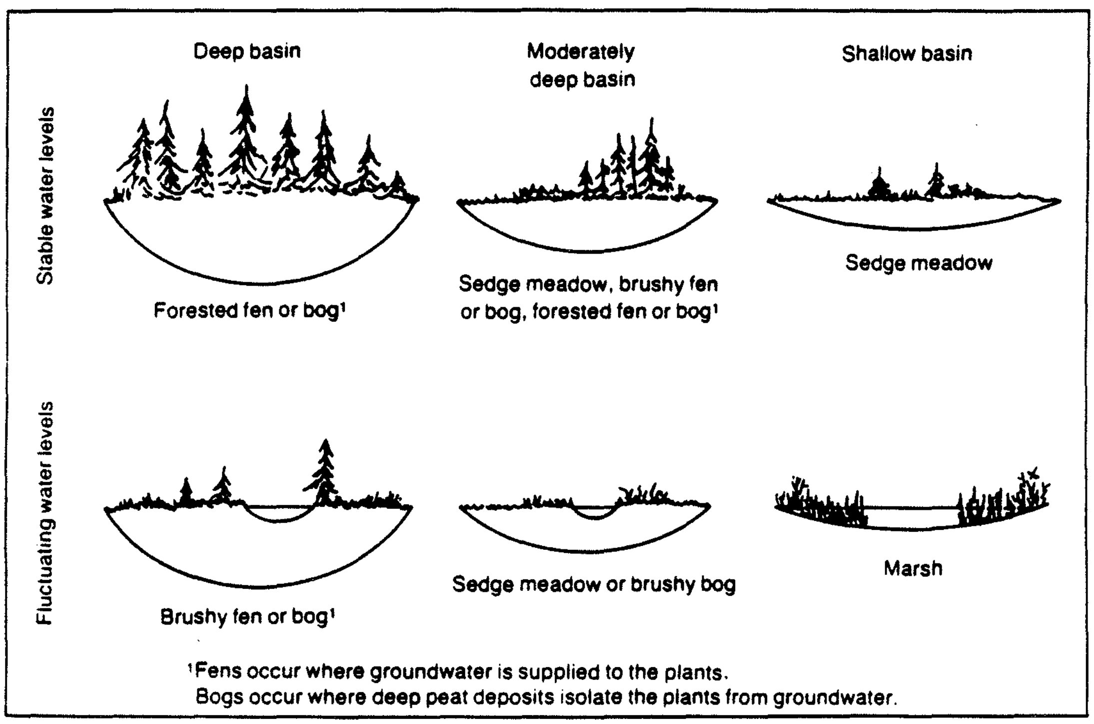

Figure 5.4. Different plant communities in groundwater depression wetlands related to basin depth and groundwater level fluctuations. (Adapted from Novitski 1978).

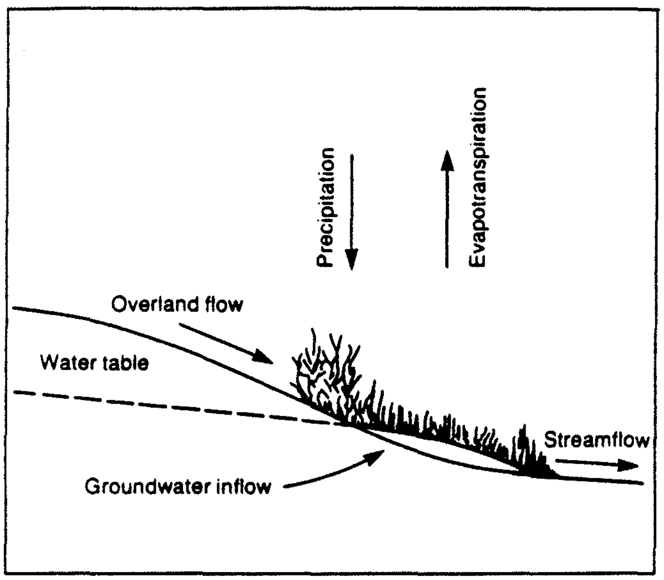

Figure 5.5. Hydrologic characteristics of groundwater slope wetlands. (Adapted from Novitski 1978).

evapotranspiration in order to provide sufficient water to maintain a raised water table (Boelter and Verry 1977). Otherwise, bogs will be absent in climatic zones where there is a deficiency in moisture, and only fens, which are fed by telluric waters, will develop. Further information about the nature of peatlands in relation to water sources can be found in Chapter 3.

Fen peatlands are found parallel to long slopes and tend to display concave cross-sections. Water inflow is confluent into the fen basin and this is commonly reflected in vegetation patterns. Fens can be recharged by direct precipitation, surface runoff, and groundwater discharge; therefore, they are better supplied with nutrients which support more diverse plant growth such as grasses, sedges, and herbs. Fens sometimes develop in artesian sites where upwelling water can form a dome or mound in the groundwater table (Winter and Carr 1980, cited in Verry and Timmons 1982). These sites can remain unfrozen in winter.

There is a scarcity of reliable data on the amount or input rate of telluric water in peatlands. Ingram (1983) commented that most hydrologic studies have focussed on the measurement of local precipitation as a means of indicating recharge to a peatland. This is probably because monitoring groundwater input is much more difficult. As a result, most hydrologic water balance studies have focussed on raised bogs where the only source is from precipitation. Bay's (1970) study showed that there is a dependent relationship between water-table levels in fens and levels in the upland catchment area, thus providing evidence that fens are recharged by surrounding uplands.

There is a natural progression from fens to bogs in the evolution of peatlands. In part this is because continued accumulation of organic material causes the water table to rise accordingly and eventually become isolated from the regional flow system. Furthermore, in large peatlands, mineral depletion in the groundwater occurs because of cation exchange as the groundwater flows from the source (Boelter and Verry 1977; Walmsley 1977). It is probable therefore that most large peatlands are likely a composite of fens and bogs. Siegel's (1983) model shows that in some of these large complexes, fens can be recharged from groundwater mounds beneath raised bogs.

5.4 THE HYDROLOGIC CYCLE AND WATER BALANCE EQUATION OF PEATLANDS

The components of the hydrologic cycle are shown schematically in Figure 5.6. In a general hydrologic cycle, precipitation enters the system in the form of rainfall or snowmelt and is transmitted to stream channels by quickflow (overland flow, pipe flow, rapid seepage, and interflow, as well as direct precipitation onto the channel) and by baseflow (sustained groundwater discharge below the basal soil horizon) following infiltration. Quickflow is the major component of open channel flow during snowmelt and heavy rainfall.

Input of water into a peatland may include only direct precipitation or components of both quickflow and baseflow. Water leaves the hydrologic system by evapotranspiration and streamflow.

In peatlands, water which does not discharge through evaporation or deep seepage (groundwater recharge), nor contributes, to the long-term storage of the peatland, discharges through open channels. At any given time water is stored in the system by the interception of precipitation on plant surfaces, as ponded water on the surface, as soil moisture in the unsaturated zone, as groundwater in the saturated zone, or as water within the stream channel.

An accurate water balance is critical in understanding of the input and output of nutrients in a peatland (LaBaugh 1986). The various input and output fluxes of a peatland water budget can be classified in numerous forms of a water balance equation (Burke 1975; Dooge 1975; Carter et al. 1978; Verry and Timmons 1982). The most simplistic form of the equation can be expressed as:

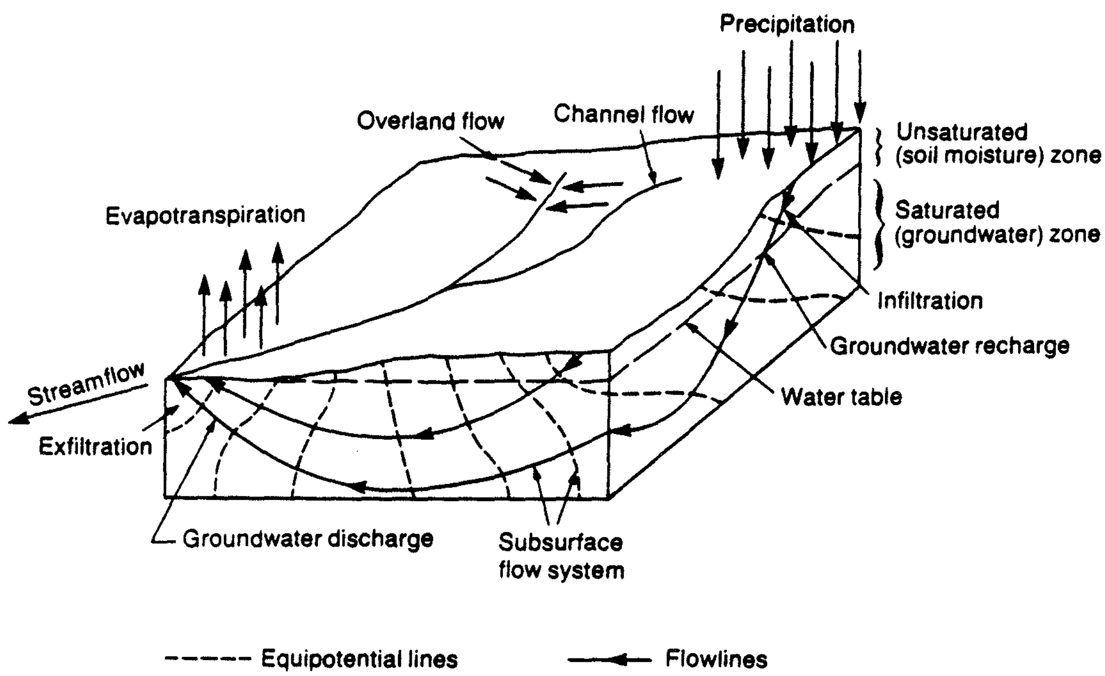

Figure 5.6. Schematic representation of the hydrologic cycle. (Adapted from Freeze and Cherry 1979).

water input - water output =
change in water storage of peatland

Dooge's (1975) equation is one of the more comprehensive and is chosen to represent a detailed hydrologic balance equation for a peatland:

$$\Delta S = PR + SI + GWI + NCI - E - SO - GWO - NCO$$

where, ΔS = change in storage in the peatland for some time interval (includes surface storage, field moisture storage, groundwater storage and channel storage.

PR = precipitation

SI = streamflow input

GWI = groundwater input

NCI = nonchannelized surface water input (interflow)

E = evapotranspiration

SO = streamflow output

GWO = groundwater output

NCO = nonchannelized surface water output.

LaBaugh's (1986) review paper discusses the components of the hydrologic balance equation, with emphasis on Winter's (1981) rigorous evaluation of errors associated with measuring each component and the effect that such errors have in balancing the equation. In the ideal water balance study, every component of the equation would be measured independently, and then inserted into the equation (Dooge 1975). Any deviation from '0' would be a residual term 'b' which can be attributed to errors in measurement;

$$b = PR + SI + GWI + NCI - E - SO - GWO - NCO - \Delta S$$

This ideal case, where every parameter can be measured in the field, is rarely achieved. A summary of methods applied in major studies in northern United States and the United Kingdom shows that in almost every study some component is either ignored, considered to have negligible importance, or is estimated as a residual (Table 5.1). This residual 'b', however, can consist entirely of errors of measurement rather than a true value of the estimated component in the equation (Winter 1981).

Most hydrologic studies of peatlands have not been designed to be comprehensive, partially because of limitations in data acquisition or limitations in the data themselves. Yet, LaBaugh (1986) comments on the emphasis that numerous workers place on the data to calculate and interpret water and chemical input-output balances. This practice of estimating unmeasured components from the differences between measured inputs and outputs is likely to continue however, because of the difficulty in acquiring data for some elements of the water budget.

5.5 EFFECTS OF EVAPOTRANSPIRATION ON WATER TABLE AND RUNOFF DISCHARGE

Evapotranspiration from peatland catchments, and specifically raised bogs, is generally greater than from upland catchment areas of comparable size but without peat development (Baden and Eggelsmann 1966, cited in Ingram 1983). Peatlands that have a high water table produce the greatest amount of evapotranspiration. As much as 65% of water in a peatland leaves as vapour (Boelter and Verry 1977; Verry and Timmons 1982), and as much as 90% of this occurs as transpiration during the growing season (Eggelsmann 1964, Bavina 1967, Bay 1968, cited in Clarke-Whistler et al. 1984).

There are conflicting opinions regarding the depth of the water table at which evapotranspiration is maximum. Boelter and Verry (1977) suggested that this occurs in open bogs when the water table is 10 cm below the surface of depressions on the bog surface. At this depth the evaporative surfaces are greatest and the root systems are well aerated. Higher water tables cause flooding and

Table 5.1 Summary of methods used to obtain water budget data for selected wetland studies.[a]

			Atmospheric Water		Surface Water				
Site	Wetland Type	Reference	Precipitation	Evaporation	Stream-flow In	Stream-flow Out	Overland Runoff	Wetland Water State	Ground-water
Rough Sike, United Kingdom	Pennine Moorland	Crisp (1966)	OG[b]	P-SO	NA	RG	NC	NC	NC
Heron Pond Illinois	Alluvial Cypress Swamp	Mitsch et al. (1979)	OG (1 km)[c]	DS,PAN (24 km)	RG	RG	DS	RG	RES
Control Watershed 2, Minnesota	Upland-Peatland	Very and Timmons (1982)	OG	RES	NA	RG	HS	RG	OM
Thoreau's Bog Massachusetts	Floating-Mat Sphagnum Bog	Hemmond (1980)	RG (4 km)	HS	NA	OM	HS	RG	OM
Great Sippewisset Marsh, Mass.	Tidal Salt Marsh	Valiela, et al. (1978)	OG	NC	NA	OM	NC	OM	DIL
Three Areas in North Dakota	Prairie Potholes	Eisenlohr (1972)	RG	MT,RES	NA	NA	DS	RG	RES
Cottonwood Lake Area, North Dakota	Prairie Potholes	Winter and Carr (1980)	RG	MT	NA	NA	NC	OG,RG	FN

[a] Adapted from the original table in La Baugh (1986).
[b] Explanation of table notation:
OG = Observer read gage, noncontinuous record.
P-SO = Difference between precipitation input and streamflow output; calibrated against lysimeter and Penman equation.
NA =Does not occur.
RG =Recording gage, continuous record (flume, weir, stilling well, etc.).
NC =Not considered.
DS =Calculated from change in wetland stage.
PAN =Evaporation pan.
RFS =Calculated as the residual of the water budget.
HS =Calculated by hydrograph separation technique.
OM =Occasional measurement.
DIL =Calculated from dilution of seawater by ground water to account for salinity at ebb tide.
MT =Mass-transfer method.
FN =Flow-net analysis with water table wells and piezometer nests.
[c] Number in parentheses following notation indicates distance of gage from site.

reduce the evaporative moss surface, which results in less transpiration. Virta (1966, cited in Ingram 1983) found that evapotranspiration is highest when the water table is about 2 cm from the surface, and that it declines steeply as the water table falls to a depth of 10 cm. At this depth evapotranspiration stabilizes at a rate equivalent to about 60% of the evaporative rate when the water table was at 2 cm. Virta attributed this rapid decrease to the greater path length for capillary rise as the water table falls, and to the decrease in hydraulic conductivity as water drains from the larger pores near the surface. With a water table drop of 30 cm from the surface, the evaporation rate is reduced to 75% of its rate at the surface (Williams 1970). Below this depth the capillary fringe of *Sphagnum* stems is exceeded and the moss dries out (Boelter and Verry 1977; Romanov 1968, cited in Ingram 1983). These may be replaced by herbaceous vegetation which is more adapted to the altered moisture regime (Bavina 1967, cited in Ingram 1983).

Fens show higher rates of evapotranspiration than do bogs, mainly because minerotrophic waters in fens can support larger and more diverse plant species that use more water. As well, continual inputs of groundwater in fens reduce water table fluctuations and maintain higher water tables. A tentative generalization is that in treeless bogs the actual evapotranspiration is approximately equal to potential evapotranspiration (as determined by the Penman or Haude methods), whereas in fens the actual evapotranspiration exceeds potential evapotranspiration by about 1.4 times (Ingram 1983).

Transpiration can significantly reduce stream discharge from a raised bog during the intense summer growing period, even though precipitation can be great during that time. Figure 5.7 shows that following a storm rainfall during the active growing season, recession curves for runoff discharge show accelerated rates of decline during daytime periods caused by increased transpiration (Bay 1969). In a wooded peatland transpiration can have a marked diurnal effect on the water table, in some cases causing two periods of rapid

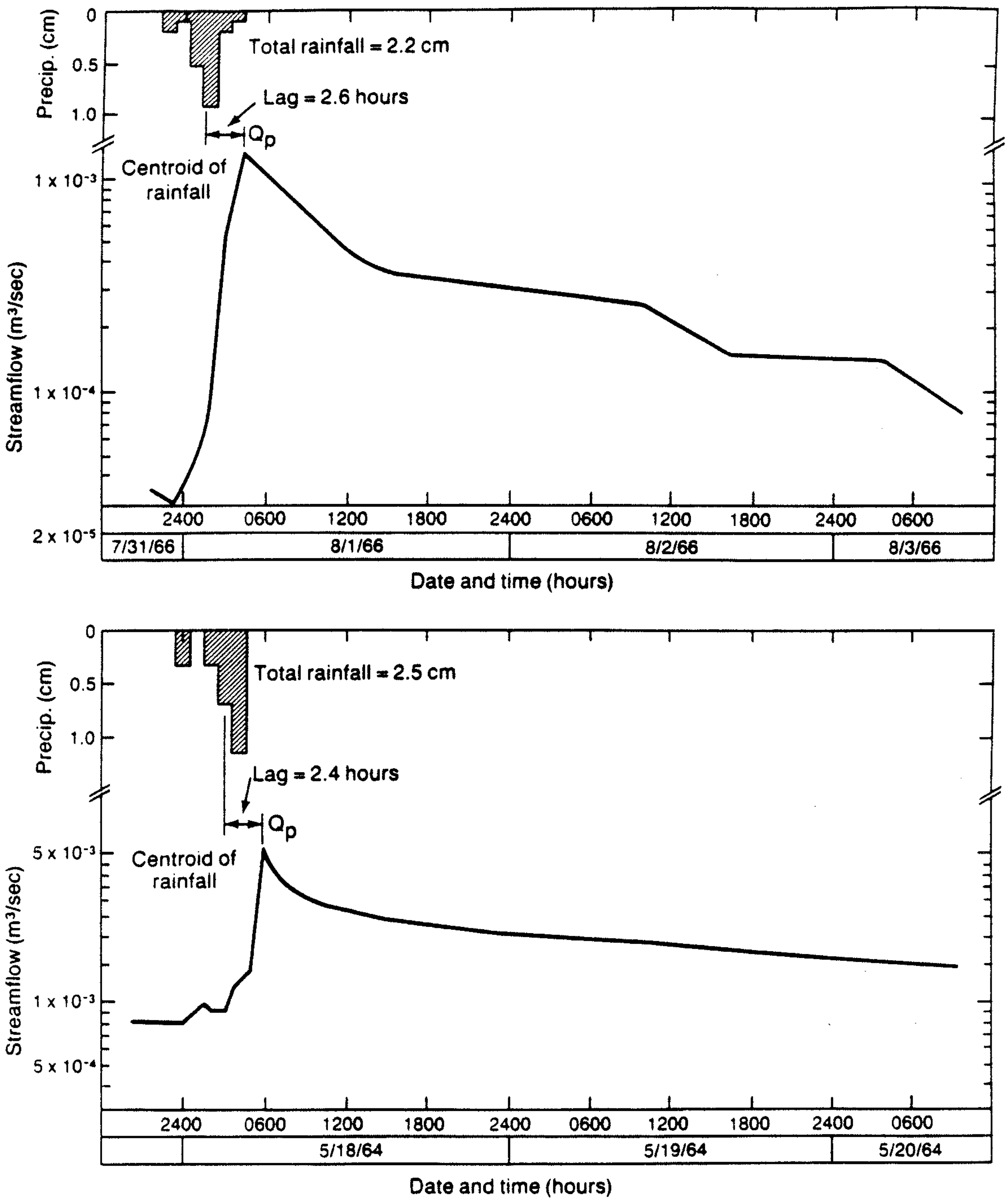

Figure 5.7. Hydrographs and accompanying hyteographs for (top) a growing season storm (2.5 cm) and (bottom) a dormant season storm on a perched bog watershed. (Adapted from Bay 1969).

decline per day corresponding to the forenoon and afternoon (Heikurainen 1963, cited in Ingram 1983).

Goode et al. (1977) cautioned that most of the work in peatland hydrology has been done in the north-central states of the USA where potential evapotranspiration rates are much higher than in the north. Conclusions regarding evapotranspiration rates and hydrologic budgets based on those studies may not apply everywhere in Canada, although for most of Alberta they may be valid because potential evapotranspiration rates are also high (Alberta Soils Advisory Committee 1987; Government and University of Alberta 1969).

5.6 WATER YIELD AND STORAGE CAPACITY OF PEAT

5.6.1 Water Yield

5.6.1.1 Acrotelm. Most water and heat exchange occurs in the upper active layer of the peat horizon where living plants and undecomposed plant fragments are present and where the peat is periodically aerated and not saturated (Dooge 1975; Goode et al. 1977; Ivanov 1981). Most studies of peatland hydrology have concentrated on this active layer, referred to as the 'acrotelm', because of its hydrodynamic properties (Chason and Siegel 1986). Ivanov (1981) characterizes the acrotelm by the following:

1. Intense exchange of moisture with the atmosphere and surrounding areas.
2. Frequent fluctuations in water table levels and moisture content.
3. High values of porosity, hydraulic conductivity (3 to 5 orders of magnitude higher than the catotelm; see data for Alberta in Tables 4.8 and 4.9, Chapter 4) and water yield, and rapid decline in these values with depth. This causes lateral flow to be greater than vertical seepage.

4. Periodic aeration which has the effect of voiding pores of water and lowering the water table.
5. Abundance of aerobic microorganisms which accelerate plant decomposition and transformation into peat.
6. Presence of a living cover at the surface.

The acrotelm represents a relatively thin unit in the peat horizon, ranging in thickness from 7 to 8 cm in fens, to 60 to 70 cm in bogs (Ivanov 1981; Chason and Siegel 1986). Where the surface of the peatland is relatively uniform, the average thickness of the acrotelm ranges between 30 and 70 cm. Where there is significant micro-relief, such as hummocks and elongated ridges and pools, the thickness of the acrotelm is measured from the stable elements of the microrelief such as the ridges and elevations, rather than pools or depressions where surface water may be present (Ivanov 1981).

The capacity of the active layer to serve as a temporary storage reservoir is not only a function of its total pore space, or porosity, but also of the pore size distribution and degree of connectivity. These properties determine the proportion of pores that will drain at a given water potential. An active layer with a high proportion of large pores will release water more easily than a layer with either a low total porosity, or a high proportion of small pores in which the water is retained by matric forces. The acrotelm may contain 90% of water by volume, and release as much as 80% of it to drainage (Boelter 1964, cited in Walmsley and Lavkulich 1975). Table 5.2 shows that for *Sphagnum* peat, increased compression and humification with depth cause a decrease in the effective porosity, and this reduces the coefficient of water yield from about 0.85 to 0.08 (Boelter 1964, 1968, 1975 and Paivanen 1973, cited in Ingram 1983).

5.6.1.2 Catotelm. Increased humification with depth results in closer packing of solid residues and this serves to reduce the size and connectivity of the pores in the anaerobic, saturated layer,

Table 5.2 Water yield as function of humification and bulk density of a *Sphagnum* peat.[a]

Bulk Density ($Mg \cdot m^{-3}$)	Approximate Humification (Van Post Scale)	Y-Yield[b]
0.05	H1	0.60
0.10	H4	0.36
0.15	H6	0.22
0.20	H9	0.18

[a] Adapted from the original table in Ingram (1983) as modified from Paivanen (1973).

[b] Y = specific yield, the ratio of (1) the volume of water that, after being saturated, a soil will yield by gravity to (2) the soil's volume.

referred to as the catotelm. Ivanov (1981) characterized the catotelm by the following:

1. It lies below the water table.
2. Moisture content is near maximum and is nearly constant.
3. Very slow exchange of water with underlying mineral-soil substrate, and area surrounding peatland.
4. Very low hydraulic conductivity; difference of 3 to 5 orders of magnitude compared with overlying acrotelm (see data for Alberta in Tables 4.8 and 4.9, Chapter 4), and equal to or less than that of glacial till or lacustrine clay (Boelter 1975; Cherry et al. 1971, cited in Siegel 1983).
5. No aerobic microorganisms, and reduced amount of other kinds in comparison with the acrotelm.

The catotelm constitutes the main body of a peat deposit, and most of the water that is stored within a peatland resides in this lower, inert layer (Walmsley and Lavkulich 1975; Ivanov 1981; Ingram 1983; Siegel 1983). There are no average thicknesses for the catotelm; thicknesses from a few cm to as much as 20 m have been recorded (Ivanov 1981). Moisture content values as high as 91 to 98% are recorded for the catotelm but only 10 to 15% of this water is released to drainage (Boelter 1964, cited in Walmsley and Lavkulich 1975; Lundin 1964, cited in Ivanov 1981). Lateral flow within the catotelm is considerably reduced and may constitute less than 1% of the total runoff from the peatland. This is because the greater degree of decomposition with depth results in a greater number of very small particles which increase the area of interface between water and solid particles and reduce the hydraulic conductivity. Thus, the catotelm functions in a water-saturated state in which hydrological properties vary little with time. Water is stable and not subject to short-term fluctuations, and any changes which do occur happen very slowly and are not a reflection of hydrometeorological conditions.

A recent paper by Chason and Siegel (1986) challenges some of the conclusions of previous hydraulic conductivity studies of peat. Their research on peat at depths below 1 m in northern Minnesota indicated that hydraulic conductivity values for the catotelm are two to six times higher than previously reported in the literature. Further, the authors saw no correlation of hydraulic conductivity to depth or to degree of decomposition, other than a decrease in value at the acrotelm - catotelm boundary. They also determined that in their study area the horizontal conductivity of the catotelm is on average about two orders of magnitude greater than vertical hydraulic conductivity, which suggests that lateral flow of water can be considerably greater than was previously described.

5.7 PEATLAND WATER TABLES

Establishing the water table level becomes critical in determining the water balance of a fen or bog because variations in the water table reflect changes in groundwater storage and soil moisture storage between the surface and the water table (Dooge 1975). Furthermore, evaporation and runoff are both affected by variable depths to the water table.

Consistently high water tables are one of the most distinctive hydrologic features of peatlands (Ingram 1983). This unique aspect of peatland water tables is attributed to the regulatory effect that the hydraulic conductivity of peat has in maintaining a stable water table (Ivanov 1981). Because of the very high conductivity of the upper part of the acrotelm, excess snowmelt or rainfall rapidly leaves the peatland through horizontal flow, ensuring that water tables do not remain at peak levels for extended periods. Conversely, if precipitation is reduced or ceases altogether, horizontal drainage stops entirely and the peatland drains only by vertical seepage. Thus, water table fluctuations are small and relatively constant despite seasonal changes in precipitation input. Minerotrophic fens, which have sustained yearly groundwater input, characteristically have high water tables and show less water table fluctuation than in bogs. Groundwater input is not a factor in bog water tables, hence fluctuations in the water table relate more to current climatic conditions.

The response of the water table to input from precipitation depends on the initial level of the water table in the peat profile. The response will be considerably greater in the catotelm than in the porous acrotelm because of the lower hydraulic conductivity and storage capacity in the catotelm. Bay (1970) summarized a comparison of water table responses to precipitation, at extreme water table positions in the peat profile (Table 5.3). A falling water table causes the peat to compress, resulting in reduced effective porosity and lower moisture content. Consequently, storage capacity is reduced and any input of precipitation will cause a rapid rise in the water table (high regression coefficients, Table 5.3). Conversely, a rising water table creates a buoyant effect on the peat, thereby increasing effective porosity and storage capacity (Ivanov 1981). Therefore, at high water table levels an equivalent amount of precipitation will cause less of a rise in the water table (low regression coefficients, Table 5.3).

If climatic and peat morphologic conditions are stable, the depth to the water table depends solely on the mean hydraulic conductivity of the acrotelm, the thickness of the acrotelm, and the mean intensity of evaporation (Ivanov 1981). This enables water table levels to be computed directly from meteorological data and offers the advantage that runoff can be predicted solely from water table levels in extensive areas of peatlands where direct monitoring can be difficult (Novikov 1964).

Table 5.3. Comparison of water level response to precipitation on three bogs with water tables at high and low extremes.[a]

Bog	Bulk density ($Mg \cdot m^{-3}$)	Regression equation[b] Y = water level response(cm) X = precipitation (cm)
	High water table	
1	.02 to .06	Y = 1.51X - .21
2	.02	Y = 1.49X - .04
3	-	Y = 1.34X - .02
	Low water table	
1	.08 to .12	Y = 2.51X + .43
2	.06 to .08	Y = 3.47X + .25
3	-	Y = 2.59X - .22

[a] Adapted from the original table in Bay (1970).

[b] Number of storms ranged from 12 to 29 for any one equation.

5.8 WATER STORAGE AND RUNOFF FROM PEATLANDS

5.8.1 Storage

The saturated soil zone is by far the most important location for water storage in peatlands. Boelter (1968, cited in Dooge 1975) suggested that an approximation of storage in peat can be made by calculating the difference between the moisture content at saturation and the moisture content at field capacity. As discussed, most changes in peatland water storage and yield occur in the upper part of the acrotelm where the effective porosity and hydraulic conductivity are greatest. As heavy precipitation or rapid snowmelt raises the water table, the shallower and more permeable regions of the acrotelm become involved in seepage flow. The acrotelm is thus an effective aquifer, with a high capacity to transmit storm- or melt-water. Factors that can influence the quantity of water released from a peatland following high levels of precipitation include:

1. Initial moisture capacity of different layers of peat;
2. The distance which the water table declines from its initial position, measured from surface of the peatland;
3. Distribution of the active porosity in the peat above the water table - this is determined by the structure of the peat matrix and the quantity of immobilized water in it; and
4. The compressibility of the kinds of peat which make up the deposit (Ivanov 1981).

Frozen water stored in palsas and peat plateau along the southern fringe of the permafrost zone is one aspect of water storage that is unique to peatlands in northern climates. Palsas and plateaux generally form in bogs where winter freezing causes ice lens growth, commonly along the peat - mineral soil boundary (Zoltai and Tarnocai 1971, 1975). Palsas can rise as much as 5 m above the surrounding peatland surface and extend as much as 100 m in length and breadth. Peat plateaux are much larger than palsas in length and breadth, but they rise only about 1 m above surrounding, unfrozen peat. The peat cover above a palsa is essentially the same thickness as that of the flat-lying peat nearby; the relief of the palsa is due primarily to the expansion by the growing ice. Palsa and plateaux development is due mainly to the insulating effects of *Sphagnum* moss (Ingram 1983). Initial ice-lens growth causes the palsa surface to rise above the local peatland and this results in the surface peat being better drained in summer and therefore drier. This dry peat has a greater insulating effect, which enables the ice lens to persist throughout the summer. Autumn precipitation rewets the peat and this, combined with the enhanced cooling of the elevated surface of the palsa by wind and reduced snow cover, causes the thermal conductivity of the peat to rise more rapidly than the surrounding peat. Ice growth is thus further enhanced. Eventual oxidation and wastage of the overlying peat ultimately exposes the ice and the palsa collapses.

In summary, water table movement is confined to the active layer which represents generally a thin layer of the peat profile. This thin layer represents the capability that the peatland has for storing easily available water, which implies that peatlands have a limited capacity to store or release water in the short term. In terms of overall potential to store water, catchments with cleared and developed peatlands, or with no peatlands at all, have much lower water table levels and therefore have greater capability for short-term storage than do peatlands where the water table is at or very near the surface.

5.8.2 Runoff

Fluctuating water table levels in peatlands significantly impact runoff discharge, as illustrated in Figure 5.8 (Bay 1969). Under conditions of low water table levels, a peatland has a greater storage capacity overall, and thus yields significantly less runoff than if the water table level were high. Bay (1969) showed that this

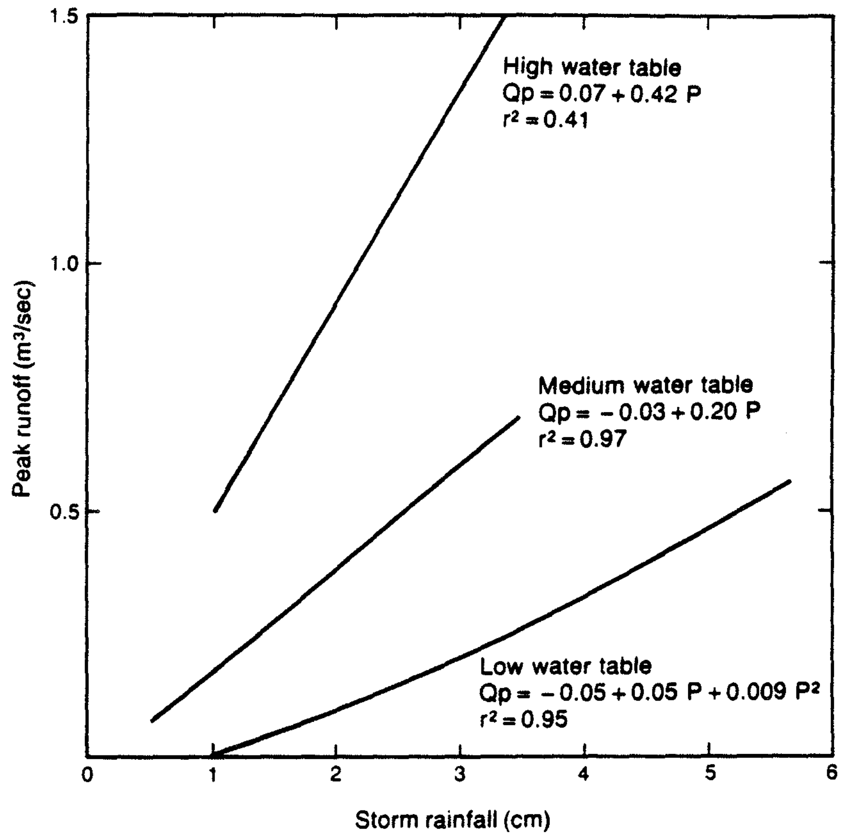

Figure 5.8. Relation of peak flow (Qp) to storm rainfall (P) and water table position in a bog. High water table conditions existed when water tables were fluctuating above the average moss surface and within the hummock and hollow micro-topography. Low water table values were computed when water tables were greater than 15 cm below the average low hollow elevation. (Adapted form Bay 1969).

relationship between water table height and runoff is exponential; a unit rise in an initially high water table results in increasingly more runoff because of the greater effective porosity and hydraulic conductivity of the upper peat profile (Figure 5.9). Conversely, when water table levels are low, considerably less runoff results from an equivalent drop in the water table level because water is retained in the less permeable part of the peat column.

Bogs have a certain depth at which any further lowering of the water table causes runoff to cease entirely. Below this depth, any further lowering of the water table level is due only to evapotranspiration (Bay 1968). Studies in northern USA, USSR, and Scotland have shown that this critical depth lies between 25 and 40 cm from the surface, essentially the base of the acrotelm.

5.8.3 Seasonal Trends in Peatland Water Tables and Effects on Runoff Discharge

Stream discharge is the net result of all processes in the hydrological cycle of a peatland catchment area. Peatlands, because of their flat topography and high water-storage capacity, have been commonly considered as large reservoirs that regulate streamflow throughout the year by storing snowmelt and rain waters during wet periods, and then releasing this water slowly to sustain runoff during drier periods (Bay 1969; Verry and Boelter 1975, 1978; Ingram 1983). Most studies have shown that this is only partly true; peatlands do have a high storage capacity overall, but a number of factors cause seasonal runoff rates to fluctuate significantly. As discussed below, the net effect of peatland growth in a drainage basin is to increase differences in, rather than regulate, baseflow discharge (Verry and Boelter 1978; Novitzki 1978).

5.8.4 Discharge From Raised Bogs

Hydrologic studies of forested peatland watersheds in northern Minnesota (Bay 1969; Verry and Boelter 1975; Boelter and Verry 1977) show that there are significant differences between

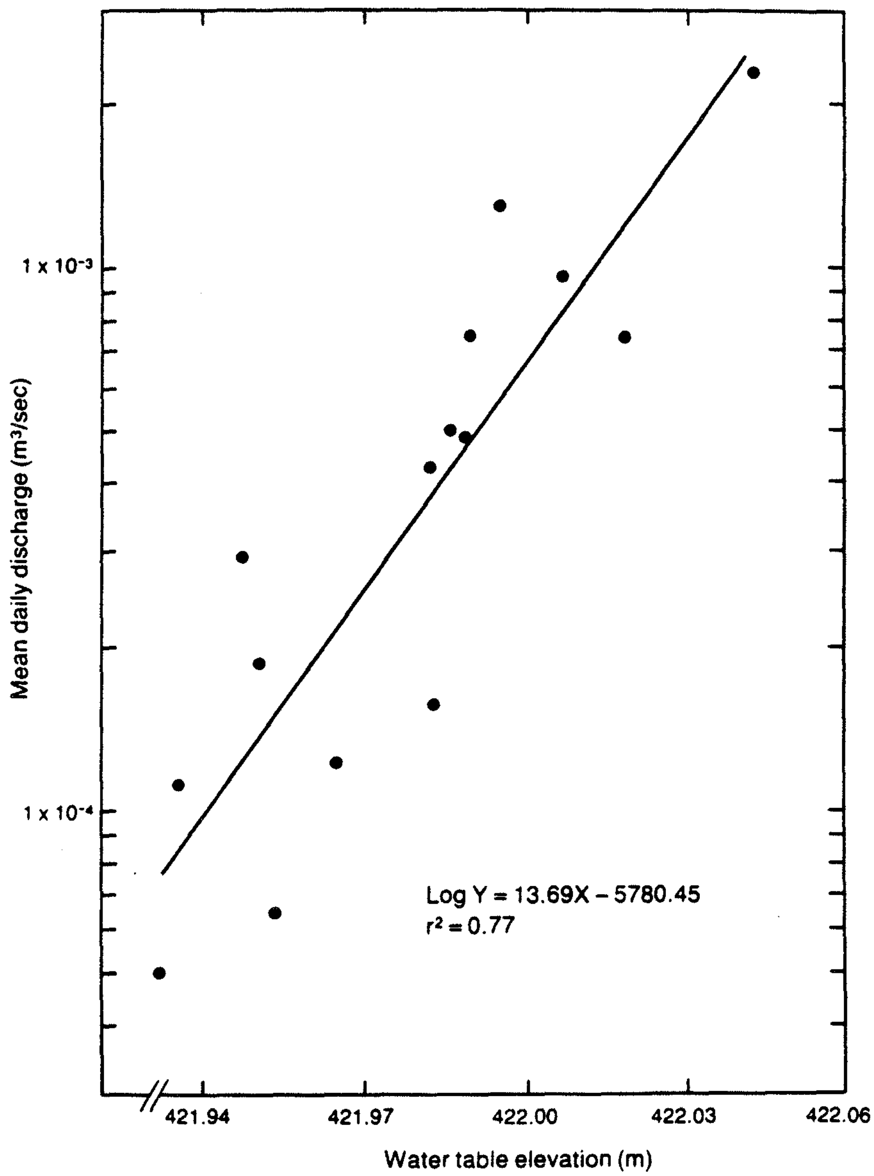

Figure 5.9. Relationship between mean daily discharge and water table elevation in a bog in Minnesota. (Adapted from Bay 1970).

seasonal distribution of streamflow from raised bogs and from groundwater fens (Figure 5.10). Annual and seasonal runoff from raised bogs is not well regulated in continental climates where winters are characterized by snow accumulation and frozen surface water. Bay's (1968) studies showed that water tables reach their lowest levels in bogs in late winter just before spring breakup. During snowmelt in spring, water tables in both raised bogs and fens rise rapidly, commonly to the highest level of the year. At this time of year all peatlands behave similarly to lakes or water-filled reservoirs. Any additional input of water leaves as runoff; in fact as much as 70% of the annual runoff from bogs occurs at this time. In raised bogs, there is a general decline in the water table throughout summer because of:

1. Increased evapotranspiration during the summer growing season;
2. Decrease in storage as the water table subsequently falls within the active layer; and
3. Decrease in the hydraulic conductivity of the peat with depth in the acrotelm due to decomposition and humification.

The net result is that as the water table falls during a dry spell, not only is the depth of the effective aquifer increasingly diminished, but flow becomes confined to the least conductive part of the active layer. Consequently, stream discharge from the bog falls significantly or ceases entirely. Even though there may be periods of intense precipitation, (as much as 42% of the rain) by late summer discharge is significantly reduced (one-fourth of the mean annual yield) or ceases altogether. This is clearly illustrated in Figure 5.11 which shows that almost all rainfall during the summer growing season in northern Minnesota is returned back to the atmosphere by evapotranspiration. Thus, prolonged

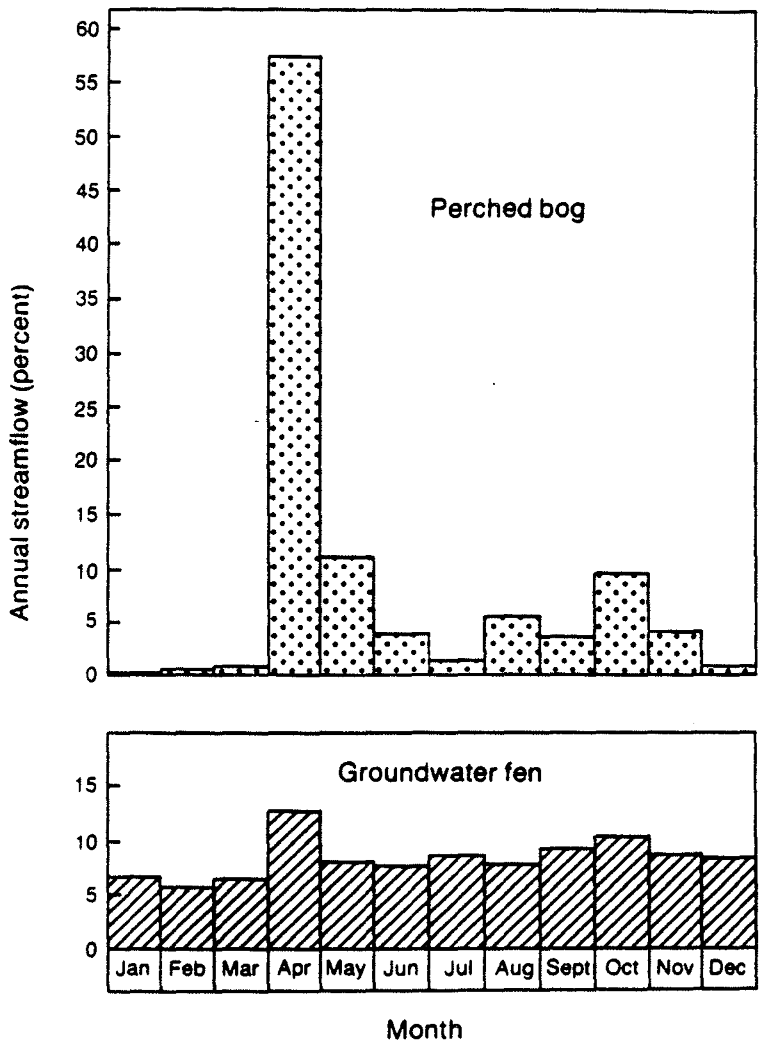

Figure 5.10. Monthly distribution of annual streamflow from a perched bog and a groundwater fen in Minnesota. (Adapted from Boelter and Verry 1977).

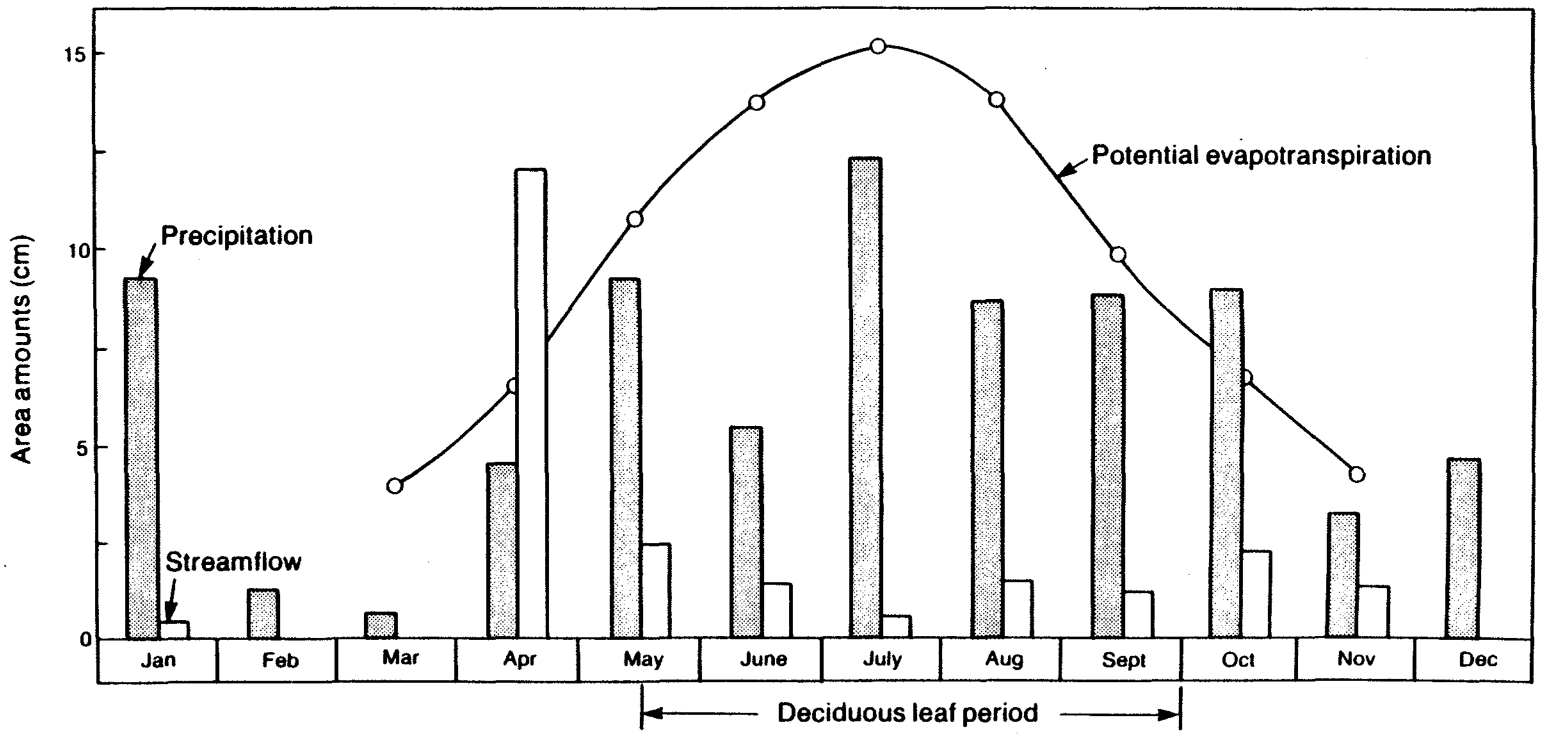

Figure 5.11. Monthly precipitation, streamflow and potential evapotranspiration for a perched bog watershed. (Adapted from Boelter and Verry 1977).

recession curves are not characteristic of summer hydrographs from raised bogs. Water yield from the peat itself appears to be minimal. Most yield results from runoff during wet periods, while the minimal storage sustains flow during dry periods. Water tables are generally stable in fall, depending on the summer moisture conditions. A wet summer results in early fall runoff, whereas following a dry summer, heavy fall precipitation recharges the depleted moisture in the peat and runoff would be negligible.

5.8.5 Discharge from Minerotrophic Fens

Although there are few studies of hydrologic balances in fens, Bay (1968) suggested that larger and more complex peatlands with variable vegetation and peat types, such as might be found in fen peatlands, may influence water yield differently from small bogs. However, the general relationships between storage, water table height, and discharge should apply. A comparison of runoff and water table characteristics of raised bog and fen peatland watersheds in northern Minnesota showed some similar trends in underground basin storage (Bay 1968). In years of normal fluctuations in the deep groundwater table, water tables in both types of peatlands reacted in a similar manner (Figure 5.12, A). The summer water levels in both peatlands responded quickly and similarly to rainfall recharge and drying periods, but the over-winter recession for the fen was less because of the recharge from the regional groundwater system. In years of high regional groundwater tables when recharge was exceptional, the water table heights within the bogs and fens showed considerable differences. The water table in the fen was sustained at a high, smooth, and steady level because of the increased groundwater discharge. In the bog the water table did not benefit from groundwater input and thus showed its normal decline through the dry summer (Figure 5.12, B).

Because of recharge from telluric sources, discharge rates from fens are characteristically less variable than those from bogs. Boelter and Verry (1977) found that stream discharge from fens is

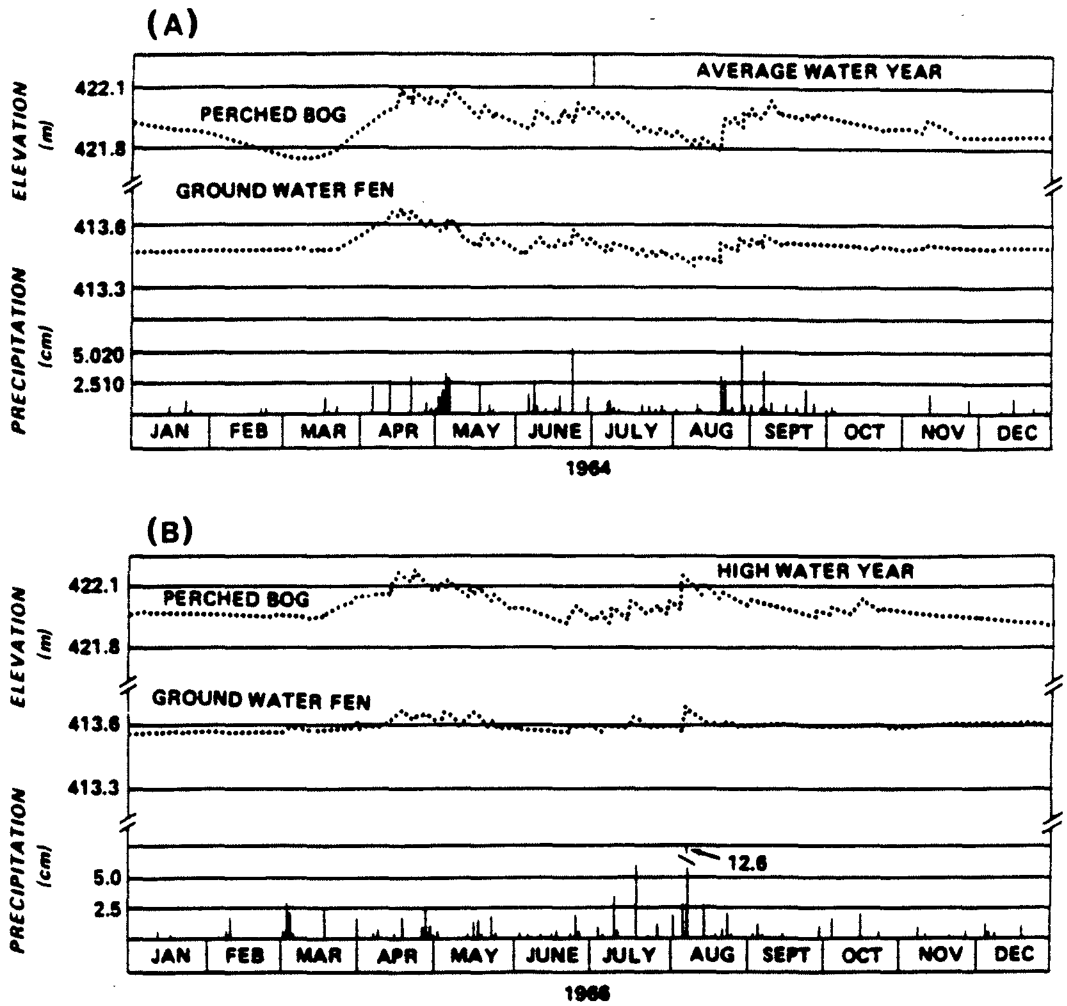

Figure 5.12. Water table levels in a perched bog and groundwater fen in (A) 1964 and (B) 1966. (Source: Bay 1970).

nearly constant about 70% of the time during the year (Figure 5.13). During the growing season, discharge rates from fens were somewhat lower than rates for surrounding mineral-soil areas because of the higher evapotranspiration rates in fens (Bay 1968). Conversely, in the dormant season discharge from the fen can be greater than from similar sized mineral-soil catchments, because peatlands in general occupy topographic lows and are thus subject to surface and groundwater contributions. Therefore, unlike bogs which do not contribute to baseflow or low streamflow during a dry summer, fens do contribute to baseflow but only because they typically function as areas of regional groundwater discharge.

5.8.6 Discharge Resulting from Storm Rainfall

Stream runoff from raised bogs will generally respond rapidly to storm rainfall, provided the water table is high and runoff has not ceased entirely. The concentration time during which stream discharge from the peatland builds up to a peak is typically very short, likely because small peatland areas are commonly located near the headwaters where mean travel time is short (Ingram 1983). There is a close dependence of the height attained by the discharge peak and the moisture conditions beforehand. A high antecedent water table allows less room for temporary storage of excess water in the peat profile. Thus, as illustrated in Figure 4.12, at a high water table level, where hydraulic conductivities are also greatest, discharge rates following storm rainfall are significantly higher than if the water table was initially low.

There is typically a lag in the time between the maximum rainfall event and the peak of the runoff discharge. This lag will increase or decrease depending on:

1. The height of the original water-table in the bog (a high water table results in decreased lag time);
2. The hydraulic conductivity of the peat at that water table depth; and
3. The slope of the system (Dooge 1975).

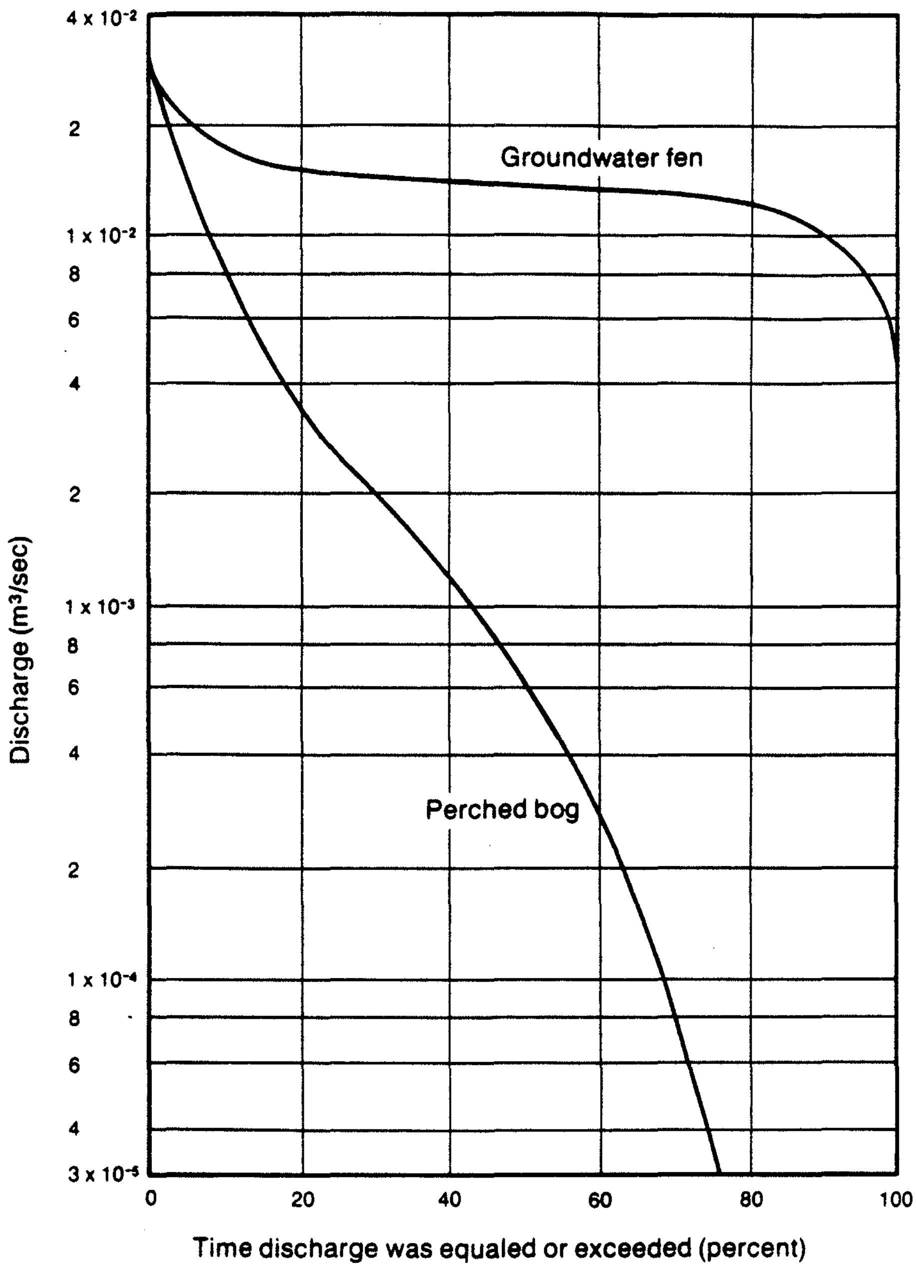

Figure 5.13. Streamflow duration curves for a groundwater fen and a perched bog watershed in Minnesota. Both watersheds are about 53 ha in size. (Adapted from Boelter and Verry 1977).

Bay (1969) showed, however, that even with a high water table, the recession curves on the hydrographs are drawn out and storm peaks are relatively low (Figures 5.7). This indicates that even at high water table levels, bogs still exert some regulatory effect by delaying short-term or storm runoff. This is attributed more to the regulating effects of the nearly-level bog topography than to the properties of the peat material (Bay 1969; Boelter and Verry 1977).

5.9 HYDROLOGIC FUNCTIONS OF PEATLANDS

5.9.1 Areas of Recharge

Considerable controversy centres on the role of peatlands as areas of recharge. Keough and Pippen (1983) demonstrated that in Michigan, USA, perched bog waters can leak out of the bog basins and mix with the local groundwaters in the surrounding drift. Verry and Boelter (1978) indicated that recharge may occur if the peat is thin and overlies a permeable sandy substrate. Thick peatlands however, do not easily permit downward percolation of water because permeability decreases significantly with depth. They concluded that recharge is likely to be greater in mineral-soil catchments than within peatlands. Novitzki (1978) found that in Wisconsin, basins with wetlands have less base flow in fall than do basins without wetlands. Hence, wetlands appear to contribute less to recharge. Conversely, Carter et al. (1978) concluded that the majority of studies indicate wetlands function as areas of discharge.

5.9.2 Erosion Control

Erosion potential is affected by two physical processes during the development of peatlands. On one hand, peat reduces erosion by intercepting rainfall, consuming water through evapotranspiration, dissipating the energy of active water, and stabilizing soil with plant roots. Native plants bind the soil above and below the water surface, they reduce stream energy by friction, and induce sedimentation by slowing stream currents (Carter et al. 1978; Ivanov 1981). On the other hand, the formation of peat clogs the pores of permeable mineral-soil substrata with organic colloidal material. This reduces vertical seepage and recharge, and enhances the horizontal flow of water, which leads to greater runoff and erosion of surface materials (Ivanov 1981).

In comparison with saturated mineral-soils, peat is generally highly resistant to erosion. As a result, streams that leave a peatland can experience rapid downcutting of the exposed mineral soil, sufficiently to produce a sharp drop in the profile of the stream bed. Small waterfalls, 0.5 to 2.0 m in height, have been recorded at the peat - mineral-soil boundary, and have been observed to migrate upstream as the headwall in the peat bank collapses.

Studies of sediment yield in drainage basins of north central Wisconsin (Novitzki 1978) suggest that peatland development is generally effective in reducing stream erosion. The results of the study (Figure 5.14) show that sediment yield was reduced by as much as 90% in basins containing 40% wetlands (lakes and peatlands), compared to basins with no wetlands.

5.9.3 Flood Storage and Storm Flow Modification

Storm flows or floods result from annual snowmelt or periods of heavy precipitation. The size of the flood depends on such factors as: basin size, antecedent moisture conditions, water content of the snow pack, rate and duration of snowmelt or rainfall, infiltration rate of the substrate, and the percentage of the basin occupied by wetlands (Boelter and Verry 1977; Verry and Boelter 1978; Novitzki 1978, cited by Carter et al. 1978). In general, lakes and wetlands in a drainage basin act to attenuate flood peaks and storm flows by temporarily storing water.

Novitzki (1978, data from Conger 1971) showed that in Wisconsin, peak flood flows may be as much as 80% lower in drainage basins that contain about 40% wetlands, compared to basins with no wetlands. Based on an analysis of a theoretical reduction of

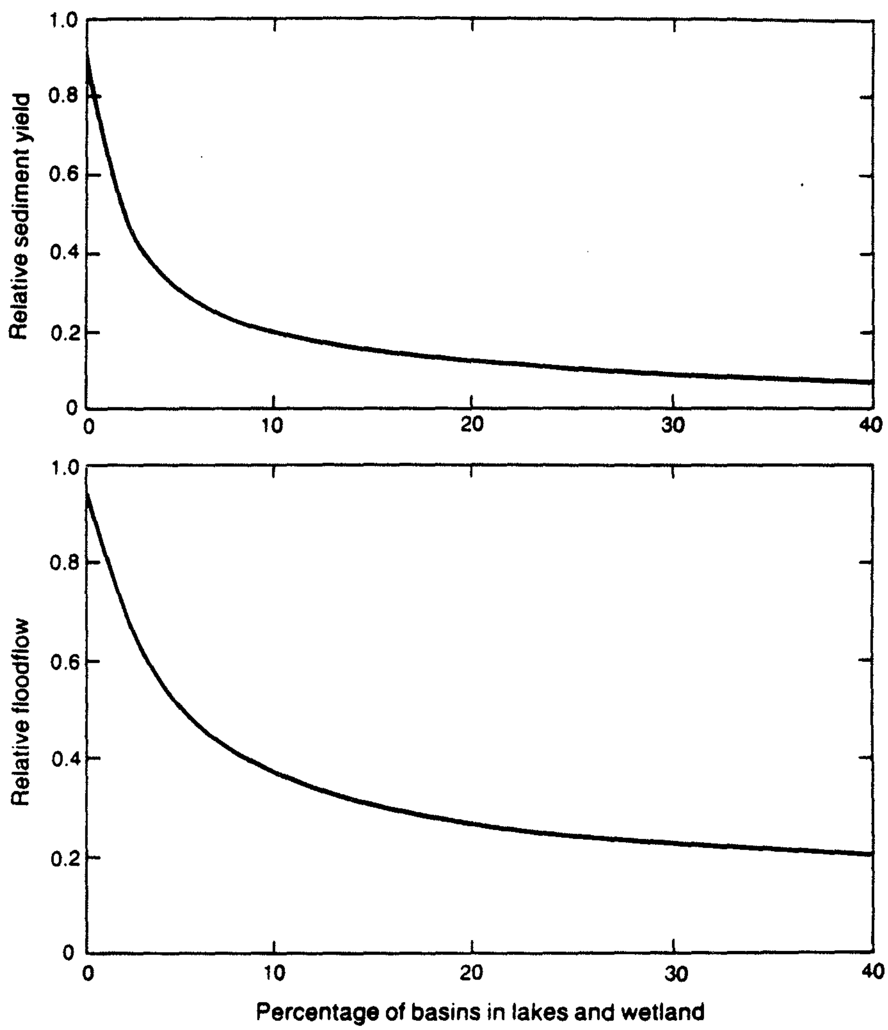

Figure 5.14. Relative sediment yield (top) and relative flood flow (bottom) in basins with different percentages of lake and wetland area. (Adapted from Novitzki 1978).

wetlands and impact on flood stages, Novitzki determined that 50% of the flood peak is reduced by the first 5% of lakes and wetlands in a basin (Figure 5.14). He concluded that losses of wetlands from basins with few initial wetlands have a greater impact on flood stages than similar losses from basins with numerous wetlands. This aspect of peatlands in attenuating flood stages is attributed mainly to the flat topography and the retarding effect that vegetation and the surface horizon of peat have on temporarily detaining peatland runoff (Bay 1968, 1969; Verry and Boelter 1975). It must be kept in mind, however, that Novitzki's conclusions are based on observations of a system of wetlands and lakes and that peatlands are not treated separately within that system.

Peatlands that are completely enclosed in a basin and which have no surface outflow, retain almost all snowmelt and precipitation and have minimal impact on flood peaks or streamflow in the basin.

5.10 EFFECT OF PEATLAND DRAINAGE AND DEVELOPMENT ON HYDROLOGY AND WATER QUALITY

5.10.1 Peat Permeability

Drainage destroys surface plant life and lowers the water table, which increases the zone of aeration and enables oxygen to penetrate the deeper horizons (Boelter and Verry 1977; Ivanov 1981). This causes greater oxidation and decomposition which leads to increased settlement and compaction, and decreased permeability. As well, macropores in the upper part of the peat are transformed into micropores following drainage, leading to increased consolidation (Figure 5.15).

Two opposing processes which affect permeability can occur following drainage (Ivanov 1981). In one case, as the water table is lowered, the pressure on the solid matrix is increased because of the increased weight of material and capillary moisture lying above the water table. The material becomes more compact, effective porosity is reduced and hydraulic conductivity is diminished. On the other hand,

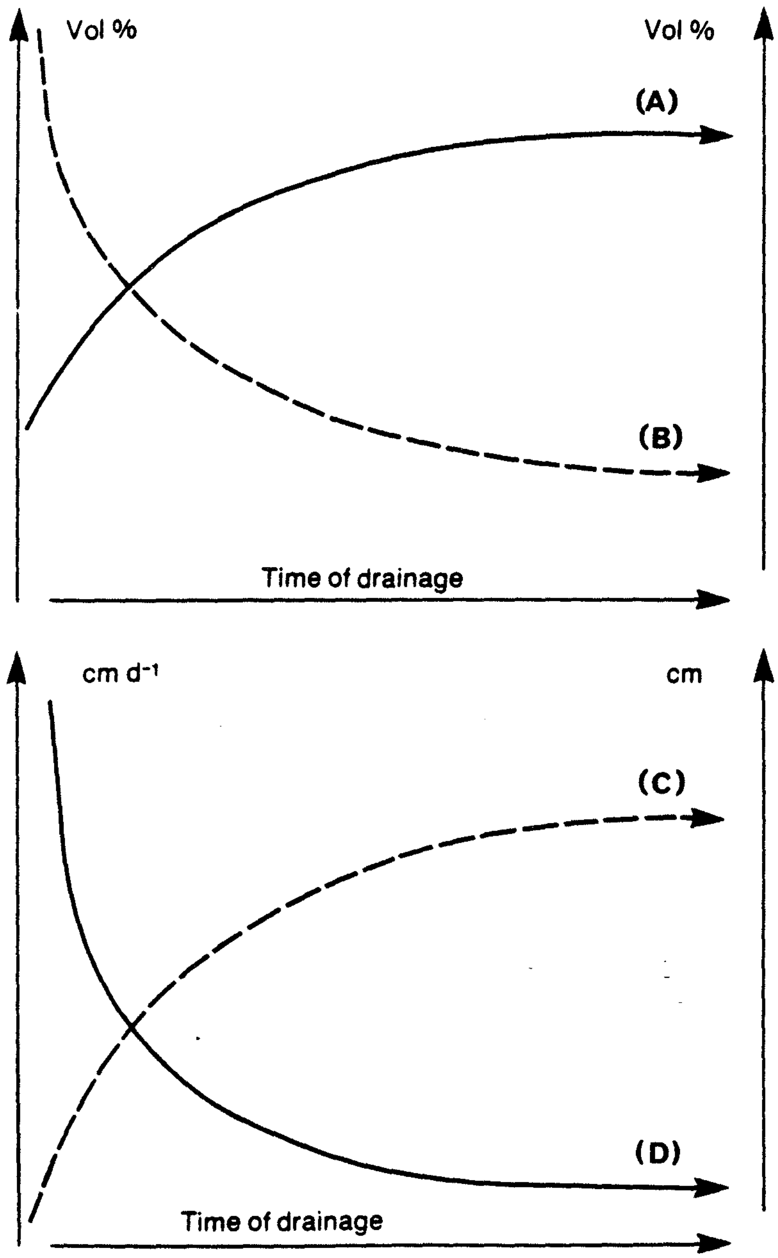

Figure 5.15. Influence of peat drainage on soil physical properties: (A) bulk density, (B) subsidence, (C) drainage pore space, and (D) permeability. (Adapted from Eggelsmann 1975).

dehydration of the peat increases above the new water table and evaporation acts to reduce the size of the particles. This increases the porosity of the material which leads to an increase in hydraulic conductivity. Ivanov suggested that the two processes compensate each other and that there is a small net change in active porosity due to drainage. However, he cited work of others that indicated the porosity values may decrease 1.2 to 100 times that of values in undrained peat. Citing work of Lundin (1964) and Maslov (1970), Ivanov (1981) showed that following drainage of peatlands for agricultural purposes, the hydraulic conductivity decreased because of compaction, reaching a minimum after about 10 years. Following this, physico-chemical changes act to increase hydraulic conductivity. Lundin (1964) quoted values of reduction 10 to 70 times for fen drainage, and 100 times for raised bogs. Maslov (1970, cited in Ivanov 1981) quoted significantly lower values of 1.2 to 5 times reduction for fen drainage. This indicated to Ivanov (1981) that the results of studies on the impact of drainage on hydraulic conductivity are far from conclusive.

5.10.2 Evapotranspiration and Stream Discharge

Verry and Boelter (1978) suggested that annual streamflow, as well as minimum flow, is likely to increase following drainage because evapotranspiration is reduced as the water table falls below the capillary zone of the plant roots. However, there is disagreement about the effects that a planted crop would have on evapotranspiration. Bulavko (1971, cited in Goode et al. 1977), noted that evaporation may be reduced by as much as 50% following drainage, and that even if crops are planted, evaporation is still 10 to 15% less than original values. This appears to contradict work done in Russia by Romanov (1961, cited in Dooge 1975) who suggested that drainage and cultivation will increase annual evaporation from bogs but will likely not affect the rates for fens.

In reclaimed bogs cultivated as grasslands in Atlantic climates, the effect of replacing mosses with more vigorous plants is to increase evaporation in summer but decrease it in fall (Figure 5.16). However, Baden and Eggelsmann (1970) concluded that the total yearly evaporation rate from cultivated bogs is essentially the same, or slightly less than that for undrained, moss-covered bogs. Evaporation in fens appears to be redistributed so that it is less in spring and greater in summer. This results in greater runoff in spring and presumably less in summer. It must be remembered, however, that the study site was located in an Atlantic climate where winter temperatures are milder and where precipitation in the form of rainfall is able to support vegetative growth in undeveloped bogs well into the late fall months.

Harvesting a forest-covered peatland may or may not raise the water table, depending on whether it is a bog or a fen (Boelter and Verry 1977). In a fen, the effect of harvesting a forest canopy, which retards direct precipitation falling on the peat surface, will be relatively minor because the contribution of groundwater to the fen exceeds that of precipitation. In raised bogs, the frequency of precipitation will decide the impact that harvesting has on the bog water table. Prolonged dry periods in harvested bogs result in lowered water tables due to effects of surface-wind evaporation and increased transpiration from the growth of deeper-rooted sedges.

Goode et al. (1977) stated that planting a forest in a reclaimed peatland may have the effect of increasing runoff. A developing forest will produce a canopy which reduces radiation on the ground surface, resulting in reduced evapotranspiration and increased annual runoff. Heikurainen (1975) agreed that a developing canopy will decrease evaporation but indicated that transpiration increases, rather than decreases, as the forest stand develops. He concluded that although evapotranspiration decreases sharply following drainage, with time evapotranspiration rates may exceed predrainage values, and runoff will be reduced.

5.10.3 Storage Capacity

Drainage and reclamation activities increase the storage

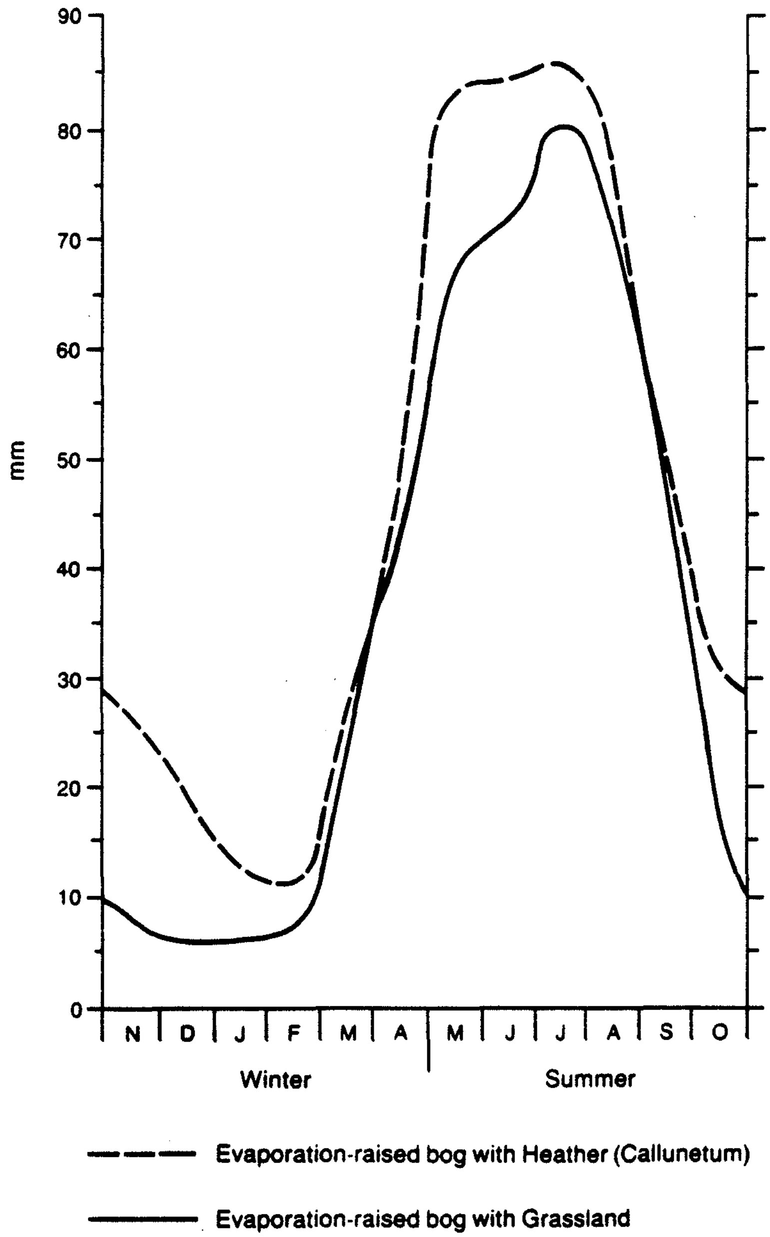

Figure 5.16. Yearly trend of evaporation in a heather-covered raised bog and a raised bog grassland in the Federal Republic of Germany. (Adapted from Baden and Eggelsmann 1970).

capacity of a peatland and thus regulate annual flow and increase annual runoff (Baden and Eggelsmann 1970; Mustonen and Seuna 1971, cited in Heikurainen 1975; Lundin 1975; Goode et al. 1977). In particular, minimum flow is increased during the dry summer months and in winter when river levels are low. Peak storm runoff is both reduced and delayed in drained peatlands because of increased storage capacity (Baden and Eggelsmann 1964, cited in Dooge 1975; Baden and Eggelsmann 1970; Burke 1968; Heikurainen 1975). In some cases peak discharge may increase if drainage ditches are narrowly spaced and rainfall or snowmelt becomes quickly channeled (Heikurainen 1975).

Lundin (1975) showed that in peatland drained for agricultural purposes, water levels fall for about two years but stabilize by about the third year. Following stabilization, the properties of the peat do not vary much with changing groundwater levels. In general, Lundin found that moisture content of a drained peatland is generally below field capacity. This results in a free storage volume in the peat which is capable of storing water as suspended moisture, without any rise in the water table. Drained peat thus behaves more as a reservoir that regulates soil moisture and stabilizes groundwater levels, thereby reducing seasonal peaks.

5.11 AREAS OF CONCERN IN FUTURE HYDROLOGIC STUDIES OF PEATLANDS

5.11.1 Problems in Monitoring Water Budget Components

A number of recent review papers that discuss components of the water balance equations of peatlands focus on the sources of errors associated with estimating values of the components (Carter et al. 1978; Winter 1981). Winter (1981) evaluated methods used in calculating components of the water budget and concluded that analysis of errors have not been performed for most water balance studies presented in the literature. A brief discussion of Winter's observations on assumptions and sources of errors in water balance studies is presented here.

5.11.2 Precipitation

Instrumentation errors in monitoring precipitation vary between 1 and 5%. Errors caused by gauge placement and gauge density range between 5 and 15% for long-term records, and as high as 75% for individual storm events. It is often necessary to interpolate between several gauges, extrapolate from single stations, or use weighting techniques to provide estimates over an entire peatland.

5.11.3 Evaporation

The energy budget method, which requires measurement of all forms of energy entering and leaving a water body as well as change in heat stored in the body, is considered to be the most accurate method for calculating evaporation. Estimated annual error is less than 10% and estimated seasonal error is less than 13%. The method is expensive, requiring extensive instrumentation and frequent monitoring.

An alternative method, the evaporatiion pan, is more commonly used because it is the least expensive, even though it requires daily servicing. However, the method has errors as large as 30%. Annual evaporation from standing water bodies is calculated to be about 0.7 times that from pans (lake-to-pan coefficient of 0.7) because wind and thermal regimes of pans and lakes are different. Winter (1981) reported that this coefficient ranges from about 0.5 to 0.9 annually, and that seasonal variations are even greater. Estimates of evapotranspiration from different peatland vegetation types range from 0.54 to 5.3 times that of pan evaporation (Carter 1986).

The mass transfer method, in which the evaporation rate is proportional to the water-vapour pressure gradient between the evaporating surface and air above the surface, is considered by Winter (1981) to be the best alternative. Reliable instruments are available, and require only weekly servicing. The drawback to this method is that a mass transfer coefficient must be calculated to account for wind profile, size of the water body, roughness of water surface, atmospheric stability, barometric pressure, and density and viscosity of air.

Most studies reviewed by Winter (1981) focussed on the calculation of evaporation from lake bodies, and do not include the effects of transpiration. Studies of vegetated water bodies show evaporation rates as much as three times that from an open water surface (Bulavko 1971, cited in Goode et al. 1977; Benton et al. 1978, cited in Winter 1981). The results suggest that methods of calculating evapotranspiration from free-water bodies may not be applicable to peatland hydrologic studies.

Carter (1986) concluded that most peatland studies estimate evapotranspiration from either the residual term of the water balance equation, from groundwater fluctuations, or by empirical formulae which relate actual evapotranspiration to potential evapotranspiration. Methods are seldom applied that measure evaporation directly. Even when analytical methods are applied to monitor evaporation, estimated errors in evaporation are commonly evaluated with respect to other methods of evaporation which also contain errors (Winter 1981).

5.11.4 Surface Water

Techniques for measuring surface water input and output of peatlands are probably accurate within 5% (Winter 1981). Flumes or weirs are the best methods for measuring discharge into and out of peatland. Nonchannelized inflow (overland flow) can be an important contributor to the water budget of a peatland but monitoring overland sheet flow is difficult because flow paths are affected by microtopography and vegetation.

5.11.5 Groundwater

Groundwater is one of the most difficult components of the peatland water budget to monitor, primarily because the installation of water table and piezometer wells in peat terrain is difficult and very expensive. The groundwater component of the water balance equation is typically considered to be negligible and is either ignored, or estimated from the residual term in the water balance equation. However, Verry and Timmons (1982) calculated that groundwater seepage accounts for about one-third of the liquid water, or 10% of the total water leaving a bog and upland catchment area. Siegel (1983) estimated that groundwater discharge and runoff accounts for at least 6% of annual runoff and is proportionally higher during dry periods. Sklash et al. (1976) showed that within non-peatland drainage basins, groundwater discharge is by far the largest component of streamflow following a storm. They suggested that only small increases in hydraulic head are needed near the stream to cause a large groundwater discharge component in storm runoff. It is conceivable that their conclusions can be also applied to drainage basins containing peatlands.

Carter (1986) commented that the behaviour of fluids moving through organic soils is not as well understood as that within mineral soils, described by Darcy's Law. Movement of water in the acrotelm is fairly well documented in the studies, primarily because most water transfer occurs in this part of the peat column. Water movement through the catotelm is considered to be significantly less but there have been a few studies which document water movement in this layer (Ivanov 1981; Siegel 1983; Chason and Siegel 1986). Chason and Siegel (1986) presented evidence that water movement through decomposed peat in the catotelm may be greater than previously thought, and that interaction between groundwater and surface water in peatland can be significant. Siegel (1983) commented that, in particular, future study of the movement of water along the peat - mineral-soil interface is required.

5.12 WETLAND HYDROLOGY MODELS

In their literature review, Carter et al. (1978) found that there have been few efforts to model wetlands in North America. In this regard, three approaches have been taken. The first considers wetlands as part of a large system, such as a drainage basin. Carter et al. cited studies by Conger (1971), Hindall (1975), and Anderson-Nichols (1971). The second considers modelling an individual wetland, such as Sander's (1976, cited in Carter et al. 1978) model for a small peat bog and its watershed. Sander's model shows that peatlands do not stabilize their hydrology but rather, amplify seasonal variations through rapid removal of water during wet periods, and consumption of water by evapotranspiration during dry periods. Dooge and Keane (1975) developed two conceptual models for peatlands which predict runoff from drained peatlands, following precipitation events. The third approach to modelling wetlands involves models of individual components or processes, or several processes within the wetland. The attempt by Romanov et al. (1975) to calculate evaporation, runoff, and water yield from climatological data, in order to predict water levels and evaporation rates in bogs is cited as an example of this approach to modelling.

Recently, Siegel (1983) applied computer simulations to predict groundwater flow paths in complex peatlands of northern Minnesota. The model shows that groundwater movements can be caused by water table mounds beneath raised bogs, and that these mounds control the distribution of large fens and bogs in an expansive peatland. In Siegel's model, groundwater initially circulates from upland recharge areas through flow paths several kilometres wide that pass through the peat column and into the mineral soil. Eventually a modern groundwater flow path is established and maintained by raised water table levels beneath raised bogs. Siegel's model predicts that beneath large raised bogs, a water table mound of only 0.3 m above the regional water table creates downward flow systems that can extend as much as 15 m into the underlying mineral soil. In Siegel's study area this water passed through glacial lacustrine sediments and dissolved carbonate materials. Artesian upwelling of this mineral-rich water supplied the nutrient requirements for the development of neighboring fens, which function as groundwater discharge sites. By this process, the model demonstrates that local groundwater flow systems can become established beneath raised bogs and that these can replace regional flow systems from nearby uplands, as areas of recharge for surrounding fens.

In Canada, a conceptual model that addresses both mass balance and water balance was designed by Clarke-Whistler et al. (1984) to simulate conditions in an intact peatland, a peatland with vegetation removed, a peatland subjected to dry mining, and a peatland subjected to wet mining. Simulations were made for a water balance, fluctuating water table levels and change in storage, and a number of chemical parameters. The authors consider the following to be the 'master' variables for the model:

1. Flow paths and rate of flow - controlled primarily by structure of peat and hydraulic conductivity;
2. pH and reduction-oxidation (redox) potential (Eh or p*e*) - controlled by the buffering system and biochemical reactions;
3. Bulk density, decay rate, adsorption properties and soluble organics of peat - function of peat characteristics and biochemical reactions; and
4. Chemical precipitates - mass of metal in soluble form, and function of pH and Eh.

This model was tested with hydrologic and chemical data from Hemond's (1980) study of Thoreau's Bog in Massachusetts, USA.

A review of models that define the impacts of land drainage has been prepared by Andres et al. (1984). This review was part of a larger project having the objective of developing a set of guidelines to assess the physical and hydrologic constraints to agricultural land drainage in Alberta. A conceptual model of basin response to single storm events was also developed as part of this project. Non-peaty types of wetlands in agricultural areas were mainly addressed by Andres et al. (1984); however, the models presented should be applicable to determining hydrologic impacts of peatland drainage as well.

Schwartz and Milne-Home (1982a, 1982b) designed a model that characterizes the chemical properties of waters from various components of the hydrologic cycle as a means to interpret the various surface and subsurface water processes in a peatland watershed in northern Alberta. This is discussed in more detail in the following section.

5.13 HYDROLOGIC STUDIES OF PEATLAND CATCHMENTS IN ALBERTA

There are few studies in Alberta which document the hydrologic role of peatlands. Schwartz and Milne-Home (1982a,b) evaluated the role that shallow groundwater and peatland systems play in controlling the quantity and chemical quality of stream water north of Fort McMurray in northeast Alberta. A chemically based method that applies mass balance techniques was used to avoid the expense of more conventional groundwater investigation methods which rely on observations from numerous test wells. The study attempted to establish the relative amounts of direct precipitation, surficial drift groundwater, and standing muskeg water input to stream discharge from the Muskeg River catchment. An assumption that waters from each of these sources are chemically unique was tested using cluster analysis. The results show that although the separation between muskeg water and direct precipitation was weak, groundwater and standing muskeg water form distinct clusters. A mixing model was developed to separate the contributions to streamflow from the various sources based on their major ion-concentration levels. The model shows that the major source of water for streamflow shifts on a seasonal basis. In the winter months when discharge is low, the water reaches its maximum concentration with respect to most ion species. Streamflow is at baseflow conditions at this time, and is maintained primarily by groundwater discharge. Following spring melt and during the remainder of the year, streamflow is maintained dominantly by drainage from the muskeg into the surface water system, with some groundwater discharge. Contributions are also made from direct precipitation and surface runoff following storm events.

A comparison of the findings from the Muskeg River basin with results from other basins showed significant differences in the relative contributions from input sources to streamflow in the respective basins (Schwartz and Milne-Home 1982b). In some basins the groundwater contribution during summer was as high as 50%, compared to less than 20% in other basins. In another basin, drainage from muskeg contributed as much as 50% of winter streamflow. The report concluded that watersheds behave in different ways and that streamflow in winter can be maintained dominantly by groundwater or by drainage from muskeg. A general observation, however, was that in none of the drainage basins studied did streamflow return to baseflow conditions during spring or summer.

In assessing the potential impact that a major land disturbance, such as strip mining of oil sands, may have on the quality and quantity of streamflow from peatland catchments, Schwartz and Milne-Home (1982a, 1982b) concluded that the magnitude will depend on (1) the hydrologic setting, (2) the proportion of the muskeg that has been altered, and (3) the manner in which the muskeg has been altered.

Muskeg may not be uniformly distributed within a catchment, and storage in ponds or lakes may be a more significant contributor to streamflow than muskeg. Even if a particular area of muskeg were removed, there would likely be little impact on the streamflow characteristics if its proportion to the remainder of the catchment were minor. Schwartz and Milne-Home believe that in the extreme case where muskeg is removed entirely and replaced by vegetative growth on mineral soil, the catchment will behave similarly to watersheds without muskeg. They predicted that because the storage capacity of the peat has been eliminated, stream discharge will be greater during spring runoff and less during summer. This conclusion is in contrast to the previous conclusions that indicate:

1. Mineral-soil catchments have a greater capacity to store excess spring meltwater because they are less saturated than muskeg; and

2. Mineral-soil catchments are better able to sustain discharge in summer because evapotranspiration rates are less than in muskeg (Bay 1968; Verry and Boelter 1978; Novitzki 1978).

5.14 SUMMARY

5.14.1 Hydrogeologic Setting of Peatlands

Peatlands develop where excess moisture conditions exist. This can occur on flat or even convex surfaces such as mountain slopes. Excess moisture leads to deficiency of oxygen in the soil, which prevents oxidation and decomposition, and enables plant remains to accumulate.

Bogs receive only nutrient-poor meteoric water (precipitation) and are able to sustain only a few plant species, mainly mosses. Fens receive both meteoric and nutrient-rich telluric waters (groundwater), which support more diverse plant species. Most hydrologic studies have focused on bogs, mainly because of the difficulty in monitoring groundwater contributions to fens.

5.14.2 Water Balance Equation

In the ideal water balance study every component is measured independently; this is rarely achieved because of the difficulty in acquiring data in peatland environments. Most studies estimate the value of some component from a residual term determined by the difference between measured input and measured output.

Errors of measurement in monitoring can account entirely for the value of the residual term.

Nutrient mass balance studies that are based on the water balance equation are subject to the same degree of error resulting from the estimation of components in the water balance equation.

5.14.3 Hydrologic Components of Water Balance Equation

Evapotranspiration rates in peatland basins are greater than for mineral-soil basins. Evapotranspiration rates in fens are also greater than in bogs.

Evapotranspiration is greatest when the peat water table is high and decreases rapidly as water table falls to a depth of about 30 cm from the surface.

Evapotranspiration can significantly reduce stream discharge during intense growing months, even if precipitation is great. Diurnal effects on stream discharge can be observed as plant transpiration increases during midday.

Potential evapotranspiration exceeds precipitation over most of Alberta, with the exception of the major uplands in the north half of the province.

Most of the water and heat exchange occurs in the more permeable, upper 30 to 70 cm of the peatland, referred to as the acrotelm. Hydraulic conductivity values in the acrotelm are three to five times greater than in the denser, saturated lower layer, referred to as the catotelm.

As much as 80% of the water residing in the acrotelm can be released to drainage. Only 10 to 15% of the water in the catotelm can be released to drainage because of the low hydraulic conductivity, even though most of the water stored in a peatland resides in this layer.

Differences in hydraulic conductivity of peat with depth ensure that water is drained by horizontal flow through the acrotelm in periods of excess moisture but is drained by slow vertical seepage through the catotelm when precipitation ceases. This process maintains generally high and stable water tables in peatlands.

Water tables in fens are sustained by groundwater input, show less fluctuation, and are higher than water tables in bogs, which are subject to prevailing climatic conditions.

The effects of precipitation on the water table depend on the antecedent moisture conditions of the peat. When water table levels are initially low, precipitation causes a rapid rise in the water table because of the low effective porosity of the peat at depth. At high water tables, an equivalent amount of precipitation causes a slow rise in the water table because effective porosity and storage capacity are greater in the upper part of the peat.

During wet conditions water storage is generally high, varying within narrow limits. Storage capacity is very low and input of storm water is quickly followed by a peak in drainage.

Peatlands temporarily reduce peak rates of discharge following storm rainfall or snowmelt but this is due in part to the flat surfaces and retarding effects of vegetation in peatlands, rather than the properties of the peat itself.

Water is released from a peatland slowly during dry periods and is not a major contributor to low streamflow. A significant amount of water is discharged to the atmosphere by evapotranspiration at the expense of streamflow, and discharge from bogs may cease altogether in summer. Fens are more efficient at regulating flow, but compared to mineral soil, they release excess water more quickly during wet periods and lose more water by evapotranspiration during dry periods.

5.14.4 Hydrologic Functions of a Peatland

Most peatlands are found in areas of discharge, although perched bogs may act as recharge sites in complex peatlands.

Peatlands can reduce erosion and sediment transport by intercepting rainfall, consuming water through evapotranspiration, dissipating energy of flowing water, and stabilizing soil with plant roots.

Peatlands can affect water quality by trapping incoming sediment and retaining dissolved materials. Peatlands can retain as much as 55% of annual input of dissolved materials and do not become easily saturated.

The flat topography and retarding effects of vegetation in a peatland significantly reduce peak flood flows in a drainage basin. Fifty percent of the flood peak can be reduced by the addition of five percent of peatlands to a basin.

5.14.5 Effects of Drainage on Hydrologic Components

The hydrologic balance and mechanisms of water movement of undisturbed peatlands are relatively stable whereas the balance in reclaimed peatlands is much less stable.

Methods used to calculate the hydrologic balance of undrained peatlands may not apply to drained peatlands where the physical properties of reclaimed peat are not stable but change with time. The acrotelm may disappear under agricultural use or peat production, and accelerated decomposition may alter the paths of water transmission. Drainage destroys surface plant life, lowers the water table, and increases the zone of aeration and oxidation. This causes greater decomposition, increased settlement and bulk density, and decreased permeability.

Lowered water tables following drainage result in increased storage capacity of the peatland. This regulates annual discharge and increases minimum flow. Peak storm runoff is both reduced and delayed because of increased storage capacity.

Annual streamflow from drained unforested peatlands is increased because losses to evapotranspiration are reduced as vegetation dies. This will be reversed if more vigorous vegetation is planted to replace mosses in peatlands.

The effects of reforestation in a drained peatland are uncertain. On one hand, a developing forest canopy will reduce radiation on the ground surface and thereby can reduce evaporation and increase runoff. On the other hand, transpiration increases as a forest stand develops, and with time, evapotranspiration will exceed pre-drainage values and reduce runoff.

The effects of drainage on the hydrologic components of a peatland are summarized in Table 5.4.

Table 5.4. Effects of drainage on the hydrology of a peatland.[a]

Component	Effect	
	Increase	Decrease
Evaporation		X
Water table level[b]		X
Storage capacity	X	
Interception[c]	X	
Peak runoff		X
Annual runoff	X	
Minimum runoff	X	
Subsidence	X	
Bulk density	X	
Porosity		X
Water content		X
Water yield coefficients		X

[a] Adapted from the original table in Clausen and Brooks (1980).

[b] Evapotranspiration may increase in forested bogs as tree growth becomes enhanced following drainage.

[c] Interception increases if plant growth is enhanced because of drainage; if clearing occurs following drainage then interception is decreased.

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6. WATER CHEMISTRY OF NATURAL PEATLANDS

L.W. Turchenek

6.1 INTRODUCTION

Characteristic chemical differences exist in the waters of different peatland types (as indicated in Chapter 3). Peat water chemistry is dependent on complex physio-chemical and biological interactions among peat, inflow waters, precipitation, and biota (Clarke-Whistler et al. 1984). Water quality studies of natural peatlands have been carried out mainly on water held in the peat matrix or in surface depressions within peatlands, and on waters draining from peatlands. Studies of peatland waters have generally been carried out to investigate the ecological nature of peatlands by relating water chemistry to vegetation or peatland type. Runoff waters have been examined mainly as part of investigations concerning impacts of peatland use on water quality. These studies are reviewed in Chapter 7. In this chapter, basic information about the inorganic and organic chemistry of peatland and runoff waters is presented.

6.2 INORGANIC CHEMISTRY OF PEATLAND WATERS

Most peatland ecology studies conducted to date in Alberta have included analyses of pH, electrical conductance, and the major cations. Data from some of these studies are presented in Table 6.1. Considerably more data have been gathered during recent research projects. Results of a Forestry Canada study are presented in Table 6.2 (Zoltai and Johnson 1987). Recently published papers and theses from the Department of Botany, University of Alberta, include Kubiw (1987), Nicholson (1987), Chee (1988), and Chee and Vitt (1989).

Data on some chemical parameters of pool waters were collected during the course of a peatland inventory project carried out in the Athabasca area, Alberta (Turchenek et al. 1984). The data, given in Table 6.3, were stratified according to pH increments and the strata were classified based on characteristics of peatland

Table 6.1. Summary of water chemistry in peatlands studied in various research projects in Alberta.

Reference	Location	Peatland Type	Sample Type	No. of Samples	pH	Specific Conductance $\mu S \cdot cm^{-1}$	Ca $mg \cdot L^{-1}$	Mg $mg \cdot L^{-1}$	K $mg \cdot L^{-1}$	Na $mg \cdot L^{-1}$
Vitt et al. 1975	Swan Hills	Poor Fen	Pool	48	5.2		2.3	0.4	0.6	3.0
Horton et al. 1979	Caribou Mountains	Bog	Pool	1-10	3.5-5.1	2.6-4.5	1.4-10.0	0.5-0.8	1.2-6.6	3.6-5.8
		Bog	Thaw Pocket	17	3.7	30	3.0	0.1	1.5	3.6
		Bog	Stream	6	4.7	39	6.4	0.3	3.4	4.4
Slack et al. 1980	Rocky Mountain House	Rich Fen	Pool	9	6.8-7.9	140-450	18-37	4-18	1.4-8.0	
Karlin and Bliss 1984	Heatherdown	Rich Fen	Pool	60	7.2-8.2		38-120	26-53		
	Wagner	Rich Fen	Pool	7	7.4-8.0		31-104	10-25		
	Vab	Transitional Fen	Pool	7	6.7-7.1		20-38	5-10		
	Alsike	Transitional Fen	Pool	4	5.0-6.5		4-51	2-11		
	Bug	Poor Fen	Pool	3	4.6-6.6		6-35	2-12		
	Nestow	Poor Fen	Pool	24	3.5-6.1		2-12	<1-3		
Makitalo 1985	Saulteaux River	Forested peatlands	Pool	12	6.3	97	18.8	4.5	0.8	5.0
			Pool	12	6.8	114	19.5	5.2	0.4	5.4
			Pool	38-42	6.4	111	20.0	5.2	0.7	5.5
			Pool	13	7.0	200	57.0	12.8	0.1	0.1
Schwartz and Milne-Home 1982 °	Muskeg River (Oil Sands)	Bog and Fen	Pool	16	-	137	17.0	4.9	0.6	4.1

° 16 sites were sampled in different peatlands every few weeks from spring to fall; other properties reported were Fe = 0.2 $mg \cdot L^{-1}$, HCO_3^- = 80.7 $mg \cdot L^{-1}$ SO_4^{2-} = 5.9 $mg \cdot L^{-1}$; Cl^- = 2.4 $mg \cdot L^{-1}$.

Table 6.2. Chemical composition of water table by nutrient classes.[a]

	Oligotrophic (Bog)	Oligotrophic (Fen)	Dystrophic (Fen)	Mesotrophic (Fen)	Mesotrophic (Conifer Swamp)	Macrotrophic
No. Sites	71	17	16	147	29	18
pH	4.5	4.7	4.9	5.8	5.8	6.5
SC[b] ($\mu S \cdot cm^{-1}$)	60	50	60	210	240	370
Elements ($mg \cdot L^{-1}$)						
Ca	2.04	2.24	3.61	24.98	30.80	53.60
Mg	0.87	0.72	1.68	10.16	11.78	14.20
Fe	0.90	0.91	1.68	0.60	0.53	0.41
S	1.21	1.11	0.79	1.68	3.38	1.73
P	0.17	0.19	0.08	0.13	0.20	0.12
Na	2.59	4.15	3.62	4.75	5.51	6.54
K	1.42	0.91	1.64	1.42	2.22	0.97
Al	0.46	0.74	0.19	0.30	0.52	0.97
Pb	0.01	0.04	0.02	0.02	0.02	0.06
Cu	0.02	0.04	0.01	0.02	0.03	0.05
Mn	0.03	0.04	0.07	0.10	0.28	0.01
Zn	0.02	0.02	0.02	0.02	0.01	0.01
Ni	0.05	0.03	0.28	0.09	0.01	0.06

[a] Adapted from the original table in Zoltai and Johnson (1987).
[b] SC = specific conductance.

Table 6.3. Some water quality parameters of peatlands stratified according to pH in the Athabasca region, Alberta.[a]

Peatland Type	No. of Samples	pH	SC[b] $\mu S \cdot cm^{-1}$	Ca	Mg	K	Na	S	P	CL	NO_3	HCO_3
				mg·L^{-1}								
Bog	≤20	<4.0	7-68[c]	1-13	0.4-2.6	0.1-1.2	0.1-1.6	0.1-1.9	0.1-0.3	1.8-7.6	0.1-1.4	0
			24	4	1.0	0.4	0.5	0.7	0.1	3.4	0.2	0
Poor Fen	≤9	4.0-4.4	25-50	3-9	1.0-2.4	0.1-1.4	0.1-0.7	0.4-1.4	0.1-0.3	2.0-4.8	0.1-0.2	0
			36	6	1.7	0.5	0.4	1.0	0.1	2.9	0.1	0
Poor Fen	≤2	4.5-4.9	24-45	3-7	1.0-2.0	0.1-2.0	0.1-0.4	0.4	0.1	2.7	0.1	0
			34	5	1.5	1.0	0.2	0.4	0.1	2.7	0.1	0
Poor Fen	≤10	5.0-5.4	38-72	5-11	1.5-3.7	0.1-5.1	0.1-2.1	0.1-1.4	0.1-0.2	0.9-5.3	0.1-0.3	0
			51	8	2.6	1.3	0.9	0.6	0.1	25	0.1	0
Transitional Rich Fen	≤25	5.5-5.9	13-139	6-17	1.7-7.5	0.1-4.9	0.1-3.4	0.1-6.9	0.1-0.4	0.9-5.0	0.1	6-74
			69	11	3.9	1.1	1.2	1.0	0.1	2.4	0.1	18
Transitional Rich Fen	≤62	6.0-6.4	31-300	4-47	0.9-13.0	0.1-6.0	0.2-13.0	0.1-20.0	0.1-0.5	0.3-4.8	0.1-3.2	10-114
			114	18	6.0	1.0	2.1	1.1	0.1	1.8	0.2	51
Transitional Rich Fen	≤62	6.5-6.9	89-506	6-70	2.5-20.0	0.1-8.6	0.3-23.0	0.1-27.0	0.1-0.8	0.3-10.0	0.1-1.7	18-279
			200	30	9.2	1.0	4.0	1.7	0.1	1.6	0.2	112
Transitional Rich Fen	≤12	7.0-7.4	86-486	12-69	4.5-21.0	0.1-2.8	1.9-41.0	0.1-7.6	0.1-0.2	0.4-12.4	0.1-1.0	44-280
			263	38	11.4	0.8	9.5	2.3	0.1	2.3	0.3	155
Extreme Rich Fen	2	>7.4	194-463	35-47	11.0-16.0	2.2-4.1	5.0-27.0	1.3-3.8	0.1-0.3	1.3-2.1	0.2	120-310
			328	41	13.5	3.2	16.0	2.5	0.2	1.7	0.2	215
All Peatlands	≤204	3.6-7.6	7-506	1-70	0.4-21.0	0.1-8.6	0.1-41.0	0.1-27.0	0.1-0.8	0.3-1.4	0.1-3.2	0-310
		6.0	130	20	6.2	1.0	2.8	1.3	0.1	2.1	0.2	62

[a] Adapted from Turchenek et al. (1984) and from authors' unpublished data.
[b] Specific conductance.
[c] Range above, mean below.

types given in Table 3.1, Section 3. Most elemental contents increased with increase in pH, or with the sequence from bog to rich fen. However, some parameters, notably P, Cl, and NO_3, did not follow this trend. A correlation matrix of these parameters is shown in Table 6.4. The matrix shows that pH is most highly correlated with electrical conductance, Ca, Mg, and Ca+Mg. Electrical conductance, Ca, Mg, Ca+Mg, and HCO_3^- were highly correlated with each other. Nitrate-N, P, and K are important plant nutrients, but they, along with Cl and S, did not appear to be correlated with each other or with any of the other variables. Many of the correlations were significant at the 5% level but they were not high.

The above observations lend support to the conclusions of Clarke-Whistler et al. (1984) wherein some water properties reflected a likely water-source influence while others were related to other factors such as biological origin and decomposition. Clarke-Whistler et al. (1984) based their conclusions on a review of some studies which demonstrated strong correlations among water chemical attributes. Calcium, Mg, Si, pH, alkalinity, and electrical conductance were found to be related and together appear to represent a groundwater influence on peatlands. Bioavailable-P, total P, NH_4^+, and K appear to be related through their organic sources and are thus dependent on degree of decomposition of organic matter. The element Na is independent of the above factors and is correlated to some extent with Cl and K, although K figures more prominently in the second group above. A combination of attributes related to acidity and redox potential also appears correlated and independent of hydrological and decomposition factors.

Other peatland water attributes which have been reported in some studies include dissolved O_2, total acidity, colour, suspended sediments, Al, Fe, NH_4-N, organic N, humic acid, fulvic acid, and chemical oxygen demand (Clarke-Whistler et al. 1984). There appear to be very few data on these attributes for Alberta peatlands. There are likewise few data on peatland runoff waters. Any available data have been collected during the course of research projects carried

Table 6.4. Correlation matrix of chemical parameters in peatland waters from the Athabasca area, Alberta.[a,b]

Variable	pH	EC	Ca	Mg	K	Na	S	P	Cl	NO_3	HCO_3	Ca+Mg[c]
pH	1.0000											
EC[c]	.6678	1.0000										
Ca[d]	.6640	.9519	1.0000									
Mg	.7362	.9286	.9276	1.0000								
K	.1712	.1290	.0465	.1428	1.0000							
Na	.3510	.6257	.4502	.4362	.1450	1.0000						
S	.1471	.3005	.2752	.2733	.0026	.2257	1.0000					
P	.0479	.1238	.1206	.1153	.2347	.1659	.0158	1.0000				
Cl	-.3328	-.0894	-.1322	-.1944	.1230	.2436	-.0009	.1103	1.0000			
NO_3	.0596	-.0702	-.0715	-.0841	.0218	-.0052	.0213	-.0848	-.0382	1.0000		
HCO_3	.6634	.9535	.9244	.9189	.1321	.5762	.1140	.1247	-.1475	-.0467	1.0000	
Ca+Mg	.6882	.9587	.9966	.9551	.0682	.4528	.2783	.1210	-.1475	-.0751	.9348	1.0000

[a] Adapted from the original tables in Turchenek et al. (1984).

[b] No. of samples = 186; degrees of freedom 184; R@0.05 = 0.1439; R @ 0.01 = 0.1884; R = critical value of correlation coefficient (r) for testing hypothesis that Ho:p=0; e.g., hypothesis that correlation coefficient = 0 between K and EC can be accepted at 5% level (since r = 0.1290 < 0.1439).

[c] EC = electrical conductance in $\mu S \cdot cm^{-1}$.

[d] Units for all parameters except pH and EC = $mg \cdot L^{-1}$.

out for specific purposes such as environmental impact assessments. These are reviewed separately in Section 7. The lack of chemical data on peatland runoff waters in Alberta is a major obstacle to evaluating impacts on water quality and necessitates that any conclusions be made on the basis of extrapolation from other regions and countries. The collection of baseline chemical data on peatland surface, subsurface and runoff waters is one of the major recommendations of this report.

A generalized summary comparing water quality characteristics of fens and bogs was prepared by Clausen and Brooks (1980) and is reproduced in Table 6.5. The summary can be used to indicate whether bogs or fens would have greater potential for affecting downstream water quality. The summary is based on data for natural peatlands from various countries, including a study in Yukon Territory by Walmsley and Lavkulich (1975), and it can readily be extrapolated to Alberta conditions.

Table 6.5. List of water quality characteristics that have greater concentrations in fens or bogs.[a]

Fen > Bog	Bog > Fen
pH	Colour
Electrical Conductance	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

[a] Source: Clausen and Brooks (1980).

6.3 NATURE AND SIGNIFICANCE OF ORGANIC MATERIALS IN NATURAL WATERS

6.3.1 Kinds, Sources, and Levels of Organic Materials

Organic materials in waters constitute a continuum of sizes from the smallest organic molecules to the relatively large phytoplankton and zooplankton organisms (Thurman 1985). The organic materials in the water are usually separated into the dissolved organic carbon (DOC), and the particulate organic carbon (POC) fractions. The distinction between DOC and POC is an operational one and depends on the filter size used in water filtration. Thurman (1985) reported that 0.45 μm is the size boundary between DOC and POC, although other workers prefer the 0.10 μm size filters. Total organic carbon refers to the sum of the DOC and POC fractions.

The forms of organic carbon in natural waters are depicted in Figure 6.1. The DOC fraction is made up mostly of humic and fulvic acids (50 to 75%), hydrophilic acids (up to 30%), low-molecular-weight organic molecules (e.g., amino acids, carbohydrates, phenols, organic acids, hydrocarbons), and materials in the colloidal size range including clay and oxide-humic acid complexes (Stumm and Morgan 1981; Thurman 1985). The POC fraction consists primarily of bacteria, zooplankton, and phytoplankton (Thurman 1985), as well as of both autochthonous and allochthonous detritus of plant and animal origin (Cole 1983). In peatlands, particular attention has been given to the dissolved component which imparts a yellow to brown colour to waters. The reasons for this are that, (1) the organic materials are acidic in nature and, therefore, influence the acid status and buffering capability of waters, and (2) the organic ligands of dissolved organic matter have a strong tendency to chelate Al and other metals, thus influencing their solubility, mobility, and toxicity in waters.

Concentrations of DOC are greatest in wetland areas as compared to other types of natural waters; they range from 3 to 400$\cdot$mg L^{-1} with an average value of 30 mg$\cdot L^{-1}$. These high values of

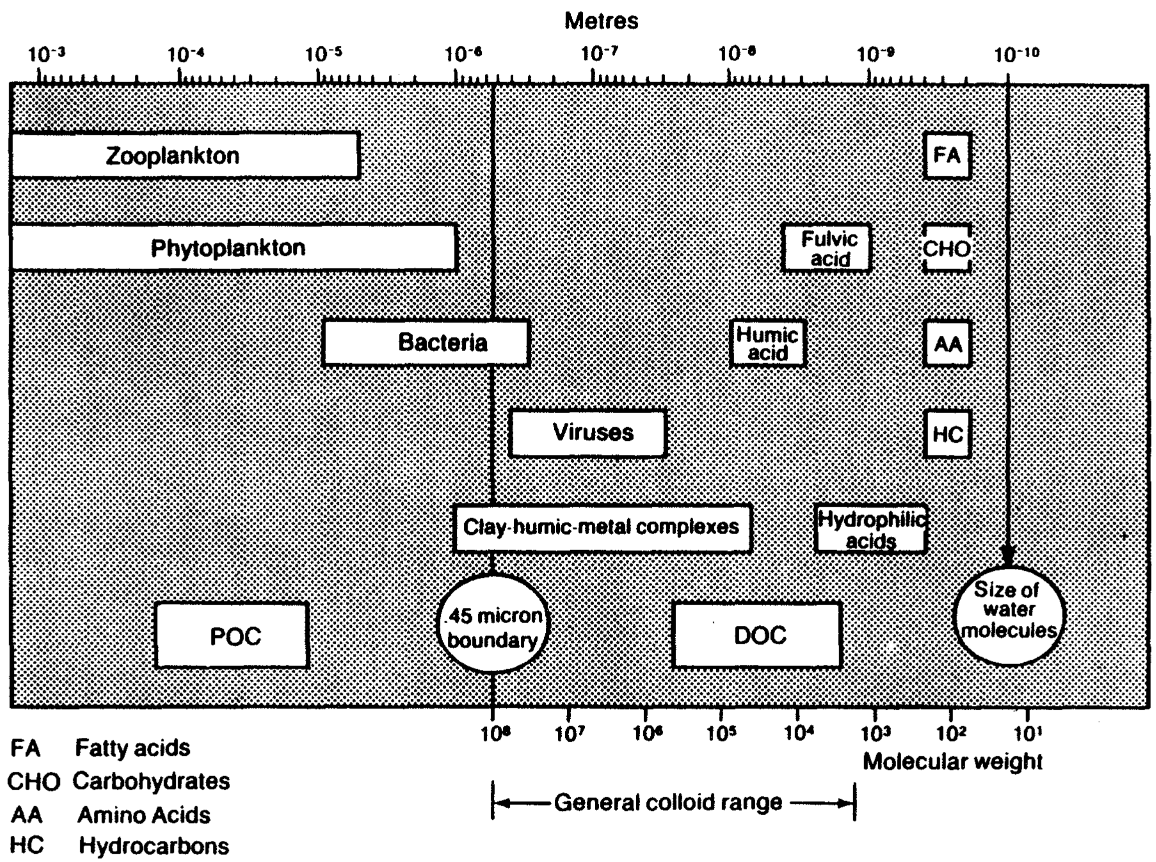

Figure 6.1. Continuum of particulate and dissolved organic carbon in natural waters. (Source: Thurman 1985).

DOC result from the accumulation of organic acids in the water from the decomposition and leaching of mosses and emergent plants and abundant surface detritus. The increase of organic carbon is accentuated by the low pH of the water, which prevents rapid bacterial decay of the organic matter. Fungi are the most important microorganisms which decompose organic matter at low pH. Of the different kinds of wetlands, marshes usually have the lowest DOC, ranging from 5 to 15 $mg \cdot L^{-1}$. Swamps have concentrations of DOC from 10 to 30 $mg \cdot L^{-1}$ and average 20 $mg \cdot L^{-1}$. Levels in transitional bogs and fens range from 3 to 10 $mg \cdot L^{-1}$, and in ombrotrophic bogs from 30 to 400 $mg \cdot L^{-1}$ (Thurman 1985). Levels of DOC in lakes and rivers are usually lower than those of wetlands. The concentration of DOC in small streams varies from 1 to 4 $mg \cdot L^{-1}$ while that of larger rivers ranges from 2 to 10 $mg \cdot L^{-1}$. Lakes generally have low DOC levels although dystrophic lakes are an exception with about 20 to 50 $mg \cdot L^{-1}$ DOC resulting from the release of yellow organic acids from adjacent wetlands and from vegetation along shorelines. The concentration of POC in swamps and marshes is commonly about 10% of the total organic carbon and that of bogs and fens varies depending on whether or not the surface water is moving or agitated in some way to keep particulates in suspension.

Wetlands represent an interesting and unusual case of water chemistry in that dissolved organic matter usually exceeds dissolved inorganic matter, which is not the normal situation in surface water bodies. About 90% of this DOC is attributable to humic substances (fulvic and humic acids) and hydrophilic acids. The dissociation of these materials contributes to a significant proportion of the anions in solution. The remaining DOC consists of identifiable compounds with abundance of different groups as follows; carbohydrates > carboxylic acids > amino acids > hydrocarbons. A more detailed listing of organic compounds and indications of their origins and transformations in waters is given in Table 6.6.

Table 6.6. Naturally occuring organic substances in aquatic systems.[a]

Life Substances	Decomposition Intermediates	Intermediates and Products Typically Found in Nonpolluted Natural Waters
Proteins	Polypeptides → $RCH(NH_2)COOH$ (amino acids) → $RCOOH$, $RCH_2OHCOOH$, RCH_2OH, RCH_3, RCH_2NH_2	NH_4^+, CO_2, HS^-, CH_4, HPO_4^{2-}, peptides, amino acids, urea, phenols, indole, fatty acids, mercaptans
Polynucleotides	Nucleotides → purine and pyrimidine bases	
Lipids Fats Waxes Oils Hydrocarbons	RCH_2CH_2COOH (fatty acids) + $CH_2OHCHOHCH_2OH$ (glycerol) → RCH_2OH, $RCOOH$, shorter chain acids, RCH_3, RH	CO_2, CH_4, aliphatic acids, acetic, lactic, citric, glycolic, malic, palmitic, stearic, oleic acids, carbohydrates, hydrocarbons
Carbohydrates Cellulose Starch Hemicellulose	→ $C_x(H_2O)_y$ → monosaccharides, oligosaccharides, chitin → hexoses, pentoses, glucosamine	HPO_4^{2-}, CO_2, CH_4, glucose, fructose, galactose, arabionose, ribose, xylose
Lignin	$(C_2H_2O)_x$ → unsaturated aromatic alcohols, polyhydroxy carboxylic acids	
Porphyrins and Plant Pigments Chlorophyll Hemin Carotenes and Xantophylls	Chlorin → pheophytin → hydrocarbons	Phytane Pristane, carotenoids Isoprenoid, alcohols, ketones, acids Porphyrins
Complex Substances Formed from Breakdown Intermediates, e.g.,	Phenols + quinones + amino compounds →	Melanins, melanoidin, gelbstoffe
	Amino compounds + breakdown products of carbohydrates →	Humic acids, fulvic acids, tannic substances

[a] Source: Stumm and Morgan 1981.

6.3.2 <u>Nature and Properties of Humic Substances</u>

Studies of organic matter in soils and waters have dealt mainly with the humic and fulvic materials. Aquatic humic substances are operationally defined as coloured, polyelectrolytic acids isolated from water by sorption onto XAD resins, weak-base ion exchange resins, or a comparable procedure (Thurman 1985). Hydrophilic acids are those humic substances which are not retained by XAD resin at pH 2. They appear to be a mixture of organic compounds that are both simple organic acids, such as volatile fatty acids and hydroxy acids, as well as complex polyelectrolytic acids that probably contain many hydroxyl and carboxyl functional groups. In fact, little is known about them as they are difficult to isolate and purify, and their study has only begun (Thurman 1985). Differentiation of humic and fulvic materials in soils is based on solubility parameters where humic materials are those soluble in alkaline but not acid solutions, and fulvic materials are soluble in both alkaline and acid solutions (Schnitzer 1978). Extraction of soil humic and fulvic materials is usually done with mild alkali (NaOH or KOH), pyrophosphate solutions, or organic solvents (Schnitzer 1978, 1982; Stevenson 1982; Hayes 1985). Methods of characterization of humic and fulvic materials include: elemental analysis (Huffman and Stuber 1985); determination of acid functional groups (Perdue 1985); ultraviolet-visible, infrared, fluorescence, Raman, and electron spin resonance spectroscopy (MacCarthy and Rice 1985); nuclear magnetic resonance spectroscopy (Wershaw 1985); molecular weight determinations (Wershaw and Aiken 1985); and electron microscopy, electron diffraction, and viscosity and surface tension measurements (Schnitzer 1978, 1984; Stevenson 1982).

Elemental analyses of aquatic humic substances have shown carbon contents of 45 to 55%, and an average of 52% (Thurman 1985). Hydrogen content of aquatic humic substances increases from 4.3% for marsh, bog, and swamp samples, to 5.0 to 5.2% for stream, river, and lake samples, and is highest (5.9%) for groundwater samples (Thurman 1985). Oxygen content is approximately 40% in humic substances of both surface waters and wetlands while that of humic substances in groundwaters is somewhat lower at 30%.

Molecular weights of humic materials are commonly determined by small-angle x-ray scattering, gel-permeation chromatography, ultrafiltration, and vapour pressure osmometry. There are limitations and problems in each of these methods and, therefore, molecular weight determinations of humic substances are usually imprecise. Nevertheless, a compilation of results from all four methods shows that: (1) aquatic fulvic acid is of molecular weight 500 to 2000 and is similar to fulvic acid from soil; and (2) aquatic humic acid is larger and often colloidal (>10 angstroms and <0.45 μm); its molecular weight ranges from 2000 to 5000 and sometimes much higher, to about 100 000 (Thurman 1985).

The structure of humic materials is commonly described by contemporary investigators in terms of a 'type' molecule which consists of micelles of a polymeric nature (Stevenson 1982). The basic structure of this micelle is an aromatic ring of the di- or trihydroxy-phenol type bridged by -O-, $-CH_2-$, -NH-, -N=, -S-, and other groups and containing both free -OH groups (phenolic OH and COOH) and the double linkages of quinones. In the natural state, the molecule may contain attached proteinaceous and carbohydrate residues.

Acid functional groups of humic substances participate in the adsorption, exchange, and complexation of hydrogen ions, micronutrients, and toxic trace metals. Humic substances generally contain strongly and weakly acid functional groups of the carboxyl and phenolic hydroxyl types respectively (Perdue 1985). Fulvic materials generally have higher total, carboxyl, and phenolic hydroxyl acidity than humic materials (Stevenson 1982). Organic acids have been found to contribute similar amounts of carboxyl acidity (about 10 $\mu mol \cdot mg^{-1}$ of organic carbon) in different kinds of freshwater systems (Oliver et al. 1983). Fulvic acids from wetlands (bogs, fens, marshes, and swamps) contain about 5.0 to 5.5 $mmol \cdot g^{-1}$ carboxyl groups and 2.5 $mmol \cdot g^{-1}$ phenolic groups. Humic acids contain 4.0 to 4.5μmmol·g^{-1} carboxyl groups and 2.5 mmol·g^{-1} phenolic groups. By comparison, streams and rivers may have slightly higher levels of carboxyl groups in fulvic acids and lower levels of phenolic groups in both fulvic and humic materials. The total acidity arising from these substances can range from 6.0 to 8.5 mmol·g^{-1} of humic substances. Further discussion of functional group analysis of humic substances can be found in Thurman (1985).

6.4 NATURE AND INTERACTIONS OF ORGANIC MATERIALS IN RELATION TO WATER QUALITY

6.4.1 Acidity

Humic substances are polyprotic acids with many of their properties determined by carboxyl and phenolic functional groups as indicated in the previous section. They can be major factors in determining the pH of natural waters and they can exert a considerable buffering capacity. In most waters with a pH between 5 and 8, fulvic and humic acids are present as organic polyanions with the magnitude of charge being pH-dependent (the higher the pH, the higher the negative charge due to the continued ionization of carboxyl and phenolic groups). These properties have several implications for water quality which include anion-cation balance, alkalinity titration, cation exchange reactions, metal complexation, and others. Organic ions contribute to the cation-anion balance, particularly in organically coloured waters of low specific conductance and moderate to low pH (typical of peatlands). Neglecting the contribution of ions to the total anionic charge can lead to serious errors in calculating the anion-cation charge balance. Furthermore, the protonation of organic acid anions during alkalinity titration can cause serious errors in alkalinity determinations (Malcolm 1985).

6.4.2 Ion Exchange

Organic material has a high cation exchange capacity (CEC) ranging from 150 to 300 $cmol \cdot kg^{-1}$. Other sediment surfaces which behave as ion exchangers include silica, with the lowest CEC at less than 1 $cmol \cdot kg^{-1}$, and expanding clay minerals such as smectites with CEC up to 100 $cmol \cdot kg^{-1}$. The cationic distribution in peat waters is controlled by organic materials since almost no inorganic ion exchangers are present. When peat waters discharge into surface water bodies containing suspended inorganic sediment, mineral surfaces, both free surfaces and those covered with organic matter, become important sites for cation exchange. Iron and manganese oxides and oxyhydroxides may serve as either cation or anion exchange sites, depending on the iso-electric point of their surfaces. Cation exchange is also an important process for nitrogen-containing organic compounds such as amino acids, polypeptides, and aliphatic and aromatic amines (Thurman 1985).

6.4.3 Sorption Properties

Fulvic and humic acids can adsorb or trap other organic substances such as alkanes, fatty acids, phthalates, and possibly also carbohydrates, peptides, and pesticides (Stumm and Morgan 1981). The sorption processes for organic substances in waters include a number of types such as hydrophobic sorption, hydrogen bonding, ligand exchange, cation exchange, and anion exchange (see Thurman (1985) for discussions of these). The reactivity, fate, and distribution of bound solutes are changed by association with aquatic humic substances. The rate of breakdown by photolysis of certain organic compounds, the rate of volatilization of polychlorinated biphenyls, the bioaccumulation of polynuclear aromatic hydrocarbons in fish, the rate of humic acid induced acid-base catalysis, and the rate of microbial decomposition are some examples of these changes (Malcolm 1985).

6.4.4 Degradation of Organic Materials by UV Irradiation

Photolysis of organic matter appears to be an important degradation mechanism in natural waters. In general, cleavage by ultraviolet (UV) radiation appears to be a prerequisite to microbial degradation of aquatic humic substances (Steinberg and Muenster 1985). Different types of organic materials and different size fractions are degraded to different extents, and complexed materials such as phosphates can be released with low doses of UV irradiation.

6.4.5 Precipitation/Flocculation

Precipitation is an important removal process for natural organic solutes. Waters of low conductance, less than 20 $\mu S \cdot cm^{-1}$, dissolve more organic carbon than waters of higher conductance (several hundred $\mu S \cdot cm^{-1}$). When there is a 10- to 100-fold increase in ionic strength, natural organic acids of low water solubility may precipitate. Consequently, humates precipitate in the presence of Ca^{2+} and Mg^{2+} and surface waters high in these cations (e.g., $>10^{-3}$ M) contain almost no humic substances (<1 mg C L^{-1}). These precipitation reactions can occur, for example, when peatland runoff mixes with water of higher ionic strength upon discharging into larger streams or rivers. Estuaries are particularly efficient in removing humic substances, probably by coagulation (by seawater cations) of humates typically bound to hydrous iron oxides. Other substances which also are removed from solution by precipitation through association with humic substances include Mn, Al, and P. Humic substances, therefore, play a major role in controlling the concentrations of inorganic elements in freshwater streams and rivers as well as in estuaries.

The precipitation process described above is sometimes referred to as 'salting out'. Hydrogen bonding or weak ion exchange mechanisms of bonding between alumina and iron hydroxide surfaces and organic acids can also lead to precipitation under certain circumstances. Waters of high acidity, organic matter content, and Al content can lose a large proportion of the Al and organic matter by precipitation by mixing with waters of higher alkalinity. This situation is analogous to the use of alum (aluminum sulphate) in water treatment plants to adsorb and precipitate organic matter (Thurman 1985). Co-precipitation of ferric iron with humic substances has also been shown (Steinberg and Muenster 1985).

Aquatic humic substances have also been found to inhibit calcium carbonate precipitation and similar processes. There are many factors involved, however. The interrelationships among the many variables, such as the amount of UV radiation, pH, water temperature, the concentration and type of humic substances, and the carbonate-bicarbonate equilibrium, lead to a nearly intractable matrix of possible outcomes (Steinberg and Muenster 1985). Humic materials may also interfere in the precipitation of other colloidal materials such as iron oxides.

6.4.6 Oxidation-Reduction

Most oxidation-reduction (redox) reactions of organic matter in water are thought to be caused by humic substances (Thurman 1985, citing various authors). Fulvic acid has a reduction potential that is approximately 0.5 v; it is a better reducing agent than humic acid which has a reducing potential of about 0.7 v. Fulvic acid can reduce a variety of substances under natural environmental conditions, including humic acids. An example of redox/photochemical reactions of natural organic matter involves oxygen consumption by a ferrous-ferric cycle in humic coloured waters. The cycle consists of reduction of ferric iron to ferrous iron by humic matter and subsequent oxidation of ferrous iron back to ferric iron by dissolved oxygen. The rate of oxygen consumption is a linear function of iron and humic colour and a nonlinear function of light energy and pH. The rate of oxygen consumption can be high and may lead to depleted oxygen levels in lakes. Peatland waters could contribute to oxygen loss in this way since they have been shown to release significant quantities of iron as well as of humic materials to downstream waters.

6.4.7 Complexation of Metal Ions

Studies of the interactions of organic matter and humic substances with metal cations are relevant to the micronutrient nutrition of plants, and the plant uptake, attenuation, and mobility of toxic trace elements. The complexation of metal ions by organic matter can be divided into different types of reactions in aquatic systems. Reactions between dissolved, suspended, and bottom sediment forms of organic matter can occur as depicted in Figure 6.2. Interaction of the metals with humic substances has been characterized by adsorption affinity sequences (Schnitzer 1978; Kerndorff and Schnitzer 1980), stability constants (Schnitzer 1978; Schnitzer and Khan 1972; Stevenson 1982; Sposito 1986), and complexation capacities (Truitt and Weber 1981; Saar and Weber 1982). Sequences of metal affinities were reported by Kerndorff and Schnitzer (1980) as follows:

Humic Acid, pH 5.8;

$$Hg = Fe = Pb = Al = Cr = Cu > Cd > Zn > Ni > Co > Mn$$

Fulvic Acid, pH 6;

$$Fe = Cr = Al > Pb = Cu > Hg > Zn = Ni = Co = Cd = Mn$$

The metal complexation by soil humic substances referred to above as well as complexation by stream humic materials does not follow the theoretical Irving-Williams stability series. Therefore, the relative order of complexation and the magnitude of the stability constant are variable (Malcolm 1985). All the methods of stability constant determination are limited in some way and often inconsistent values between and among methods are obtained (Saar and Weber 1982). Stability constants can be calculated from data for the titrations of humic substances with metal solutions, but a knowledge of the molecular weight of the organic material is necessary (Stevenson 1982; Fitch and Stevenson 1984; Sposito 1986). The difficulty in measuring the molecular weight of the humic substances, and the

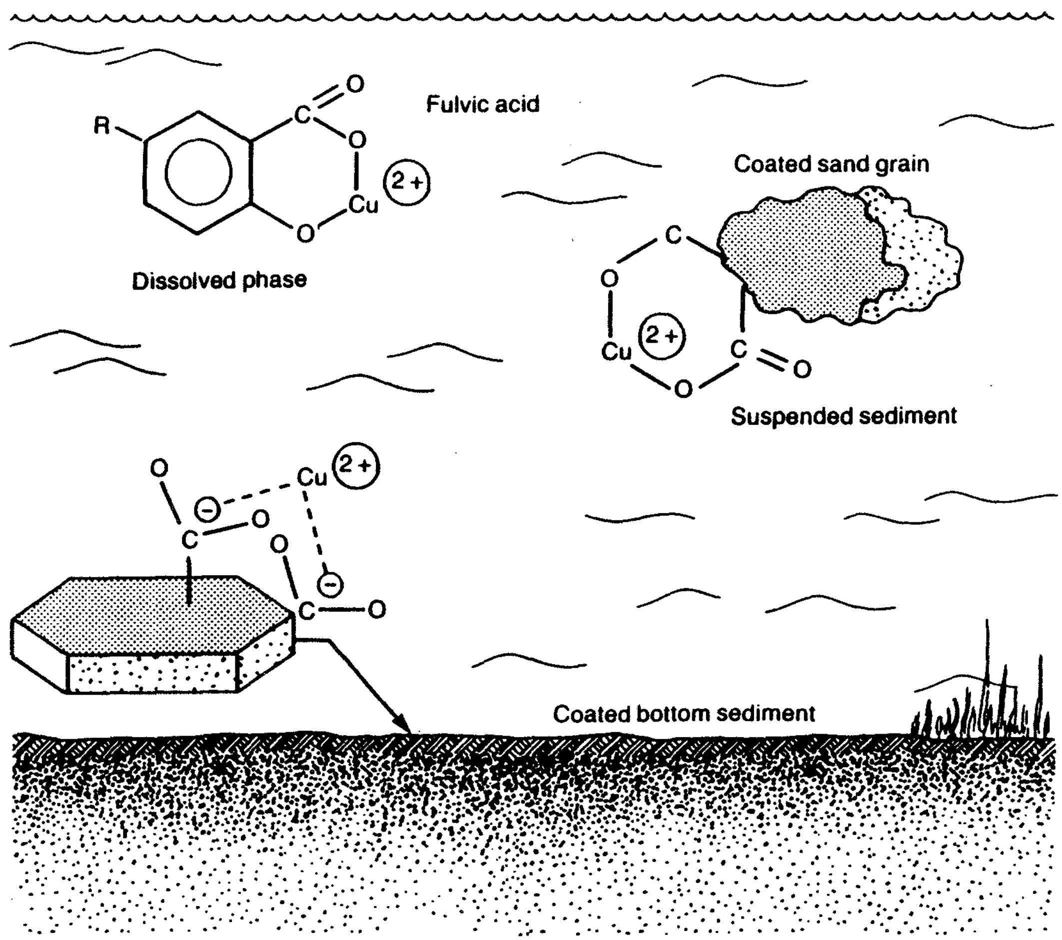

Figure 6.2. Complexation of metal ions by organic matter in suspended sediment, bottom sediment, and colloidal and dissolved phases. (Source: Thurman 1985).

sensitivity of the stability constants to the effects of pH and metal solution concentration, led researchers to measure a 'conditional stability constant'. This constant is valid only for the conditions stated and the sample examined. It is dependent on the humic material and metal concentrations, pH, and ionic strength of the solution (Thurman 1985). Several approaches used to calculate the conditional stability constants of humic substances with metal ions have been reviewed by Fitch and Stevenson (1984) and Sposito (1986). The stability constants and binding site data obtained for humic substances are best used as estimates of the intensity and the capacity of the humic materials to complex metals.

In general, H, Al, Fe, Pb, and Cu have larger stability constants than Ni, Cr, Co, Zn, or Mn. Stability constants for Ca, Mg, and Cd, are the lowest. Theoretically, monovalent cations such as Na and K can compete for complexing sites on humic substances only if their concentrations are very high (Malcolm 1985).

Metal complexation by humic substances is significant in the natural environment since it is the activity of individual species of metals, rather than their total concentration, which determines their solubility, toxicity, and availability. However, for any given system, the relative importance of organic-to-inorganic metal binding can be determined only after a thorough evaluation of the water chemistry (Malcolm 1985). Shotyk (1986a, 1986b) noted that simple organic molecules can also form extremely stable complexes with specific metals; the extent of organic complexing is probably underestimated and the significance of these complexes should not be discounted simply on the basis of their low concentrations in natural waters. Shotyk (1986a, 1986b) recommended that in order to better understand chemical speciation in peatland waters, more of the following data are required:

1. Abundance of dissolved metals, especially trace metals such as Ni and Co, and ultra-trace metals such as Ag, Cd, and Hg;
2. Abundance of dissolved ligands, including inorganic

species such as PO_4^{3-}, F^-, CN^-, SCN^-, etc. and organic species, especially aliphatic and aromatic carboxylic acids; and

3. Stability constants of metal-organic complexes.

6.5 SEASONAL VARIATION IN WATER CHEMISTRY

Seasonal differences are an important consideration in evaluating peatland water quality parameters. It has been shown in Alberta that streamflow water sources in peatland basins shift on a seasonal basis and that the water chemistry varies accordingly. These characteristics, based on a study of basins in the Athabasca oil sands area by Schwartz and Milne-Home (1982), are discussed in Chapter 4, Section 4.13 and Chapter 8, Section 8.6 of this report.

Gorham (1956) observed that concentrations of many constituents in peatland pools increased in dry weather due to evaporation. In runoff waters from peatlands, concentrations of many constituents appear to be 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; electrical conductance, acidity, K, Ca, Mn, Fe, Al, and SO_4 (Sparling 1966; Walmsley and Lavkulich 1975; Verry 1975). Verry (1975) also stressed the importance of considering flow rates when evaluating water quality by showing that annual yields on an aerial basis ($kg \cdot ha^{-1} \cdot y^{-1}$) 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 (total-P, total-N, and total-Fe) had similar values in equal volumes of streamflow for perched bogs and groundwater fens but that concentrations of minerals in groundwater solution were higher in the fens.

6.6 WATERSHED INFLUENCES ON CHEMISTRY OF PEATLAND RUNOFF WATERS

In view of the numerous interactions of organic materials and other solutes as indicated in the above discussions, the chemistry of peatland water is likely to change significantly upon runoff into streams. The influence of location and extent of a peatland within a drainage basin is an important consideration in evaluating the chemistry of downstream waters. Peatlands may occupy a part or all of a basin of first, second, or higher order (Figure 6.3). If a peatland occupies a first-order basin, the chemistry of the water in the first-order stream within that basin will likely be similar to that of the water within the peatland. There is a greater contribution of overland flow and interflow, both of which are in contact with upper peat layers, to the stream discharge in first-order basins. In the lower parts of basins, groundwater discharge becomes a more significant contributor to streamflow. Consequently, as peatland runoff water enters second- and third-order streams, there will be (1) a dilution effect, and (2) possible reactions with solutes in the higher-order streams.

The influence of peatlands on the chemistry of downstream waters within a particular basin depends on the proportion of peatlands within that basin. It follows that the magnitude of alteration of water chemistry due to development of peatlands depends on the area and proportion of peatlands developed within a basin. This also applies to alteration of the water balance in basins.

6.7 SUMMARY

Information about the chemistry of peatland waters is required for predicting and understanding possible impacts of drainage, acid deposition, and other human influences on peatlands and associated waters. The main attributes of peatland waters discussed in this chapter are summarized below.

Information about the water chemistry of natural peatlands in Alberta is limited to a few studies in which pH, electrical

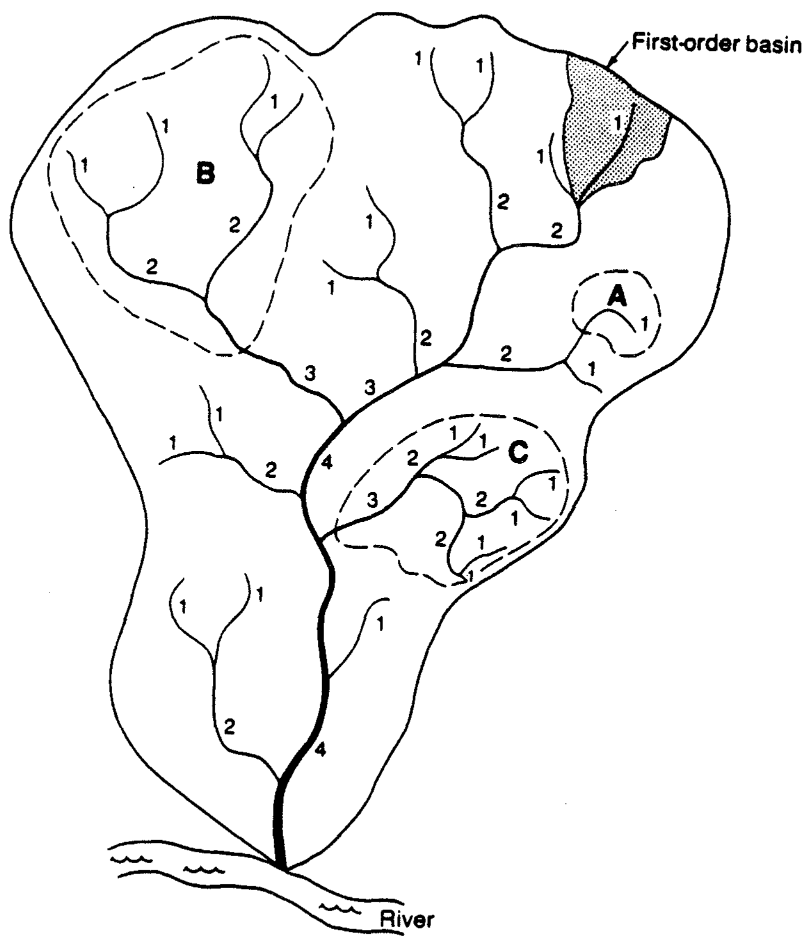

Figure 6.3. Stream orders and peatland occupation of basins with (A) one, (B) two, and (C) three orders of streams. (Adapted from Hewlett 1982).

conductance, and major ions have been reported. There is little or no information about nutrients, minor or trace elements, and properties of organic matter. However, levels of peat water constituents in Alberta are likely similar to those found in other provinces and countries.

Seasonal variations occur in peatland runoff flow rates and in concentrations of many constituents of waters. High flows produce higher values of pH, dissolved O_2, and K, and lower levels of electrical conductance, acidity, Ca, Mn, Fe, Al, and SO_4.

The major ions in peatland waters are Ca^{2+}, Mg^{2+}, H^+, and HCO_3^-. These are highly correlated with each other, and with pH and electrical conductance. Determinations of pH or electrical conductance are, therefore, useful and easily measured indicators of peat type.

Peatland waters are characterized by high levels of dissolved organic materials; bog waters are generally higher in these coloured substances than fen or swamp waters.

Functional groups on humic materials participate in the adsorption, exchange, and complexation of hydrogen ions, micronutrients, and toxic trace metals. They thus buffer acidity and bind trace elements, thereby reducing toxicity of certain elements. However, they may maintain the mobility of complexed substances and thus cause potential problems downstream.

Interactions of humic materials with other substances are highly complex. Interactions and reactions include: sorption of other organics, degradation by photolysis, precipitation or flocculation in waters of sufficient ionic strength, oxidation-reduction, and complexation of metal ions.

The contributions of peatlands to the chemistry of downstream waters depends on their extent in the watershed. Moreover, the amounts of inorganic and organic solutes in peatland and runoff waters can vary seasonally.

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7. PEATLAND AND RUNOFF WATER PROPERTIES IN RELATION TO SURFACE WATER QUALITY GUIDELINES

L.W. Turchenek

7.1 INTRODUCTION

Wetland waters interact with other biotic and abiotic components of the ecosystem, and hydrochemistry can thus be altered by passage through a wetland ecosystem (Kadlec and Kadlec 1978). Nutrients and other dissolved constituents, heavy metals, suspended solids, and microbiota can move into and out of the system with entering and receiving waters. Additions and removals of these constituents can occur simultaneously, and their concentrations can be altered by uptake, cycling, dilution, precipitation, and other mechanisms which transfer or transform them. Water quality parameters are influenced by these processes in wetlands. Certain water quality parameters established by society to protect water users may not be met by some systems in their natural state. Anthropogenic disturbances of peatlands may serve to aggravate such situations and, moreover, may alter the kinds and rates of some processes thus leading to unwanted changes in some of the water quality parameters.

To interpret the performance of a peatland in altering any water quality parameter, a knowledge of the hydrology of the wetland is required, as well as of concentrations, mass flow, and storage of the constituents of interest. Data on hydrologic inputs and outputs are necessary in evaluating the total change in a parameter when using a mass balance approach. However, few water quality studies in wetlands include simultaneous determinations of hydrology and concentrations of constituents of interest. Evaluations of effects of peatlands on water quality parameters are further complicated by the time scales of processes. Both stochastic and time-varying effects are of great importance in wetlands and they can be important on diurnal, seasonal, or even historic time scales (Kadlec and

Kadlec 1978). There probably have been no wetland studies in which a majority of water quality parameters, along with complete hydrologic measurements and all time scales of interest, have been addressed. The information on water quality in relation to peatlands is thus largely from fragmented and diverse sources. Conclusions from a review of such studies are mainly qualitative, and in many instances it is difficult to make any definite conclusions at all.

There are different frameworks within which water quality parameters in relation to peatlands can be viewed. At the largest scale, the peatland can be viewed in its entirety. This is a kind of 'black box' approach in which assessments are made only on the basis of inputs and outputs of the system. To break the system down into more detail, it can be subdivided into compartments such as the water, soil, and biotic components. These compartments can be broken down further, and the exchanges of substances between them can be addressed in terms of fluxes between compartments. The approach taken in this report is mainly to consider the inputs and outputs of the aquatic component of peatland systems. Other components such as soils and biota, further subdivision of these into smaller compartments, and the dynamics of processes between them are discussed insofar as they help to explain the reasons for behaviour of water quality parameters.

There are three general areas of peatland development or impingement which are of environmental concern with regard to water quality and quantity. One area involves development of peatlands for uses such as agriculture and peat moss extraction. A common feature of these developments is that peatlands are drained. Another area involves deposition of materials of anthropogenic origin on otherwise natural peatlands. Examples are atmospheric deposition of pollutants, oil spills, and brine spills. A third area of concern involves physical disturbances of peatlands but no actual exploitation of the peatland. Examples include construction of pipelines, roads, and other mainly linear features through peatlands. Consequences of human activities on peatlands are discussed in this section without stratification into types of activities. Specific kinds of activities and their influences on peatlands are discussed in subsequent sections.

In order to differentiate among potential changes in peatland waters brought about by human activities, it is helpful to refer to different classes of peatlands, particularly in terms of their hydrologic properties. The classes, as described in Chapter 3 will be used to discuss these changes in the following sections (i.e., bog, poor fen, transitional rich fen, extreme rich fen). There are numerous water quality parameters that describe the peatland system and associated waters. An understanding of the importance of these parameters and of their perceived safe levels for various uses is required. Descriptions of the various parameters and discussion of levels in developed and in undisturbed peatlands in relation to surface water quality guidelines are presented in this section.

The various water quality parameters are discussed individually or in groups of related properties in this section. The general sequence follows the grouping of parameters in the CCREM (1987) water quality guidelines, the groups being inorganic, organic, physical, and biological parameters. The sequence also more or less follows the natural categories of water quality parameters described by Kadlec and Kadlec (1978). The first group consists of dissolved substances that occur in varying concentrations in most natural waters. The second category comprises the nitrogen and phosphorus compounds, because often they are nutrients of greatest importance to a wetland ecosystem, and they are also a measure of the degree of eutrophication of the system itself. Heavy metals, which can be passed up the food chain with possible deleterious results, form another category; however, the category is referred to as minor and trace elements in this report as elements other than those of the metallic groups (e.g., boron) are usually included. Man-made chemicals such as pesticides and industrial wastes also affect water quality; Kadlec and Kadlec (1978) called these refractory chemicals, due to the persistence of many of these in the environment. Suspended solids and pathogens are other major categories according to Kadlec and Kadlec (1978), but these are included with physical and biological parameters, respectively, in this report.

7.2 WATER QUALITY GUIDELINES

Guidelines for surface water quality in Alberta are presented in 'Alberta Surface Water Quality Objectives' (Alberta Environment 1977). There is currently a transition, however, from use of the above guidelines to those recently developed by the Canadian Council of Resource and Environment Ministers Task Force on Water Quality Guidelines (CCREM 1987). The CCREM guidelines are used for comparisons with peatland water data in this review. Guidelines selected for comparison are those for freshwater aquatic life, for domestic drinking water quality, and for livestock drinking water quality. The CCREM guidelines for these are reproduced in Tables 6.1, 6.2, and 6.3. There are few guidelines for recreational water quality and aesthetics, and these are introduced in the discussions below as necessary. Guidelines for irrigation and industrial water supplies are not discussed herein as these are not likely to be major uses in peatland areas.

In the following sections, water quality data for drainage waters from developed, and some undeveloped, peatlands are presented. The data sources are mainly other countries, although some information originates from other provinces as well. In many instances the data are discussed in terms of the Canadian (i.e., CCREM) water quality guidelines. This is done for comparative purposes. It cannot be concluded, for example, from a study of developed peatlands in another country in which water quality parameters exceed Canadian water quality guidelines, that in fact a similar development in Canada will lead to infringement of guidelines. However, such studies identify potential problems which should be examined. The Canadian guidelines are useful as a reference to which trends noted in the literature can be related.

Table 7.1. CCREM guidelines for freshwater aquatic life.

Parameter	Guideline	Comments
	Inorganic parameters	
Aluminum[a]	0.005 $mg \cdot L^{-1}$	pH<6.5;$[Ca^{2+}]$<4.0 $mg \cdot L^{-1}$; DOC<2.0 $mg \cdot L^{-1}$
	0.1 $mg \cdot L^{-1}$	pH≥6.5;$[Ca^{2+}]$≥4.0 $mg \cdot L^{-1}$; DOC≥2.0 $mg \cdot L^{-1}$
Antimony	ID[b]	
Arsenic	0.05 $mg \cdot L^{-1}$	
Beryllium	ID	
Cadmium	0.2$\mu g \cdot L^{-1}$	Hardness[c] 0-60 $mg \cdot L^{-1}$
	0.8$\mu g \cdot L^{-1}$	Hardness 60-120 $mg \cdot L^{-1}$
	1.3$\mu g \cdot L^{-1}$	Hardness 120-180 $mg \cdot L^{-1}$
	1.8$\mu g \cdot L^{-1}$	Hardness >180 $mg \cdot L^{-1}$
Chlorine (total residual)	2.0 $\mu g \cdot L^{-1}$	Measured by amperometric or equivalent method
Chromium	0.02 $mg \cdot L^{-1}$	To protect fish
	2.0 $\mu g \cdot L^{-1}$	To protect aquatic life, including zooplankton and phytoplankton
Copper	2 $\mu g \cdot L^{-1}$	Hardness 0-60 $mg \cdot L^{-1}$
	2 $\mu g \cdot L^{-1}$	Hardness 60-120 $mg \cdot L^{-1}$
	3 $\mu g \cdot L^{-1}$	Hardness 120-180 $mg \cdot L^{-1}$
	4 $\mu g \cdot L^{-1}$	Hardness >180 $mg \cdot L^{-1}$
Cyanide	5.0 $\mu g \cdot L^{-1}$	Free cyanide as CN
Dissolved oxygen	6.0 $mg \cdot L^{-1}$	Warm-water biota - early life stages
	5.0 $mg \cdot L^{-1}$	- other life stages
	9.5 $mg \cdot L^{-1}$	Cold-water biota - early life stages
	6.5 $mg \cdot L^{-1}$	- other life stages
Iron	0.3 $mg \cdot L^{-1}$	
Lead	1 $\mu g \cdot L^{-1}$	Hardness 0-60 $mg \cdot L^{-1}$
	2 $\mu g \cdot L^{-1}$	Hardness 60-120 $mg \cdot L^{-1}$
	4 $\mu g \cdot L^{-1}$	Hardness 120-180 $mg \cdot L^{-1}$
	7 $\mu g \cdot L^{-1}$	Hardness >180 $mg \cdot L^{-1}$
Mercury	0.1 $\mu g \cdot L^{-1}$	
Nickel	25 $\mu g \cdot L^{-1}$	Hardness 0-60 $mg \cdot L^{-1}$
	65 $\mu g \cdot L^{-1}$	Hardness 60-120 $mg \cdot L^{-1}$
	110 $\mu g \cdot L^{-1}$	Hardness 120-180 $mg \cdot L^{-1}$
	150 $\mu g \cdot L^{-1}$	Hardness >180 $mg \cdot L^{-1}$

continued . . .

Table 7.1. Continued.

Parameter	Guideline	Comments
Nitrogen		
Ammonia (total)	2.2 $mg \cdot L^{-1}$	pH 6.5; temperature 10°C
	1.37 $mg \cdot L^{-1}$	pH 8.0; temperature 10°C
Nitrite	0.06 $mg \cdot L^{-1}$	
Nitrate		Concentrations that stimulate prolific weed growth should be avoided.
Nitrosamines	ID	
pH	6.5-9.0	
Selenium	1 $\mu g \cdot L^{-1}$	
Silver	0.1 $\mu g \cdot L^{-1}$	
Thallium[d]	ID	
Zinc	0.03 $mg \cdot L^{-1}$	
	Organic parameters	
Acrolein	ID	
Aldrin/dieldrin[d]	4 $ng \cdot L^{-1}$ (dieldrin)	
Benzene	0.3 $mg \cdot L^{-1}$	
Chlordane	6 $ng \cdot L^{-1}$	
Chlorinated benzenes[c]		
Monochlorobenzene	15 $\mu g \cdot L^{-1}$	
Dichlorobenzene		
1,2- and 1,3-	2.5 $\mu g \cdot L^{-1}$	
1,4-	4.0 $\mu g \cdot L^{-1}$	
Trichlorobenzene 1,2,3-	0.9 $\mu g \cdot L^{-1}$	
1,2,4-	0.5 $\mu g \cdot L^{-1}$	
1.3.5-	0.65 $\mu g \cdot L^{-1}$	
Tetrachlorobenzene 1,2,3,4-	0.10 $\mu g \cdot L^{-1}$	
1,2,3,5-	0.10 $\mu g \cdot L^{-1}$	
1,2,4,5-	0.15 $\mu g \cdot L^{-1}$	
Pentachlorobenzene	0.030 $\mu g \cdot L^{-1}$	
Hexachlorobenzene	0.0065 $\mu g \cdot L^{-1}$	

continued . . .

Table 7.1. Continued.

Parameter	Guideline	Comments
Chlorinated ethylenes[d]		
Tetrachloro-ethylene	260 $\mu g \cdot L^{-1}$	
Di- and tri-chloroethylenes	ID	
Chlorinated phenols		
Monochlorophenols	7 $\mu g \cdot L^{-1}$	
Dichlorophenols	0.2 $\mu g \cdot L^{-1}$	
Trichlorophenols	18 $\mu g \cdot L^{-1}$	
Tetrachloro-phenols	1 $\mu g \cdot L^{-1}$	
Pentachlorophenol	0.5 $\mu g \cdot L^{-1}$	
DDT	1 $ng \cdot L^{-1}$	
Dinitrotoluenes	ID	
Diphenylhydrazine	ID	
Endosulfan	0.02 $\mu g \cdot L^{-1}$	
Endrin	2.3 $ng \cdot L^{-1}$	
Ethylbenzene[d]	0.7 $mg \cdot L^{-1}$	
Halogenated ethers	ID	
Heptachlor + Heptachlor epoxide	0.01 $\mu g \cdot L^{-1}$	
Hexachloro-butadiene	0.1 $\mu g \cdot L^{-1}$	
Hexachloro-cyclohexane isomers	0.01 $\mu g \cdot L^{-1}$	
Hexachloro-cyclopentadiene	ID	
Phenols (total)	1 $\mu g \cdot L^{-1}$	
Nitrobenzene	ID	
Nitrophenols	ID	
Phenoxy herbicides (2,4-D)	4.0 $\mu g \cdot L^{-1}$	

continued . . .

Table 7.1. Concluded.

Parameter	Guideline	Comments
Phthalate esters		
DBP	4 $\mu g \cdot L^{-1}$	
DEHP	0.6 $\mu g \cdot L^{-1}$	
Other phthalate esters	0.2 $\mu g \cdot L^{-1}$	
Polychlorinated biphenyls (total)	1 $ng \cdot L^{-1}$	
Polycyclic aromatic hydrocarbons	ID	
Toluene	0.3 $mg \cdot L^{-1}$	
Toxaphene	8 $ng \cdot L^{-1}$	
	Physical parameters	
Temperature		Thermal additions should not alter thermal stratification or turnover rates, exceed maximum weekly mean temperatures, and exceed maximum short-term temperatures.
Total suspended solids	increase of 10.0 $mg \cdot L^{-1}$	Background suspended solids≤100.0 $mg \cdot L^{-1}$
	increase of 10% above background	Background suspended solids >100.0 $mg \cdot L^{-1}$

[a] Concentrations of heavy metals reported as total metal in an unfiltered sample.
[b] ID = insufficient data to recommend a guideline.
[c] Expressed as $CaCO_3$.
[d] Tentative guideline.

Table 7.2 CCREM guidelines for domestic drinking water quality.

Parameter	Maximum acceptable concentration ($mg \cdot L^{-1}$)[a,b]
	Inorganic Parameters
Antimony	--
Arsenic	0.05
Asbestos	--
Barium	1.0
Boron	5.0
Cadmium	5 $\mu g \cdot L^{-1}$
Chloride	250
Chromium	0.05
Copper	1.0
Cyanide	0.2
Fluoride	1.5
Hardness	--
Iron	0.3
Lead	0.05
Manganese	0.05
Mercury	1 $\mu g \cdot L^{-1}$
Nitrate (as N)	10.0[c]
Nitrite (as N)	1.0
pH	6.5-8.5[d]
Selenium	0.01
Silver	0.05
Sulphate	500
Sulphide (as H_2S)	0.05
Total dissolved solids	500
Uranium	0.02
Zinc	5.0

continued . . .

Table 7.2 Continued.

Parameter	Maximum acceptable concentration $(mg \cdot L^{-1})$[a,b]
	Organic Parameters
Aldrin + dieldrin	0.7 $\mu g \cdot L^{-1}$
Carbaryl	70 $\mu g \cdot L^{-1}$
Chlordane (total isomers)	7 $\mu g \cdot L^{-1}$
2,4-D	0.1
DDT (total isomers)	0.03
Diazinon	14 $\mu g \cdot L^{-1}$
Dieldrin + aldrin	0.7 $\mu g \cdot L^{-1}$
Endrin	0.2 $\mu g \cdot L^{-1}$
Heptachlor + heptachlor epoxide	3 $\mu g \cdot L^{-1}$
Lindane	4 $\mu g \cdot L^{-1}$
Methoxychlor	0.1
Methyl parathion	7 $\mu g \cdot L^{-1}$
Nitrilotriacetic acid (NTA)	0.05
Parathion	35 $\mu g \cdot L^{-1}$
Pesticides (total)[e]	0.1
Phenols	2 $\mu g \cdot L^{-1}$
2,4,5-TP	0.01
Toxaphene	5 $\mu g \cdot L^{-1}$
Trihalomethanes	0.35
	Physical Parameters
Colour	15 TCU[f]
Odour	--
Taste	--
Temperature	15°C
Turbidity	5 NTU[g]
	Radiological Parameters[h]
^{137}Cs (Cesium)	50 $Bq \cdot L^{-1}$
^{131}I (Iodine)	10 $Bq \cdot L^{-1}$
^{226}Ra (Radium	1 $Bq \cdot L^{-1}$
^{90}Sr (Strontium	10 $Bq \cdot L^{-1}$
^{3}H (Tritium)	40 000 $Bq \cdot L^{-1}$

continued . . .

Table 7.2 Concluded.

Parameter	Maximum acceptable concentration ($mg \cdot L^{-1}$)[a,b]
	Microbiological Parameters
Microorganisms	a. No sample should contain more than 10 total coliform organisms per 100 mL; b. Not more than 10% of the samples taken in a 30-d period should show the presence of coliform organisms; c. Not more than two consecutive samples from the same site should show the presence of coliform organisms; and d. None of the coliform organisms detected should be fecal coliforms.

[a] Unless otherwise indicated.
[b] Total unless otherwise indicated.
[c] Where both nitrate and nitrite are present the total nitrate- plus nitrite-nitrogen should not exceed 10 $mg \cdot L^{-1}$.
[d] Logarithmic scale, no units.
[e] The 'total pesticide' limit applies to water in which more than one of the pesticides mentioned in Chapter 1 are present. The sum of their concentrations should not exceed 0.1 $mg \cdot L^{-1}$.
[f] True colour units.
[g] Nephelometric turbidity units.
[h] Becquerel.

Table 7.3 CCREM guidelines for livestock drinking water quality.[a]

Parameter	Guideline ($mg \cdot L^{-1}$)
	Major Ions and Nutrients
Calcium	1000
Nitrate plus nitrite	100
Nitrite alone	10.0
Sulphate	1000
Total dissolved solids (salinity)	3000
	Heavy Metals and Trace Ions[b]
Aluminum	5.0
Arsenic	0.5 (5.0 when not added to feed)
Beryllium[c]	0.1
Boron	5.0
Cadmium	0.02
Chromium	1.0
Cobalt	1.0
Copper	1.0 (cattle) 5.0 (swine and poultry) 0.5 (sheep)
Fluoride	2.0 (1.0 if feed contains fluoride)
Iron[d]	--
Lead	0.1
Manganese[d]	--
Mercury	0.003
Molybdenum	0.5
Nickel	1.0
Selenium	0.05
Uranium	0.2
Vanadium	0.1
Zinc	50.0
	Biological Parameters
Blue-green algae	Avoid heavy growths of blue-green algae
Pathogens and parasites	Water of high quality should be used

[a] For pesticides, see Domestic Drinking Water Guidelines, Table 7.2.
[b] Guidelines expressed as total concentrations.
[c] Tentative guideline.
[d] No guideline recommended at this time.

7.3 MAJOR DISSOLVED SUBSTANCES

7.3.1 Dissolved Oxygen

Adequate amounts of dissolved oxygen must be available for fish and other aquatic organisms. The influences of oxygen on aquatic life and the factors affecting oxygen levels are discussed by Alberta Environment (1977), McNeely et al. (1979), CCREM (1987), and Kadlec and Kadlec (1978). Parameters closely related to oxygen include biological oxygen demand (BOD) and chemical oxygen demand (COD); these are discussed below. Only one study reporting oxygen levels in peatland waters has been found. Clausen and Brooks (1983a, 1983b) carried out a comprehensive study of quality of runoff from undisturbed and mixed peatlands in Minnesota. Their data for oxygen, shown in Table 7.4, suggest that peatland waters themselves are marginal with respect to guidelines for protection of aquatic life. However, oxygen levels are dependent on so many factors that it is difficult to predict how levels in peatland runoff might influence levels in downstream waters.

7.3.2 Chemical Oxygen Demand and Biochemical Oxygen Demand

These parameters are included in this section because of their close relationship to total oxygen. There are no CCREM (1987) guidelines for BOD and COD. The Alberta Environment (1977) surface water quality objective is as follows: dependent on the assimilative capacity of the receiving water, the BOD must not exceed a limit that would create a dissolved O_2 content of less than 5 $mg \cdot L^{-1}$. McNeely et al. (1979) indicated that no specific guidelines for BOD have been proposed, but waters with BOD levels less than 4 $mg \cdot L^{-1}$ are deemed reasonably clean while levels in excess of 10 $mg \cdot L^{-1}$ are considered polluted. McNeely et al. (1979) also discussed chemical oxygen demand (COD), but no water quality guidelines have been proposed.

BOD levels of peatland waters have been found in only one report. COD levels have been reported in a number of studies, as

Table 7.4. Dissolved oxygen in runoff of natural and developed peatlands in Minnesota.

Source	Peatland Type	Peatland Use	No. of Samples	O_2 $mg \cdot L^{-1}$	% O_2 Saturation	Temperature °C
Clausen 1980	fen	natural	46	6.2	49	9
Minnesota	fen	natural	14	7.2	72	15
	bog	mined	46	4.4	40	11
	fen	cultivated	12	2.8	30	18
Clausen and	fen	natural	54	5.6	53	13
Brooks	fen	natural	55	5.4	50	13
1983a,b	bog	natural	66	6.4	56	11
Minnesota	bog	mined	5	4.8	70	16

indicated in Table 7.5. COD is a measure of the amount of oxygen required to chemically oxidize the organic matter in a water sample; it estimates the amount of organic and other reducing material present in water. It may also be referred to as oxygen consumed on dichromate oxygen demand. However, acid potassium permanganate and other oxidizing agents can also be used for this procedure (Golterman and Clymo 1969). The dissolved COD is measured on samples which have been filtered or centrifuged to separate particulate organic matter. BOD is the amount of oxygen required to oxidize the organic matter by aerobic microbial decomposition. BOD usually refers to the amount of oxygen consumed over a 5-day period at an incubation temperature of 20°C. COD almost always exceeds BOD in water. However, there is not necessarily a correlation between the two measures (McNeely et al. 1979). Therefore, the COD values reported in the literature do not necessarily reflect relative BOD values among different kinds of peat waters.

The water quality guideline for BOD is expressed in terms of its influence on O_2 levels in water. Organic carbon affects several other water properties, and has been the subject of considerable interest and debate in recent years. A separate review of this topic is provided in Chapter 6.

7.3.3 pH, Acidity and Alkalinity

The Canadian water quality guidelines indicate that pH should be in the range of 6.5 to 9.0 for freshwater aquatic life and 6.5 to 8.5 for domestic usage. Alberta Environment (1977) and the International Joint Commission (1977) also suggested that the pH should not be altered by more than 0.5 units from the background value.

The pH is a measure of the intensity of acidity or alkalinity and is related to taste, corrosiveness, efficiency of chlorination, coagulation, and suitability for industrial application. Aquatic life can only survive in a narrow range of pH, and excessive pH levels can cause eventual destruction and sterilization of a

Table 7.5. Chemical oxygen demand and biochemical oxygen demand in natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	COD[a] $mg \cdot L^{-1}$	BOD[b] $mg \cdot L^{-1}$
Brown et al. 1980	shallow	wild rice	supply	-	-	2.0
Minnesota	peat		discharge	-	-	3.3
	soil					
Clausen and	fen	natural	runoff	54	97	-
Brooks 1983a.b	fen	natural	runoff	55	104	-
Minnesota	bog	natural	runoff	66	118	-
	bog	mined	runoff	5	143	-
Largin 1976	bog	natural	pit	-	90-270	-
RSFSR	bog	cultivated	ditch	-	130-290	-
	fen	natural	pit	-	15-120	-
	fen	cultivated	ditch	-	30-90	-
Sallantaus and	bog	natural	weir	several	30-80	-
Patila 1983	bog	mined	weir	from	51-56	-
Finland	bog	mined	weir	April	39-69	-
	bog	mined	weir	to	89-93	-
	bog	mined	weir	October	54-83	-
Washburn and Gillis Associates Ltd. 1983						
Newfoundland	bog	natural	pit	2	43-95	-
Ontario	bog	natural	pit	2	107-126	-
British Columbia	bog	mined	pit	2	914-996[c]	-
					292-326	
			recycle pond		17000	-

[a] COD = chemical oxygen demand.

[b] BOD = biochemical oxygen demand.

[c] Value for total COD; all others in column are for dissolved COD.

watershed area (Alberta Environment 1977). The pH of water can influence the species composition of an aquatic environment and affect the availability of nutrients and the relative toxicity of many trace elements.

The pH of peat and peat waters was discussed previously in Chapter 6. In this section, only those research reports that compare developed and undeveloped peatlands are reviewed so that a general assessment of the overall impact on pH can be made. Data from various reports are summarized in Table 7.6.

Most studies have indicated that the pH values of runoff waters from developed peatlands are about the same as or higher than those in undeveloped peatlands. Also, most studies compare natural peatlands with peatlands that have been developed for some time. Monitoring of runoff prior to as well as after development has not generally been carried out. The Newfoundland study (Osborne 1983; Table 7.6) is unique in that water quality both upstream and downstream from a mined bog was determined. The results showed that pH dropped during passage of the watercourse through the peatland, but it increased to near upstream values upon leaving the peatland. In Minnesota, it has generally been found that pH increases a short distance downstream from outlets of mined peatlands due to the influence of mineral contact in the stream bed (N. Aaseng, Minnesota Department of Natural Resources, Minneapolis, 1986).

Acidity refers to the capacity of water to neutralize hydroxyl ions. It is operationally defined as the amount of base required to bring a sample to pH 8.3 (Golterman and Clymo 1969). Acidity is a capacity factor which accounts for reserve acidity of the system as well as its free acidity. The pH, on the other hand, is an intensity factor that reflects only the active acidity in solution, that is, the amount of hydronium in solution. Sources of acidity include free carbon dioxide, carbonic acid, mineral acids such as sulphuric acid, bisulphate (HSO_4), phosphoric acid, and organic materials such as simple organic acids and more complex humic substances.

Table 7.6. pH, acidity, and alkalinity in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	pH	Acidity $mg \cdot L^{-1}$	Alkalinity $mg \cdot L^{-1}$
Clausen 1980	fen	natural		46	6.1	21	25
Minnesota	fen	natural	runoff	14	7.0	13	56
	bog	mined	runoff	46	5.2	46	11
	fen	cultivated	runoff	12	6.5	52	74
Clausen and Brooks 1983a,b	fen	natural		54	7.0	18	74
Minnesota	fen	natural		55	6.5	17	29
	bog	natural		66	5.6	21	10
	bog	mined	ditch	5	5.6	45	9
Ferda and Novak 1976		natural	runoff	10	3.5	-	-
Czechoslovakia		drained-forestry	runoff	10	3.9	-	-
		drained-mineral	runoff	10	4.4	-	-
Heikurainen et al. 1978	bog	natural	-	-	5.8	-	-
Finland	bog	drained-forestry	-	-	6.2	-	-
	swamp	natural	-	-	4.5	-	-
	swamp	drained, fertilized-forestry	-	-	5.0	-	-
Kenttamies and Laine 1984	bog	undrained (3y)	-	-	4.0	-	-
Finland	bog	drained-forestry (2y)	-	-	4.6	-	-
	bog	fertilized, forestry (2y)	-	-	4.7	-	-
Largin 1976	bog	natural	pit	-	3.5-5.9	-	-
RSFSR	bog	cultivated	ditch	-	4.0-6.6	-	-
	fen	natural	pit	-	5.3-7.1	-	-
	fen	cultivated	ditch	-	6.6-7.7	-	-
Sallantaus and Patila 1983	bog	natural	weir	several	3.7-4.3	-	-
Finland	bog	mined	weir	from	4.2-6.4	-	-
	bog	mined	weir	April	4.8-6.5	-	-
	bog	mined	weir	to	4.4-5.3	-	-
	bog	mined	weir	October	3.9-4.7	-	-
Osborne, in Washburn and Gillis Associates Ltd. 1983	bog	mined	upstream	13-15	6.1-6.9	-	
Newfoundland			discharge	from	3.8-4.6	-	-
			downstream	July - Nov	5.9-6.6	-	-

There are no water quality guidelines for acidity. Clausen (1980) and Clausen and Brooks (1983a, 1983b) reported acidity in some investigations of natural and developed peatlands in Minnesota (Table 7.6). Clausen and Brooks (1980) cited McKee and Wolfe (1963) who indicated that waters receiving acid coal mine drainage having a pH of 4.0 and a total acidity of 100 $mg \cdot L^{-1}$ could not sustain fish life. Levels of acidity reported by Clausen (1980) and Clausen and Brooks (1983a, 1983b) reached only half of the 100 $mg \cdot L^{-1}$ level. However, levels were higher in cultivated and mined peatland runoff than in runoff from undeveloped peatlands. It would appear that much of this elevated acidity is due to increases in levels of organic material in the runoff from developed peatlands. The discussions in Chapter 6, Sections 6.3 and 6.4, also indicate that organic materials in natural waters are a much more significant source of acidity than was previously thought.

Alkalinity is a measure of the capacity of water to neutralize acidity. It is also a capacity factor and is linked to the pH intensity factor. Sources of alkalinity are mainly carbonates, bicarbonates, hydroxides, and, less significantly, borates, silicates, phosphates, and organic substances. Total alkalinity is measured by titrating a water sample with acid to the phenolphthalein end-point (about pH 5.3). A high alkalinity is generally associated with high pH, hardness, and excessive dissolved solids, and is indicative of high acid buffering capacity.

There are no water quality guidelines for alkalinity, although guidelines of >20 or >30 $mg \cdot L^{-1}$ have been suggested (McNeely et al. 1979). It has also been suggested that alkalinity should be maintained at natural background levels with no sudden variations, in order to protect the aquatic environment.

Peatland waters generally have low alkalinity levels. Clausen (1980) and Clausen and Brooks (1983a, 1983b) reported levels as high as 74 $mg \cdot L^{-1}$ for both natural and cultivated fens. Levels in both natural and mined bogs were low, in the order of 10 $mg \cdot L^{-1}$. It appears that alkalinity levels in waters downstream from peatlands may increase upon development of fens, and may decrease with bog developments.

7.3.4 Major Cations

Calcium, magnesium, sodium, and potassium are the most abundant elements in inland surface waters and, in relation to each other, are usually most abundant in decreasing order as listed above (Cole 1983). Levels of these elements in natural and developed peatland runoff waters are presented in Table 7.7.

Calcium is relatively abundant in the earth's crust and is readily soluble in water. It remains soluble in water as long as CO_2 is present; that is, at pH values less than 7 to 8. At more alkaline pH values Ca will precipitate as $CaCO_3$. Ca^{2+} is one of the major ions contributing to hardness, total dissolved solids, and specific conductivity (McNeely et al. 1979; Cole 1983).

Calcium guidelines for livestock watering (Table 7.3), as well as other uses, are given by CCREM (1987). Levels of Ca in natural peatland waters are generally quite low as compared to the CCREM guidelines (Table 7.7). Comparison of Ca levels between natural and developed peatlands shows that Ca levels may increase as a result of development, but even in these situations the levels do not reach those of the guidelines.

Magnesium is a constituent of most of the silicate rocks, dolomite and limestone. It occurs in water in ionic form and, like Ca, its solubility increases with greater CO_2 concentrations or lower pH. Mg is an essential nutrient, being important, for example, in chlorophyll of plants. It is nontoxic and does not pose a concern to aquatic life or public health. Concerns about usability of waters high in Mg are mainly in relation to its contribution to hardness and to its laxative effect (McNeely et al. 1979).

The levels of Mg in peatlands are generally less than 20 $mg \cdot L^{-1}$. Studies of Mg in waters from natural and developed peatlands show that levels can increase with development, particularly when used for agriculture (Table 7.7). Nevertheless,

Table 7.7. Major cations and specific conductance (SC) in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Ca $mg \cdot L^{-1}$	Mg $mg \cdot L^{-1}$	K $mg \cdot L^{-1}$	Na $mg \cdot L^{-1}$	SC $\mu S \cdot cm^{-1}$
Ahti 1984	bog	forestry							
Finland		fertilized	runoff	-	-	-	0.1	-	-
		unfertilized	runoff	-	-	-	0.1	-	-
		fertilized	runoff	-	-	-	0.2	-	-
		unfertilized	runoff	-	-	-	0.1	-	-
		fertilized	runoff	-	-	-	0.3	-	-
		unfertilized	runoff	-	-	-	0.2	-	-
Brown et al. 1980 Minnesota	shallow peat soil	wild rice	supply	-	-	-	1.2	-	-
			discharge	-	-	-	2.2	-	-
Clausen 1980	fen	natural	runoff	46	10.3	3.0	1.3	1.2	51
Minnesota	fen	natural	runoff	14	17.1	6.2	<0.8	1.6	105
	bog	mined	runoff	46	4.3	1.7	1.9	3.3	66
	fen	cultivated	runoff	12	35.0	9.8	1.8	3.0	202
Clausen and	fen	natural	runoff	54	27.3	7.9	1.2	2.7	171
Brooks 1983a.b	fen	natural	runoff	55	11.6	3.9	1.0	1.3	72
Minnesota	bog	natural	runoff	66	6.8	2.4	1.2	1.8	45
	bog	mined	ditch	5	5.1	2.0	0.9	3.0	67
Ferda and		natural	runff	10	2.7	-	0.8	19.8	202
Novak 1976 Czechoslovakia		drained-forestry	runoff	10	2.8	-	1.2	16.0	111
		drained-mineral	runoff	10	3.3	-	1.3	23.0	84
Heikuraimen et al.	bog	natural	-	-	-	-	-	-	32
1978 Finland	bog	drained-forestry	-	-	-	-	-	-	56
	swamp	natural	-	-	-	-	-	-	31
	swamp	drained-forestry	-	-	-	-	-	-	35

continued . . .

Table 7.7. Concluded.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Ca $mg \cdot L^{-1}$	Mg $mg \cdot L^{-1}$	K $mg \cdot L^{-1}$	Na $mg \cdot L^{-1}$	SC $\mu S \cdot cm^{-1}$
Kenttamies 1981	bog	natural	-	-	-	-	0.4	-	-
Finland	bog	drained, fertilized-forestry	-	-	-	-	0.7	-	-
Kenttamies and Laine 1984	bog	undrained (3y)	-	-	1.2-1.5		0.3-0.4	-	-
Finland	bog	drained-forestry (2y)	-	-	1.3-2.1		0.4-0.6	-	-
	bog	fertilized-forestry (2y)	-	-	1.9		0.9	-	-
Largin 1976	bog	natural	pit	-	1.2-8	1.2-2.8	-	-	-
RSFSR	bog	cultivated	ditch	-	8-30	0.5-6.0	-	-	-
	fen	natural	pit	-	12-100	0.5-10.0	-	-	-
	fen	cultivated	ditch	-	50-88	9.0-18.0	-	-	-
Miller 1979	marsh	farmed,	tile	23	199	41	74	-	730
Ontario	(Holland)	fertilized	drainage	22	165	22	33	-	550
	Marsh)	3 sites		39	193	28	16	-	515
Osborne, in	bog	mined	upstream	13-15	5.0	1.7	0.1	-	37
Washburn and			discharge	from	0.8	0.3	0.1	-	26
Gillis Associates			downstream	July	3.8	1.2	0.2	-	30
Ltd. 1983				- Nov					
Newfoundland									
Washburn and Gillis Associates Ltd. 1983									
Newfoundland	bog	drained	pit	2	-	-	0.6	-	-
Ontario	bog	drained	pit	2	-	-	0.2-0.5	-	-
British Columbia	bog	mined	pit	2	-	-	0.4-0.5	-	-
			recycle pond	1	-	-	0.8	-	-

the levels are generally low and, therefore, should not have adverse effects on water quality.

Sodium is abundant in igneous and sedimentary rocks, as well as in evaporite sediments. It is very reactive and soluble and, when leached from rocks and soils, its compounds tend to remain in solution. It is commonly the third most abundant element in lakes and streams, and in some situations can be the predominant cation. In the UK, sodium in peatland waters has been found to be related to atmospheric sources from sea spray and has been found in concentrations as high as 63 $mg \cdot L^{-1}$ (Gorham 1956; Gorham and Cragg 1960). Levels are much lower in continental peatlands, and are generally lower in bogs than in fens (Chapter 6).

The Alberta Environment (1977) surface water quality objectives indicate that Na should account for 30 to 75% of the total cations in water. This guideline is applied to suitability of water for irrigation as are Environment Canada guidelines for sodium adsorption ratio (CCREM 1987). However, levels and proportions in peatlands are much lower than the guidelines (Table 7.7). Levels of Na in developed peatlands may be marginally higher than in natural peatlands. The overall effect of peatland runoff is that for both natural and developed peatlands, downstream waters would usually become diluted with respect to Na levels.

Potassium is usually closely associated with Na, and is commonly the fourth ranking cation in lake waters. Both Na and K are alkali metals with similar chemical characteristics, except that K may be more readily adsorbed by colloids and can be incorporated into new stable minerals such as clay minerals. Potassium is an essential element for plant and animal nutrition, and moderate quantities have no adverse effects. Very high concentrations can be harmful to humans, but no Alberta or federal guidelines have been established. The levels in peatland waters are very low, even in developed peatlands, and dilution would be the net effect on many downstream waters. Levels of K in are often higher in developed than in natural peatlands, but they are nevertheless very low (Table 7.7).

7.3.5 Specific Conductance, Total Dissolved Solids, and Hardness

Specific conductance, or conductivity, is a numerical expression of a water's ability to conduct an electrical current. The conductivity of water is related to its ionic concentrations and to temperature. It is measured in microsiemens per centimetre ($\mu S \cdot cm^{-1}$), or in siemens per metre ($S \cdot m^{-1}$) in SI units, and is corrected to a standard temperature, usually 25°C. Conductivities of peatland waters are usually corrected and reduced to compensate for the high hydrogen ion content. When this is done, specific conductance (SC) correlates strongly with the total of the predominant cations in solution. For example, Turchenek et al. (1984) found a correlation of $r^2 = 0.96$ between corrected conductance and the sum of Ca and Mg in peat waters from the Athabasca area, Alberta (see Table 6.4, Chapter 6).

No provincial or federal guidelines have been established for specific conductance. McNeely et al. (1979) noted that high levels are found to correlate with total dissolved solids which have defined guidelines for some uses. Specific conductance is nevertheless a commonly determined property in peatland waters. Levels as high as 200 $\mu S \cdot cm^{-1}$ have been recorded (Table 7.7). The data from various studies, though inconclusive, seem to indicate that development of fens increases the SC of runoff waters, whereas bog development results in decreases of SC in runoff waters. Bogs are low in dissolved cations other than H^+, and their runoff contributes somewhat less to the EC of downstream waters than does fen runoff.

CCREM (1987) guidelines have been established for total dissolved solids and hardness in waters used for domestic and livestock consumption (Tables 7.2 and 7.3). Values for these properties are not generally reported in the literature. However, conclusions with regard to effects of peatland development on these properties would be the same as those for the major cations and for specific conductance due to their high intercorrelations.

7.4 MAJOR NUTRIENTS

7.4.1 Sulphur

The sulphate form of S is usually second to carbonate in abundance in fresh waters, although chloride occasionally surpasses it. Sulphur sources include gypsum and many of the igneous and sedimentary rocks. Weathering and oxidation of these materials in well aerated water yields sulphate (SO_4^{2-}) ions which are carried off in waters. Precipitation can be another source of SO_4^{2-}, particularly in more populated and industrial areas. Under reducing conditions, sulphide-S forms may be found in waters. The principal species present below pH 5 is H_2S. HS^- and H_2S concentrations are approximately equal at pH 7, and HS^- predominates above pH 7. Sulphate reduction to sulphide can occur under severe reducing conditions if certain S-bacteria and organic matter are present. Organic-S in peat can be microbially converted to SO_4^{2-} under oxidizing conditions and to sulphide under reducing conditions. A more thorough discussion of sulphur can be found in McNeely et al. (1979), Cole (1983), (CCREM 1987) and several other references.

Sulphate concentrations are rarely high enough to affect aquatic life adversely. CCREM water quality guidelines with respect to domestic consumption and to livestock (Tables 7.2 and 7.3) have been established because of effects on taste of water and on the gastro-intestinal system. Hydrogen sulphide is highly poisonous and the guidelines are consequently at very low values.

Most peatland waters are low in SO_4^{2-}. Levels in bogs are lower than in fens, but SO_4^{2-} accounts for a higher proportion of the inorganic ions in bogs as compared to fens. Few sulphate data and no sulphide data were found in reports comparing developed and undeveloped peatlands. Largin's (1976) data from the USSR tend to show that SO_4^{2-} increases in runoff waters of cultivated bogs and fens as compared to natural peatlands (Table 7.8). In peatlands drained for forestry in Czechoslovakia, Ferda and Novak (1976) found little change as compared to undeveloped peatlands (Table 7.8).

Table 7.8. Sulphate in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	$SO_4^{=}$-S $mg \cdot L^{-1}$
Ferda and Novak 1976 Czechoslovakia		natural	runoff	10	20
		drained-forestry	runoff	10	16
		drained-mineral	runoff	10	23
Largin 1976 RSFSR	bog	natural	pit	-	2-8
	bog	cultivated	ditch	-	2-24
	fen	natural	pit	-	3-16
	fen	cultivated	ditch	-	8-80
Miller 1979 Ontario	marsh (Holland Marsh)	farmed, fertilized 3 sites	tile drainage	23	30
				22	43
				39	50

Data for sulphide levels in peatland runoff waters have not been found in the literature. The main impact of sulphides on water chemistry occurs through their effects on acidification. Oxidation of H_2S, FeS_2, and other sulphide and organic S forms result in generation of acidity. If H_2S is present in anaerobic waters, phototrophic and chemotrophic bacteria can oxidize it to elemental sulphur and later to H_2SO_4. Oxidation can occur whenever the water table in a peatland is lowered or when runoff mixes with more highly oxygenated waters. Depression in pH of runoff waters discussed in Section 7.3.3 can thus occur as a result of oxidation of reduced S forms as well as of input of acidic humic materials. Further discussion of sulphur in peatlands, particularly in relation to acidification, is given in Chapter 9.

7.4.2 Nitrogen

Nitrogen is a major nutrient which has a complex biogeochemical cycle and varied mode of origin. The sources of N include weathering of rock, electrical discharge and photochemical formation in the atmosphere, stores in soils, and various biological forms. Forms of N include ammonia (NH_3 and NH_4^+), nitrate (NO_3^-), nitrite (NO_2^-), and numerous organic forms. Atmospheric nitrogen gas (N_2) is converted to organic N by N-fixing plants and microorganisms. Decomposition of plant material results in NH_3 formation which in turn can be nitrified by aerobic microorganisms. In the absence of oxygen, NO_3 can be reduced back to NH_3 or, through denitrification, to N_2 gas. This may be a very important process in many wetlands. Shallow waters with fluctuations in O_2 status and organic substrates are likely to be sites of high denitrification, with N_2 production rates approaching 2 to 4 $mg \cdot L^{-1} \cdot d^{-1}$. As with sulphur, most of the N in peatlands is present in reduced forms, as NH_3 and predominantly in organic forms. All of the N forms can be present in surface waters in dissolved as well as in solid particulate forms.

Criteria for NO_3-N, NO_2-N, and NH_3-N are provided for various use categories by CCREM (1987) (Tables 7.1, 7.2, and 7.3).

Only total N was included in the Alberta Environment (1977) surface water quality objectives. McNeely et al. (1979) discussed the adverse effects of excessive amounts of NH_3/NH_4^+, NO_3^-, or NO_2^- on health of humans, livestock, wildlife, and fish.

Several comparative studies of runoff water quality of developed and undeveloped peatlands have included analyses of various N forms (Table 7.9). In different studies carried out in Minnesota, total N levels exceeded 1 $mg{\cdot}L^{-1}$ in both natural and developed peatlands ($\leq$1 $mg{\cdot}L^{-1}$ being the Alberta surface water quality objectives). The N appeared to be predominantly organically bound and little appeared to be in NO_3^- and NO_2^- forms. However, NH_4-N reached excessive levels in some cases. Nitrite was shown to reach excessive levels in drainage ditch water of cultivated fens in the USSR (Largin 1976, Table 7.9). Levels of NO_3-N and total N in studies of drained peatland and bog mine discharge in Newfoundland, Ontario, and British Columbia would exceed Alberta guidelines for total N (Table 7.9).

Levels of NO_3 in drainage waters of farmed peat soils in southern Ontario have been found to be very high (Table 7.9). Duxbury and Peverly (1978) and Miller (1979) collected samples directly from tile drainage; the NO_3-N levels thus obtained approximate soil solution levels. NO_3-N levels reported by Nichólls and MacCrimmon (1974) for marshes in southern Ontario are considerably lower than those of Miller (1979) for similar soils in the same region. Nevertheless, the results showed that the combined effects of fertilization, drainage, and hence, oxidizing and nitrifying conditions, yielded about 40 to 50 times more NO_3-N in runoff water from cultivated marsh areas than from uncultivated marsh areas. Miller (1979) showed that the total NO_3-N in farmed peatland outflow was very high, being in some seasons more than double the amount of fertilizer N added. This was probably due to N released by decomposition of the organic soils. Duxbury and Peverly (1978) similarly concluded that much of the NO_3-N source was native mineralized N. They also concluded that organic soils can contribute

Table 7.9. Nitrogen in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Total N $mg \cdot L^{-1}$	NO_3-N+ NO_2-N $mg \cdot L^{-1}$	NO_2-N $mg \cdot L^{-1}$	NO_3-N $mg \cdot L^{-1}$	NH_4-N $mg \cdot L^{-1}$	Organic N $mg \cdot L^{-1}$
Brown et al. 1980	shallow	wild rice	supply	-	1.4	<1.0	-	-	-	-
Minnesota	peat soil		discharge	-	2.2	<1.0	-	-	-	-
Clausen 1980	fen	natural	runoff	46	1.5	<0.1	<0.01	-	0.2	1.2
Minnesota	fen	natural	runoff	14	1.2	<0.1	<0.01	-	<0.1	1.1
	bog	mined	runoff	46	3.6	0.1	0.01	-	2.2	1.7
	fen	cultivated	runoff	12	6.3	0.2	0.01	-	1.1	5.2
Clausen and	fen	natural	runoff	54	1.8	-	-	-	0.2	1.6
Brooks 1983a,b	fen	natural	runoff	55	1.6	-	-	-	0.2	1.4
Minnesota	bog	natural	runoff	66	1.5	-	-	0.06	0.1	1.4
	bog	mined	ditch	5	3.7	-	-	0.05	1.8	1.9
Duxbury and Peverly 1978 New York	swamp	farmed, fertilized	tile drainage	11	-	-	-	10.3-33.5	-	-
Ferda and		natural	runoff	10	-	0	0	-	0.4	-
Novak 1976		drained-forestry	runoff	10	-	0	0	-	0.3	-
Czechoslovakia		drained-mineral	runoff	10	-	0	0	-	0.1	-
Heikurainen et al.	bog	natural	-	-	0.5	-	-	-	-	-
1978	bog	drained-								
Finland		forestry	-	-	0.6	-	-	-	-	-
	swamp	natural	-	-	0.4	-	-	-	-	-
	swamp	drained-forestry	-	-	0.5	-	-	-	-	-
Kenttamies and	bog	undrained (3y)	-	-	0.4-0.5	-	-	-	2.0-8.9	-
Laine 1984										
Finland	bog	drained-forestry (2y)	-	-	0.4-0.8	-	-	-	7.5-92.8	-
	bog	fertilized-forestry (2y)	-	-	0.7	-	-	-	133.2	-

continued . . .

Table 7.9. Concluded.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Total N $mg \cdot L^{-1}$	NO_3-N+ NO_2-N $mg \cdot L^{-1}$	NO_2-N $mg \cdot L^{-1}$	NO_3-N $mg \cdot L^{-1}$	NH_4-N $mg \cdot L^{-1}$	Organic N $mg \cdot L^{-1}$
Largin 1976	bog	natural	pit	-	-	-	0.1	0.1-1.0	-	-
RSFSR	bog	cultivated	ditch	-	-	-	-	0.1-0.5	-	-
	fen	natural	pit	-	-	-	0.1	0.1	-	-
	fen	cultivated	ditch	-	-	-	0.5-1.5	0.5-2.0	-	-
Miller 1979	marsh	farmed,	tile	23	-	-	-	34.5	-	-
Ontario		fertilized	drainage	22	-	-	-	33.8	-	-
		3 sites		39	-	-	-	38.0	-	-
Nicholls and	marsh	farmed,	piezometer	8	-	-	0.37	0.685	0.81	-
MacCrimmon	(Holland	fertilized	well	March	-	-				
1974	marsh)			- Sept	-	-	0.001	<0.07	0.09	-
Ontario		undeveloped		1971						
Osborne, in	bog	mined	upstream	13-15	-	-	-	0.8	-	-
Washburn and			discharge	July	-	-	-	1.0	-	-
Gillis Associates			downstream	- Nov	-	-	-	1.1	-	-
Ltd. 1983				1982						
Newfoundland										
Washburn and										
Gillis Associates										
Ltd. 1983										
Newfoundland	bog	drained	pit	2	0.9-7.5	-	-	-	-	-
Ontario	bog	drained	pit	2	1.1-2.7	-	-	-	-	-
British Columbia	bog	mined	pit	2	3.7-5.5	-	-	-	-	-
			recycle pond	1	38.1	-	-	-	-	-

N and P in land runoff in much greater proportions than indicated in their study area.

7.4.3 Phosphorus

Phosphorus is an essential plant nutrient which is somewhat more scarce than the other major components of organisms (i.e., C, H, O, N, and S). It is taken up rapidly and concentrated by living organisms. Sources of P to water bodies and wetlands include weathering of igneous rocks, soil leaching, organic wastes, bottom sediments, and decomposition of organic matter. The P cycle is complicated and poorly understood. Phosphorus exists in numerous organic and inorganic forms, and can be present in water as dissolved or particulate species. In the pH range of most peatland waters (5 to 6), $H_2PO_4^-$ is the most common form of P. At higher pH values, HPO_4^{2-} occurs in higher concentrations. Phosphates are strongly sorbed by carbonates and by oxyhydroxides of Al and Fe.

Excessive P levels in water contribute to eutrophication. When sufficient N compounds are present, P concentrations above 0.1 $mg \cdot L^{-1}$ may promote algal and other growth in water bodies. Environment Canada had provided guidelines for P concentrations in surface water based on prevention of accelerated eutrophication as well as on other use factors (McNeely et al. 1979). The Alberta Environment (1977) surface water quality objectives indicate that the total organic and inorganic P should be less than 0.15 $mg \cdot L^{-1}$. CCREM (1987) guidelines for phosphorus have not been provided for aquatic life, or domestic and livestock use.

In studies of peatland runoff waters, total P and occasionally PO_4-P levels are reported (Table 7.10). Total P levels exceeding 0.15 $mg \cdot L^{-1}$ were found in only a few of these studies. The excessive total P levels were found only in developed peatlands, and there is evidence that particularly high levels can be reached where peatlands are fertilized. Very high levels of PO_4-P have been found in farmed marsh and swamp tile drainage waters in Ontario and New York (Duxbury and Peverly 1978; Miller 1979). Levels found by

Table 7.10. Phosphorus in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Total P $mg \cdot L^{-1}$	PO_4-P $mg \cdot L^{-1}$
Ahti 1984 Finland	bog	forestry				
		fertilized	runoff	-	0.09	-
		unfertilized	runoff	-	0.02	-
		fertilized	runoff	-	0.10	-
		unfertilized	runoff	-	0.01	-
		fertilized	runoff	-	0.11	-
		unfertilized	runoff	-	0.02	-
Brown et al. 1980 Minnesota	shallow peat soil	wild rice	supply	-	-	0.17
			discharge	-	-	0.45
Clausen 1980 Minnesota	fen	natural	runoff	46	0.15	-
	fen	natural	runoff	14	0.04	-
	bog	mined	runoff	46	0.11	-
	fen	cultivated	runoff	12	0.64	-
Clausen and Brooks 1983a,b Minnesota	fen	natural	runoff	54	0.08	-
	fen	natural	runoff	55	0.05	-
	bog	natural	runoff	66	0.06	-
	bog	mined	runoff	5	0.09	-
Duxbury and Peverly 1978 New York	swamp	farmed, fertilized	tile drainage	11	-	0.2-7.8

continued . . .

Table 7.10. Continued.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Total P $mg \cdot L^{-1}$	PO_4-P $mg \cdot L^{-1}$
Ferda and Novak 1976 Czechoslovakia	-	natural	runoff	10	-	0.18
		drained-forestry	runoff	10	-	0.04
		drained-mineral	runoff	10	-	0.03
Heikurainen et al. 1978 Finland	bog	natural	-	-	0.03	-
	bog	drained-forestry	-	-	0.03	-
	swamp	natural	-	-	0.02	-
	swamp	drained-forestry	-	-	0.10	-
Kenttamies 1981 Finland	bog	natural	-	-	0.21	-
	bog	drained, fertilized-forestry	-	-	0.75	-
Kenttamies and Laine 1984 Finland	bog	undrained (3y)	-	-	0.01-0.02	0.004
	bog	drained-(2y) forestry	-	-	0.02-0.02	0.004-0.005
	bog	fertilized (2y) forestry	-	-	0.03-0.04	0.02

continued . . .

Table 7.10. Concluded.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Total P $mg \cdot L^{-1}$	PO_4-P $mg \cdot L^{-1}$
Miller 1979	marsh	farmed,	tile	23	-	13.5
Ontario		fertilized	drainage	22	-	5.4
		3 sites		39	-	4.0
Nicholls and	marsh	farmed,	piezometer	7-8	-	0.03
MacCrimmon	(Holland	fertilized	well	May -		
1974	Marsh)	undeveloped	piezometer	Sept		
Ontario			well	1971	-	0.02
Sallantaus and	bog	natural	weir	several	0.03-0.04	-
Patila 1983	bog	mined	weir	April	0.09-0.20	-
Finland	bog	mined	weir	- Oct	0.04-0.04	-
	bog	mined	weir		0.03-0.04	-
	bog	mined	weir		0.02-0.02	-
Osborne, in	bog	mined	upstream	13-15	<0.005	-
Washburn and			discharge	July	<0.005	-
Gillis Associates			downstream	- Nov	<0.005	-
Ltd. 1983				1982		
Newfoundland						
Washburn and						
Gillis Associates						
Ltd. 1983						
Newfoundland	bog	drained	pit	2	<0.05	-
Ontario	bog	drained	pit	2	<0.05	-
British Columbia	bog	mined	pit	2	0.10-0.25	-
			recycle			
			pond	1	1.0	

Nicholls and MacCrimmon (1974) in another Ontario study were much lower than those reported by Miller (1979), but they nevertheless concluded that farmed and fertilized peatlands contributed four to five times more P than uncultivated marsh soils. These researchers also noted that most of the P lost in drainage water is released during a five to six week pumping period during the spring and that more than 90% of the total P in runoff was in soluble reactive form. It was, therefore, readily available for algal and aquatic plant growth in downstream waters.

7.5 MINOR AND TRACE ELEMENTS

A large number of elements that occur in wetlands and surface waters in small quantities are discussed in this section. Some are significant in that they are nutrients essential in minute quantities to animal or plant life, or both. Others are important in that they may be toxic to living organisms. Still others are neither nutrients nor are they harmful to life, but they are significant from a geochemical viewpoint only.

Properties of these elements are only briefly discussed in the following sections, while levels in peatland waters and in comparative studies of natural and undeveloped peatlands are emphasized. The elements listed by McNeely et al. (1979) and CCREM (1987) are included in the discussions, but they have been grouped based on similarities according to the periodic table. The above publications should be referred to for discussions of the properties and significance of these elements. CCREM guidelines are presented in Tables 7.1, 7.2, and 7.3.

7.5.1 Aluminum

Aluminum in terrestrial systems has received much attention because of its toxicity to plants and its availability in acidic soils. It has been implicated in dieback of forests in Germany where deposition of acidic and acidifying substances has been high for a number of years (Matzner and Ulrich 1985). Aluminum concentration in waters is generally low, because it readily precipitates and settles out. Nevertheless, Al present in low concentrations can be toxic to fish, and guidelines for protection of aquatic life are low (Table 7.1). The toxicity of Al is also dependent on its form and on pH. The effect of pH on Al speciation and toxicity to fish has been reviewed by Ramamoorthy (1987). Complexing of Al by humic materials has been found to diminish toxity.

Levels of total Al in peatland runoff waters commonly exceed the guideline for protection of aquatic life (Table 7.11). The levels are well within guidelines for livestock, wildlife, and domestic consumption. Comparisons among different peatland types and uses are difficult to make. Higher levels of Al appear to be associated with bog waters (which are more acidic), but not necessarily with developed as compared to natural peatlands.

7.5.2 Boron

Boron occurs in waters mainly as the borate (BO_3^{3-}) anion. Weathering of rocks, soil leaching, and groundwater discharge are sources of B to water bodies and wetlands. The CCREM guideline for livestock and domestic use is $<5.0 \text{ mg} \cdot L^{-1}$. Boron data from peatland studies in Minnesota, Newfoundland, Ontario, and British Columbia indicate that levels are generally much lower than the guideline (Table 7.12).

7.5.3 Silicon

Silicon is second only to oxygen in its abundance in the lithosphere. It is commonly present in surface waters in very low quantities in colloidal form. Silica in water is not harmful to organisms and no water quality guidelines have been established. Levels of silica in peatland waters have not been reported but predicted levels would be very low.

Table 7.11. Aluminum in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Al $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	0.26
Minnesota	fen	natural	runoff	14	0.08
	bog	mined	runoff	46	0.73
	fen	cultivated	runoff	12	0.38
Clausen and	fen	natural	runoff	59	0.25
Brooks 1983 a, b	fen	natural	runoff	55	0.25
	bog	natural	runoff	66	0.55
	bog	mined	runoff	5	0.41
Osborne, in Washburn and Gillis Associates Ltd. 1982 Newfoundland	bog	mined	upstream	9-15	0.05
			discharge	9-15	0.06
			downstream	9-15	0.08
Washburn and Gillis Associates Ltd. 1983					
Newfoundland	bog	drained	pit	2	0.09-0.39
Ontario	bog	drained	pit	2	0.32-0.51
British Columbia	bog	mined	pit	2	0.27-.32
			recycle pond	1	4.72

Table 7.12. Boron in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	B $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	0.01
Minnesota	fen	natural	runoff	14	0.01
	bog	mined	runoff	46	0.02
	fen	cultivated	runoff	12	0.04
Washburn and Gillis Associates Ltd. 1983					
Newfoundland	bog	drained	pit	2	<0.05-0.08
Ontario	bog	drained	pit	2	<0.05
British Columbia	bog	mined	pit	2	<0.05
			recycle pond	1	<0.001

7.5.4 Titanium and Vanadium

Titanium and vanadium are transition elements which exhibit several valence states. Both occur in water, mainly as colloidal oxyhydroxy or silicate forms. Vanadium levels are of concern because of toxicity if present in the form of a pentavalent salt (McNeely et al. 1979). Although, a guideline of $<0.1\ mg \cdot L^{-1}$ for protection of livestock and wildlife has been recommended by Environment Canada (McNeely et al. 1979), there are no CCREM guidelines for this element. Levels reported for drained and mined bogs in Canada are considerably lower than this level (Table 7.13).

7.5.5 Chromium, Molybdenum, and Tungsten

These elements constitute Group 6b of the transition elements in the periodic table. Small amounts of Cr are essential to mammals, while larger quantities, in the Cr^{6+} form, may be harmful. Mo is an essential nutrient for plants as well, and larger quantities can be harmful to livestock. It occurs in water primarily as the weakly soluble anion MoO_4^{2-}. Tungsten (W) is very insoluble in water, and no water quality guidelines have been established for it. The CCREM guideline for Cr is a limit of $1.0\ mg \cdot L^{-1}$ for livestock water, $0.05\ mg \cdot L^{-1}$ for domestic water supplies, $0.02\ mg \cdot L^{-1}$ for fish, and $2.0\ mg \cdot L^{-1}$ for other aquatic life. An Mo limit of $0.5\ mg \cdot L^{-1}$ is the guideline for livestock (CCREM 1987). Levels in some developed and undeveloped peatlands are presented in Table 7.14. The data from Minnesota, as well various areas in Canada, show that levels in any of the peatland runoff waters do not approach the recommended maximum levels.

7.5.6 Manganese

Manganese is a transition element (Group 7b) which is similar to iron in behaviour and is active in redox reactions. It is an essential element for plants and animals. Guidelines are based on domestic consumption, and levels for animal and plant use have not been established as Mn does not reach toxic levels in natural waters.

Table 7.13. Titanium and vanadium in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Ti $mg \cdot L^{-1}$	V $mg \cdot L^{-1}$
Washburn and Gillis Associates Ltd. 1983						
Newfoundland	bog	drained	pit	2	0.01-0.02	<0.002
Ontario	bog	drained	pit	2	0.02-0.03	<0.002
British Columbia	bog	mined	pit	2	0.01	<0.002
			recycle pond	1	0.04	0.005

Table 7.14. Chromium and molybdenum in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Cr $mg \cdot L^{-1}$	Mo $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	<0.01	
Minnesota	fen	natural	runoff	14	<0.01	
	bog	mined	runoff	46	0.02	
	fen	cultivated	runoff	12	<0.01	
Osborne, in	bog	mined	upstream	9-15		<0.01
Washburn and			discharge	9-15		<0.01
Gillis Associates			downstream	9-15		<0.01
Ltd. 1983						
Newfoundland						
Washburn and						
Gillis Associates						
Ltd. 1983						
Newfoundland	bog	drained	pit	2	0.001	<0.005
Ontario	bog	drained	pit	2	<0.001	0.005
British Columbia	bog	mined	pit	2	0.001	<0.005
			recycle pond	1	0.005	<0.005

The CCREM guideline is ≤0.05 $mg \cdot L^{-1}$. Levels of Mn in peatland runoff waters reported in Minnesota and Czechoslovakia substantially exceeded this value (Table 7.15). However, data for Canadian peatland waters do not appear to reach these levels (see Chapter 8, Section 8.10 regarding levels reported in Alberta).

7.5.7 Iron, Cobalt, and Nickel

These are transition elements in Group 8b of the periodic table. Iron is active in redox reactions and can be found in surface waters in both ferric (Fe^{3+}) and ferrous (Fe^{2+}) forms. Iron is an essential plant nutrient but can be toxic at high levels. Aquatic organisms may be deleteriously affected at low levels. Cobalt is an essential trace nutrient in some plants and animals, but can also be toxic in elevated concentrations. Similarly Ni can be toxic to some plants and aquatic organisms. CCREM guidelines for Fe are provided for domestic livestock and aquatic life categories, Co for livestock only, and Ni for livestock and aquatic categories. Some data for peatland waters are presented in Table 7.16. Most of the values for Fe in peatland runoff waters would considerably exceed the water quality guidelines. Levels of Co and Ni are well within the guidelines. High values of Fe were noted especially in runoff waters of mined and cultivated peatlands in Minnesota. However, there has been no clear indication as to whether fen or bog waters are higher in Fe.

7.5.8 Copper and Silver

These elements constitute a part of Group 1b of the transition elements in the periodic table. Copper is an essential nutrient to all plants and animals. Its solubility is relatively high in acidic waters, and it precipitates as a hydroxide or carbonate in alkaline waters. At elevated concentrations, Cu can be toxic. Guidelines for various use categories are presented in Tables 7.1, 7.2, and 7.3 for aquatic, domestic and livestock categories. Silver is not essential to plants and animals but it can have some

Table 7.15. Manganese in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Mn $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	1.0
Minnesota	fen	natural	runoff	14	0.1
	bog	mined	runoff	46	0.2
	fen	cultivated	runoff	12	0.1
Clausen and	fen	natural	runoff	54	0.1
Brook 1983a,b	fen	natrual	runoff	55	0.3
	bog	natrual	runoff	66	0.3
	bog	mined	runoff	5	0.2
Ferda and	-	natural	runoff	10	0.1
Novak 1976 Czechoslovakia		drained-forestry	runoff	10	0.1
		drained-mineral	runoff	10	0.2
Osborne, in	bog	mined	upstream	9-15	0.03
Washburn and			discharge	9-15	0.06
Gillis Associates			downstream	9-15	0.08
Ltd. 1983					
Newfoundland					
Washburn and Gillis Associates Ltd. 1983					
Newfoundland	bog	drained	pit	2	0.002-0.008
Ontario	bog	drained	pit	2	0.01-0.98
British Columbia	bog	mined	pit	2	0.03-0.04
			recycle pond	1	0.04

Table 7.16. Iron, cobalt, and nickel in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Fe $mg \cdot L^{-1}$	Co $mg \cdot L^{-1}$	Ni $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	3.0	-	<0.04
Minnesota	fen	natural	runoff	14	0.4	-	<0.04
	bog	mined	runoff	46	4.0	-	<0.04
	fen	cultivated	runoff	12	8.2	-	<0.04
Clausen and	fen	natural	runoff	54	2.4	-	-
Brook 1983a,b	fen	natural	runoff	55	2.1	-	-
	bog	natural	runoff	66	2.6	-	-
	bog	mined	runoff	5	4.3	-	-
Ferda and	-	natural	runoff	10	4.9	-	-
Novak 1976		drained-					
Czechoslovakia		forestry	runoff	10	4.3	-	-
		drained-					
		mineral	runoff	10	2.5	-	-
Kenttamies and	bog	undrained					
Laine 1984		(3y)	-	-	0.5-0.8	-	-
Finland	bog	drained					
		(2y)-					
		forestry	-	-	0.7-1.6	-	-
	bog	fertilized					
		(2y)-					
		forestry	-	-	1.4	-	-
Osborne, in	bog	mined	upstream	9-15	0.4	<0.01	-
Washburn and			discharge	9-15	0.8	<0.01	-
Gillis Associates			downstream	9-15	1.8	<0.01	-
Ltd. 1983							
Newfoundland							
Washburn and							
Gillis Associates							
Ltd. 1983							
Newfoundland	bog	drained	pit	2	0.1-0.2	<0.001	<0.004
Ontario	bog	drained	pit	2	0.6-1.0	<0.001	<0.004
British Columbia	bog	mined	pit	2	0.2-0.4	<0.001	<0.004
			recycle				
			pond	1	2.0	<0.001	<0.004

deleterious effects. The water quality guidelines are established mainly on the basis of possible human effects. Levels of Cu in peatland waters (Table 7.17) are well below the Alberta guidelines. Data for Ag are not available.

7.5.9 Zinc, Cadmium, and Mercury

These elements are members of Group 2b in the periodic table. Zinc is an essential element for plants, animals, and humans and has deleterious effects only at high concentrations. However, it can be very toxic to aquatic life, particularly fish. Cadmium and mercury are highly toxic to man, animals, and aquatic life. Guidelines are presented in Tables 7.1, 7.2, and 7.3, and data for peatland waters are given in Table 7.18. Levels of Cd and Zn reported in studies from Minnesota and some Canadian provinces should pose no threat to water quality. Comparatively high values have been found for Hg, however.

Further review of increased discharge of Hg related to peatland drainage is presented here because of its potential to affect water quality. Mercury can occur in various forms including elemental Hg, inorganic Hg compounds, short-chain alkyl mercurials (methyl and ethyl), and other organomercury compounds. Mercury ions are found as Hg^0, Hg^+, and Hg^{2+}, and are readily interconnected in the environment (Moore and Ramamoorthy 1984). A significant proportion (~50%) of global Hg cycling is anthropogenic in origin, major sources being mining, smelting, and other industrial activities. In Alberta, Hg is widely present due to past uses of Hg-containing fungicides and other pesticides (Moore et al. 1986). The various interactions of Hg and its compounds in aquatic systems and the influence on biota have been reviewed by Moore and Ramamoorthy (1984).

Evidence for Hg discharge from drained peatlands was found by Lodenius (1983) who noted increased Hg levels in fish, especially pike, in lakes receiving peatland drainage. The increased Hg levels were considered to be related to several environmental factors

Table 7.17. Copper in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Cu mg·L^{-1}
Clausen 1980	fen	natural	runoff	46	<0.01
Minnesota	fen	natural	runoff	14	<0.01
	bog	mined	runoff	46	0.01
	fen	cultivated	runoff	12	0.01
Osborne, in Washburn and Gillis Associates Ltd. 1983 Newfoundland	bog	mined	upstream	9-15	<0.01
			discharge	9-15	<0.01
			downstream	9-15	<0.01
Washburn and Gillis Associates Ltd. 1983					
Newfoundland	bog	drained	pit	2	<0.02
Ontario	bog	drained	pit	2	<0.02
British Columbia	bog	mined	pit	2	<0.02
			recycle pond	1	<0.02

Table 7.18. Zinc, cadmium, and mercury in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Zn $mg \cdot L^{-1}$	Cd $mg \cdot L^{-1}$	Hg $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	-	0.01	2.6
Minnesota	fen	natural	runoff	14	-	<0.01	2.4
	bog	mined	runoff	46	-	<0.01	2.9
	fen	cultivated	runoff	12	-	<0.01	3.7
Clausen and	fen	natural	runoff	54		-	5.0
Brooks 1983a,b	fen	natural	runoff	55	-	-	3.0
	bog	natural	runoff	66	-	-	6.0
	bog	mined	runoff	5	-	-	4.0
Osborne, in	bog	mined	upstream	9-15	-	<0.01	0.7
Washburn and			discharge	9-15	-	<0.01	0.05-0.4
Gillis Associates			downstream	9-15	-	<0.01	<0.05-0.1
Ltd. 1983							
Newfoundland							
Washburn and							
Gillis Associates							
Ltd. 1983							
Newfoundland	bog	drained	pit	2	<0.004	<0.001	<1.0
Ontario	bog	drained	pit	2	0.01-0.02	<0.001	<1.0
British Columbia	bog	mined	pit	2	0.06-0.09	<0.001	<1.0
			recycle pond	1	0.02	<0.001	<1.0

including low pH, high humic matter content, low temperature, and low primary production. Lodenius et al. (1983) further found, through use of peat lysimeters, that transport of Hg from organic soils was facilitated by humic materials. However, increased acidity promoted Hg binding to peat and resulted in increased enrichment. Thus, acid deposition would theoretically reduce the leachability of Hg. On the other hand, Jackson et al. (1980) found that although Hg was deposited in lake bottoms as humic complexes, the assimilation into bottom sediment was inhibited by increased acidification. Jackson et al. (1980) speculated that enhanced production of water-soluble CH_3Hg^+ ions is another possible reason for lowered efficiency of Hg accumulation by sediments under acidified conditions. This is of concern, as methyl mercury is one of the most toxic of Hg compounds, and is considerably magnified in the aquatic food chain. Toxicities of Hg compounds have been reviewed by Moore and Ramamoorthy (1984). Much remains to be learned with regard to Hg in peatland drainage waters, since some of the research to date has been contradictory (Glooschenko et al. 1985).

Increases of Hg levels in lakes in Finland do appear related to peatland drainage as opposed to increased deposition from other sources. Simola and Lodenius (1982) showed that Hg in lake bottom sediments began to increase in concentration during the period 1942 to 1952. The Hg flux increased two times in the 1960s and about five times during the period 1967 to 1977. Since no agricultural or industrial sources were apparent, increased peatland drainage was considered to be the cause of the Hg deposition.

7.5.10 Lead and Tin

Lead is a metallic element of Group 4a in the periodic table. Tin, also discussed by McNeely et al. (1979), is in the same group. Lead is found in both soluble and suspended forms in waters. It is toxic to humans, animals, and fish, but its toxicity to plants is low. Tin has no function or toxicity to organisms and no water quality guidelines exist for it. However, alkyl tins have been reported to be toxic at very low levels (Loughlin and Fench 1980). The guidelines for Pb are given in Tables 7.1, 7.2, and 7.3. Levels of Pb in peatland runoff waters in Table 7.19 are below detection limits. Comparison with the very low guideline levels for aquatic life (Table 7.1) is, therefore, not possible.

7.5.11 Arsenic and Antimony

Along with N and P, As and Sb are members of the 5a group of elements in the periodic table. Arsenic is present in surface waters mainly in the form of the weakly soluble anion AsO_4^{3-} and may occur as arsenite AsO_3^{3-}. Arsenates are strongly sorbed onto inorganic particulates, but they may also form highly toxic organic compounds. Arsenic is toxic to all life forms to various extents, and water quality guidelines are, therefore, very low. Antimony occurs in very low quantities in natural waters, as it is readily sorbed to particulate material or precipitated as an oxide from solution. Toxicity to man has been shown, but no water quality guidelines have been established. Levels of As reported for peatland runoff waters are not high compared to the guidelines (Tables 7.1, 7.2 and 7.3). Data of Washburn and Gillis Associates Ltd. (1983) appear to be below a detection limit, and it is difficult to conclude whether As levels are high compared to the guidelines (Table 7.20).

7.5.12 Selenium

Selenium accompanies sulphur and oxygen in Group 6a of the periodic table. Dissolved species are SeO_3^{2-} and SeO_4^{2-}, but Se levels are predominantly controlled by sorption to sediments and precipitation from solution. Selenium is both an essential element and a cumulative poison to animals and man. It is moderately toxic to plants and aquatic life. It has also been shown to offer protection against toxicity from Hg and Cd in plants and animals. Guidelines have been established only for human, livestock, and aquatic life uses (Tables 7.1, 7.2, and 7.3). Levels reported for peatland waters (Table 7.21) are well below the guideline levels.

Table 7.19. Lead in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Pb $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	<0.13
Minnesota	fen	natural	runoff	14	<0.13
	bog	mined	runoff	46	<0.13
	fen	cultivated	runoff	12	<0.13
Clausen and	fen	natural	runoff	54	-
Brooks 1983a,b	fen	natural	runoff	55	-
	bog	natural	runoff	66	-
	bog	mined	runoff	5	-
Osborne, in	bog	mined	upstream	9-15	<0.02
Washburn and			discharge	9-15	<0.02
Gillis Associates			downstream	9-15	<0.02
Ltd. 1983					
Newfoundland					
Washburn and					
Gillis Associates					
Ltd. 1983					
Newfoundland	bog	drained	pit	2	<0.01
Ontario	bog	drained	pit	2	<0.01
British Columbia	bog	mined	pit	2	<0.01
			recycle pond	1	<0.01

Table 7.20. Arsenic and antimony in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	AS $ug \cdot L^{-1}$	Sb $ug \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	3.0	-
Minnesota	fen	natural	runoff	14	2.0	-
	bog	mined	runoff	46	3.0	-
	fen	cultivated	runoff	12	<1.0	-
Clausen and	fen	natural	runoff	54	3.0	-
Brooks 1983a,b	fen	natural	runoff	55	3.0	-
	bog	natural	runoff	66	2.0	-
	bog	mined	runoff	5	4.0	-
Osborne, in	bog	mined	upstream	9-15	<5.0	-
Washburn and			discharge	9-15	<5.0	-
Gillis Associates			downstream	9-15	<5.0	-
Ltd. 1983						
Newfoundland						
Washburn and						
Gillis Associates						
Ltd. 1983						
Newfoundland	bog	drained	pit	2	<2.0	<5.0
Ontario	bog	drained	pit	2	<2.0	<5.0
British Columbia	bog	mined	pit	2	<2.0	<5.0
			recycle pond	1	<2.0	<5.0

Table 7.21. Selenium in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Se $mg \cdot L^{-1}$
Clausen 1980	fen	natural	runoff	46	0.001
Minnesota	fen	natural	runoff	14	0.001
	bog	mined	runoff	46	0.001
	fen	cultivated	runoff	12	<0.001
Clausen and	fen	natural	runoff	54	0.001
Brooks 1983a,b	fen	natural	runoff	55	0.001
	bog	natural	runoff	66	0.001
	bog	mined	runoff	5	0.001
Osborne, in	bog	mined	upstream	9-15	<0.05
Washburn and			discharge	9-15	<0.05
Gillis Associates			downstream	9-15	<0.05
Ltd. 1983					
Newfoundland					
Washburn and					
Gillis Associates					
Ltd. 1983					
Newfoundland	bog	drained	pit	2	<0.004
Ontario	bog	drained	pit	2	<0.004
British Columbia	bog	mined	pit	2	<0.004
			recycle pond	1	<0.004

7.5.13 Beryllium, Strontium, and Barium

These elements, along with Mg and Ca, constitute the alkaline earth metals of Groups 2a in the periodic table. Beryllium is the lightest of these and is present in particulate rather than in dissolved form in all but very strongly acidic waters. Beryllium and its compounds are very toxic to animal and plant life. Barium and strontium are heavier members of the alkaline earths which occur in waters in very low amounts. Barium is toxic to organisms, whereas Sr does not seem to have harmful effects. Livestock and aquatic life guidelines have been established for Be, and a Ba limit exists for domestic consumption (Tables 7.1, 7.2, and 7.3). Levels of these elements in peatland waters are presented in Table 7.22. The levels for Ba are substantially below the recommended limits. It is not possible to conclude whether Be may be a concern in peatland waters since reported levels were below the detection limit.

7.5.14 Halides

Bromide, chloride, and flouride properties and water quality guidelines are discussed by McNeely et al. (1979) and CCREM (1987). However, few data for these elements in peatland runoff waters have been found for comparison. Cl levels of 68 to 168 $mg \cdot L^{-1}$, found in tile drainage waters in farmed peat soils of Holland Marsh, Ontario (Miller 1979), are below the CCREM domestic drinking water level (Table 7.2). Chloride can occur in significant quantities in surface waters, but Br and F likely occur in minute quantities in peatland waters.

7.6 ORGANIC MATERIALS

Organic parameters listed in the Alberta Environment (1977) surface water quality objectives are: carbon chloroform and carbon alcohol extracts, which are measures of materials such as insecticides, pesticides, herbicides, and synthetic industrial chemicals which are toxic in trace quantities; methylene-blue-active substances, which are basically detergents; methyl mercaptan;

Table 7.22. Berryllium, strontium, and barium in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Be $ug \cdot L^{-1}$	Sr $mg \cdot L^{-1}$	Ba $mg \cdot L^{-1}$
Osborne, in Washburn and Gillis Associates Ltd. 1983 Newfoundland	bog	mined	upstream	9-15	-	-	0.02
			discharge	9-15	-	-	0.17
			downstream	9-15	-	-	0.05
Washburn and Gillis Associates Ltd. 1983							
Newfoundland	bog	drained	pit	2	<0.1	0.03	<0.005
Ontario	bog	drained	pit	2	<1.0	0.14-0.51	<0.005-0.006
British Columbia	bog	mined	pit	2	<1.0	0.03-0.06	<0.005-0.11
			recycle pond	1	<1.0	0.09	21

ether-soluble oil and grease; pesticides; phenolics; and resin acids. The CCREM guidelines in this category pertain to numerous individual and groups of organic compounds. The Alberta Environment (1977) surface water quality objectives are referred to here since some of the pertinent literature is based on these guidelines.

Grease and oil from sources such as manufacturing industries and the operation of outboard motors causes hydrocarbon pollution in fresh water. Hydrocarbons are organic compounds, containing only hydrogen and carbon, and are divided into aliphatic and aromatic structural classes. Natural plant and animal hydrocarbons are ubiquitous in both terrestrial and aquatic environments, while petroleum hydrocarbons constitute the major pollution source.

Data on phenolics and oil and grease have been collected for several creeks and rivers in the Athabasca oil sands area (Corkum 1985). Many of these water courses flow through extensive peatland areas. Highest oil and grease values were found for the Muskeg River. However, these were not related to water quality guidelines, as no quantitative criteria have been established for Alberta. (The guideline indicates that oil and grease should be substantially absent, with no iridescent sheen.) The Alberta guideline for oil and grease is similar to that suggested for hydrocarbons by McNeely et al. (1979) which indicates, in addition to the above, that petroleum hydrocarbons should not be present in concentrations that can be detected by odour, that can cause tainting of edible aquatic organisms, or that can form deposits on shorelines and bottom sediments that are detectable by sight or odour, or are deleterious to resident aquatic organisms.

In the study by Corkum (1985), levels of phenolics exceeded the Alberta Environment (1977) surface water quality objectives in several of the watercourses in the region, and were highest at some sampling points in the Muskeg River and in Big Point Channel in the Athabasca delta area. Sources of phenolic material include industrial and municipal wastes as well as decaying plant material.

Since only the latter source exists in most of the watercourses in the Athabasca oil sands area, natural sources likely predominate. However, it is not possible to ascertain the degree to which peatlands account for the source.

Phenolic compounds consist of aromatic benzene rings with one or more attached hydroxyl groups. A limit of 0.2 $mg \cdot L^{-1}$ phenolics is considered to be safe for fish and aquatic life, but the water quality standard is considerably lower based on fish flesh tainting concentrations (McNeely et al. 1979). The limit for freshwater aquatic life is 1 $\mu g \cdot L^{-1}$, while that for domestic and livestock consumption is 2 $\mu g \cdot L^{-1}$ (CCREM 1987).

Guidelines have been provided by CCREM (1987) for a large number of organic compounds, almost all of which are of industrial or related anthropogenic origin. These include numerous pesticides. There have been very few studies in which these parameters were examined in peatland or related downstream waters. Data for some organic compounds were obtained for peatland waters by Washburn and Gillis Associates Ltd. (1983). The data were collected from mined bogs in Newfoundland, Ontario, and British Columbia. Three groups of compounds were investigated as indicated in the list below:

1. Phenolics
   - 2-chlorophenol
   - 2-nitrophenol
   - 2,4-dimethylphenol
   - 2,4-dichlorophenol
   - 4-chloro-3-methylphenol
   - 2,4,6-trichlorophenol
   - 2,4-dinitrophenol
   - 4-nitrophenol
   - 2-methyl-4,6-dinitrophenol
   - pentachlorophenol
2. Polyaromatic hydrocarbons
   - napthalene
   - acenaphthalene

fluorene
phenanthrene
anthracene
fluoranthene
pyrene
chrysene
benz(a)pyrene

3. Phthalate esters
    dimethylphthalate
    diethylphthalate
    di-n-butylphthalate
    bis(2-ethylhexyl)phthalate
    acenaphthene (extract std)

The only compounds found at levels greater than trace amounts in any of the water samples were di-n-butylphthalate, diethylphthalate, and bis(2-ethylhexyl)phthalate. In all cases these were found in concentrations less than 20 $mg \cdot L^{-1}$ which is much lower than acute toxicity levels for marine systems. However, the levels in some samples exceeded guidelines for prevention of chronic toxic effects in aquatic systems. The authors indicated that further data on the frequency of occurrence of levels above the guidelines, and on movement of phthalates in receiving waters, were necessary to determine whether bog waters could produce toxic effects (Washburn and Gillis Associates Ltd. 1983).

The clorinated phenols, nitrophenols, polyaromatic hydrocarbons, and phthalates investigated in the above study are all industrial chemicals. The increasing usage and numbers of industrial and agricultural chemicals have led to increased concentrations in various aquatic components of the biosphere. Peatlands are not sources of these materials, but application of these substances to peat soils can have a bearing on quality of runoff and downstream waters. They are complex compounds, they generally possess persistent toxic properties, and their concentrations are usually very low. Thus, analytical procedures for them are difficult, but continuing new improvements have enabled lowering of detection limits.

Another indication of levels of various organic compounds has been obtained by estimation of water-soluble substances extracted by steam distillation from air-dried peat (Naucke et al. 1972; reviewed by Washburn and Gillis Associates Ltd. 1982). Only trace levels of the following groups of compounds were found: phenols and derivatives, aromatic acids, aromatic aldehydes and ketones, aliphatic acids, furan derivatives, and chinoidic substances.

Abundances of phenolic compounds in peat as well as of other groups of compounds have been determined using a variety of extractants (Fuchsman 1980; Morita 1980). However, data obtained through use of relatively strong extractants are very difficult to relate to levels in natural waters. Since there are very few studies of levels of organic compounds in waters of peatland and associated water bodies, it is apparent that more chemical investigations are required.

7.7 PHYSICAL PARAMETERS

Water quality parameters in this class include colour, temperature, odour, taste, dissolved gas supersaturation, specific conductance, total dissolved solids, turbidity, suspended solids, and clarity. Some of these have already been discussed in other sections due to their close association with other chemical properties.

7.7.1 Suspended Solids

Suspended solids are a measure of material exceeding colloidal size (>0.45 μm). Turbidity is a measure of both the colloidal and suspended solids load. Materials accounting for these parameters include silt, clay, organic matter, plankton, and microscopic organisms which are usually held in suspension by turbulent flow and Brownian movement. Excessive suspended solids and turbidity can have various adverse effects on aquatic life and on suitability for domestic use, as discussed by McNeely et al. (1979)

and CCREM (1987). CCREM guidelines for suspended solids are provided for freshwater aquatic life (Table 7.1).

Some data for suspended solids in peatland waters are provided in Table 7.23. In Minnesota, Clausen and Brooks (1983b) have provided data which show that the mean suspended sediment level in mined bog runoff, at 13.7 $mg \cdot L^{-1}$, was significantly different from the mean level of 5.1 $mg \cdot L^{-1}$ in control bogs.

Sallantaus and Patila (1983) and Heikurainen and Kenttamies (1978) have reported that the concentrations of suspended solids in natural peatlands in Finland are usually negligible (<1 $mg \cdot L^{-1}$). Ditch excavations in peatlands result in considerable increase in suspended solids in runoff waters. The highest level, found after drainage, was 38 $mg \cdot L^{-1}$, while the mean level from about 50 sites was about 10 $mg \cdot L^{-1}$. The total discharge for a drained peatland was found to be 12 $t \cdot km^{-2}$; this specific discharge value was not considered to be a high value. The researchers indicated that these suspended solids were nearly totally organic and very voluminous. A considerable amount of this material probably sedimented in the water dammed behind the weir where water samples were taken for analysis. Where considerable erosion of mineral soil has occurred, specific discharges of suspended sediment of up to several hundred $t \cdot km^{-2}$ can occur in forest drainage areas (Seuna 1982; cited in Sallantaus and Patila 1983). However, erosion of peat can result in much larger volumes of sediment as compared to mineral soils due to the lower bulk densities (Sallantaus and Patila 1983).

Data from R.S.F.S.R. (Largin 1976) indicate that there is little difference between drainage waters of natural and cultivated bogs and fens (Table 7.23). Largin's data were obtained from peatlands which had been drained and cultivated from several years. The data of Sallantaus and Patila (1983) show that the greatest impact on concentration of suspended solids is during and immediately after excavation. In the mined bogs of Minnesota, drainage and peat moss extraction activities occur more or less continuously and the suspended solids levels would be expected to be elevated for the

Table 7.23. Suspended solids and turbidity in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	Suspended Solids, $mg \cdot L^{-1}$	Turbidity Jackson Units
Brown et al. 1980 Minnesota	shallow peat	wild rice	supply	-	-	2.8
			discharge	-	-	2.3
Clausen 1980 Minnesota	fen	natural	runoff	46	3.9	-
	fen	natural	runoff	14	2.4	-
	bog	mined	runoff	46	15.8	-
	fen	cultivated	runoff	12	16.7	-
Clausen and Brooks 1983a,b Minnesota	fen	natural	runoff	54	5.4	-
	fen	natural	runoff	55	5.3	-
	bog	natural	runoff	66	5.1	-
	bog	mined	runoff	5	13.7	-
Kenttamies 1981 Finland	bog	natural	-	-	6.3	-
	bog	drained, fertilized-forestry	-	-	7.6	-
Kenttamies and Laine 1984 Finland	bog	undrained (3y)	-	-	0.1-0.9	-
	bog	drained (2y)-forestry	-	-	0.1-6.2	-
	bog	fertilized (2y)-forestry	-	-	4.7-7.4	-
Largin 1976 RSFSR	bog	natural	pit	-	8-20	-
	bog	cultivated	ditch	-	8-21	-
	fen	natural	pit	-	15-35	-
	fen	cultivated	ditch	-	26-35	-
Sallantaus and Patila 1983 Finland	bog	natural	weir	several April - Oct	<1	-
	bog	mined	weir		10	-
	bog	mined	weir		<10	-
	bog	mined	weir		<10	-
	bog	mined	weir		<10	-
Osborne, in Washburn and Gillis Associates Ltd. 1983 Newfoundland	bog	mined	upstream	13-15 July - Nov 1982	2.5	0.67
			discharge		9.1	3.53
			downstream		3.2	2.46

duration of mining activities. Each of the above researchers also noted that suspended sediment levels and total runoff amounts depend on the season and on precipitation events. The highest total runoff occurs during spring thaw and high precipitation events. However, concentrations aren't necessarily highest during these periods.

7.7.2 Turbidity

CCREM guidelines for turbidity pertain to domestic water quality (Table 7.2); this limit (<5.0 NTU) also applies to recreational water quality (CCREM 1987). Few data on turbidity of peatland and runoff waters can be found in the literature. Osborne (1983) reported turbidity data for a drained and mined bog in Newfoundland (Table 7.23). Values for waters sampled upstream from the bog varied between 0.5 and 2 Jackson turbidity units between July and November 1980. At a sample point where the stream ran through the mined bog area, values ranged from 0.5 to about 15 units. At a sampling station 1.5 km downstream from the mined area, turbidity was relatively constant over the sampling period and ranged between 1 and 4 Jackson units. These data thus indicate that peatland developments such as mining can increase the turbidity levels in streams. However, the magnitude of the increase in the Newfoundland study was within the CCREM water quality guidelines, and the levels indicated above are not high when compared to high turbidity levels of some natural waters, which are in the range of several hundred Jackson units (McNeely et al. 1979).

7.7.3 Colour

Colour in peatland and associated waters is probably closely related to levels of organic materials because of the pigmentation produced by humic materials. Generally, waters with less than 3.0 $mg \cdot L^{-1}$ total organic carbon are considered to be relatively clear (McNeely et al. 1979).

Criteria for colour have been developed mainly with regard to human use of water. The acceptable limit for true colour in public drinking waters is 15 Pt-Co units (or true colour units) (McNeely et al. 1979; CCREM 1987). For waters used for direct recreational contact such as swimming, a guideline for the related parameter of clarity has been established by CCREM (1987). This guideline indicates that water should be sufficiently clear for a Secchi disc to be visible at a minimum of 1.2 m.

There are two measures of colour. True colour is imparted by dissolved colouring compounds, while apparent colour includes the influence of suspended material in water. This parameter is measured according to the platinum-cobalt (Pt-Co) scale, which compares the colour of the water sample with that of a series of standard chemical solutions. Water with colour less than 10 Pt-Co units appears visually clear, while a value of 100 resembles black tea and may be encountered in waters draining peat deposits (McNeely et al. 1979).

Data for colour in runoff waters of developed and undeveloped peatlands are given in Table 7.24. Data for total carbon and for the humic and fulvic acid forms of organic carbon are also provided in Table 7.24 because of the close relationship with colour. Most of the colour data indicate that peatland runoff waters are generally very dark, and that waters from bogs are usually somewhat darker than those from fens. The dark colours are indicative of high levels of humic substances, as discussed in Chapter 6.

7.7.4 Temperature

The CCREM guideline for temperature indicates that thermal additions should not alter thermal stratification or turnover rates, exceed maximum weekly average temperatures, or exceed minimum short-term temperatures.

Clausen (1980) found the following mean temperatures during a year of monitoring in 1978 to 1979 in Minnesota:

1. Transitional fen - 9°C;
2. Fen - 15°C;
3. Mined bog - 11°C; and
4. Fen cultivated for 25 years - 18°C.

Table 7.24. Colour and organic materials in waters of natural and developed peatlands.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	C-Total $mg \cdot L^{-1}$	C-Total Organic $mg \cdot L^{-1}$	C-Total Inorganic $mg \cdot L^{-1}$	Humic Acid $mg \cdot L^{-1}$	Fulvic Acid $mg \cdot L^{-1}$	Colour Pt-Co Units
Brown et al. 1980 Minnesota	shallow peat soil	wild rice	supply	-	-	-	-	-	-	249
			discharge	-	-	-	-	-	-	249
Clausen 1980 Minnesota	fen	natural	runoff	46	-	-	-	-	-	266
	fen	natural	runoff	14	-	-	-	-	-	129
	bog	mined	runoff	46	-	-	-	-	-	372
	fen	cultivated	runoff	12	-	-	-	-	-	529
Clausen and Brooks 1983a.b	fen	natural	runoff	54	-	-	-	8	86	242
	fen	natural	runoff	55	-	-	-	9	89	260
	bog	natural	runoff	66	-	-	-	11	100	311
	bog	mined	runoff	5	-	-	-	8	136	401
Heikurainen et al. 1978 Finland	bog	natural	-	-	-	22	-	-	-	-
	bog	drained-forestry	-	-	-	19	-	-	-	-
	swamp	natural	-	-	-	25	-	-	-	-
	swamp	drained-forestry	-	-	-	22	-	-	-	-
Kenttamies 1981 Finland	bog	natural	-	-	-	23	-	-	-	-
	bog	drained, fertilized-forestry	-	-	-	21	-	-	-	-
Kenttamies and Laine 1984 Finland	bog	undrained (3y)	-	-	-	30-39	-	-	-	-
	bog	drained (2y) forestry	-	-	-	28-34	-	-	-	-
	bog	fertilized (2y) forestry	-	-	-	23-24	-	-	-	-

continued . . .

Table 7.24. Concluded.

Source	Peatland Type	Peatland Use	Sample Type	No. of Samples	C-Total $mg \cdot L^{-1}$	C-Total Organic $mg \cdot L^{-1}$	C-Total Inorganic $mg \cdot L^{-1}$	Humic Acid $mg \cdot L^{-1}$	Fulvic Acid $mg \cdot L^{-1}$	Colour Pt-Co Units
Largin 1976	bog	natural	pit	-	-	-	-	90-270	-	-
RSFSR	bog	cultivated	ditch	-	-	-	-	130-290	-	-
	fen	natural	pit	-	-	-	-	15-120	-	-
	fen	cultivated	ditch	-	-	-	-	30-90	-	-
Sallantaus and	bog	natural	weir	several	-	18-50	-	-	-	-
Patila 1983	bog	mined	weir	from	-	32-35	-	-	-	-
Finland	bog	mined	weir	April to	-	24-43	-	-	-	-
	bog	mined	weir	October	-	55-58	-	-	-	-
	bog	mined	weir		-	34-52	-	-	-	-
Osborne, in	bog	mined	upstream	13-15,	-	19	-	-	-	-
Washburn and			discharge	July to	-	29	-	-	-	-
Gillis Associates			down-	November	-	18	-	-	-	-
Ltd. 1983			stream	1982						
Newfoundland										
Washburn and										
Gillis Associates										
Ltd. 1983										
Newfoundland	bog	drained	pit	2	21-40	21-40	<2	-	-	-
Ontario	bog	drained	pit	2	48-59	47-51	2-8	-	-	-
British Columbia	bog	mined	pit	2	90-97	88-93	2-5	-	-	-
			recycle pond	1	136	136	<2	-	-	-

In Minnesota, Clausen and Brooks (1983a) found that the temperature of mined bog runoff at 16.1°C was significantly different from that of control bogs at 11.1°C. The increase in runoff temperature likely resulted from removal of the peatland vegetation along stream banks, and from increases in surface peat temperature following vegetation removal (Clausen and Brooks 1983a). These studies did not indicate to what degree water temperatures were affected downstream from sample collection points. It is unlikely that a significant downstream effect would occur as a result of small areas of peatland development. The effect could be significant where development for agriculture or mining is regionally extensive.

7.7.5 Odour

Odour is a parameter of importance with regard to domestic water supplies, but no CCREM guideline has been established. The Alberta Environment (1977) surface water quality objectives indicate that the cold (20°C) threshold odour number (TON) is not to exceed eight. Reports of odour levels in peatland and runoff waters have not been found. Nevertheless, peat water may contribute to odours due to its content of phenolics in tannins, lignins, and humic substances. Water treatment can convert weak smelling substances, such as phenols and amines, to substances with very strong odours, such as chlorophenols and chloramines (CCREM 1987). Strong odours during early spring in Edmonton have been attributed to materials in runoff of snowmelt. The North Saskatchewan River watershed upstream from Edmonton consists predominantly of forest land with a significant proportion of peatlands (see map accompanying this report). It is possible that organic compounds from peatlands as well as from surface organic (litter) layers of mineral soils contribute to the odour problem in municipal water supplies.

7.8 OTHER PARAMETERS

Radiological and biological parameters are two other categories of water quality guidelines considered by CCREM (1987).

These have generally not been considered in terms of peatland impacts on water quality. These are not discussed further in this report and CCREM (1987) should be consulted for specifics of the guidelines and for other information. Some baseline information on the microbiology of peats in Alberta can be found in Christensen and Cook (1970).

7.9 SUMMARY OF SOME RECENT RESEARCH RESULTS

Since the initial compilation of information from the literature for this report, a number of other reports on water quality impacts of peatland drainage, particularly for forestry, have been published. Some of these recent studies are summarized in this section.

The effects of drainage on groundwater levels and stream water quality were investigated on a coniferous swamp in Alberta as part of a program on forestry improvement (Hillman 1988). Chemical parameters, suspended sediment and specific conductance were measured upstream and downstream from points where drainage entered a stream channel. No significant differences were detected between upstream and downstream levels of suspended sediment, specific conductance, and 13 of 16 inorganic elements. The drainage increased levels of Fe in the stream, but appeared to lower the levels of Al and K. The inclusion of sediment ponds and buffer zones between drainage ditches and the stream were considered to minimize the deleterious effects of ditching on stream water quality.

A study, similar to the above, was carried out on black spruce swamp ecosystems in Ontario (Berry 1989). Concentrations of the common ions did not exceed CCREM guidelines in downstream waters. Al, Fe, and Cu were found to be consistently higher than guideline levels at both control and downstream sites. Other trace metals exceeded guidelines only a few times over two years of the study. None of the nutrients exceeded guidelines. Alkalinity and pH had increased as a result of drainage, but values remained within CCREM limits. Turbidity and suspended solids were found to exceed guidelines during high flow periods. This problem was particularly apparent at a site where a collector pond had nearly filled with sediment and was, therefore, no longer efficient in removal of suspended solids. This observation suggests, as in the Alberta study, that sediment ponds and buffer zones are effective in minimizing impacts on water quality.

A considerable amount of work on drainage for forestry is being conducted in Sweden and Finland. Much of what work, however, is being carried out on mineral soils that have become waterlogged as a result of clear-cutting. In one such study in Sweden, thin peaty soils and true peatlands occurred commonly in these study areas (Lundin 1988). Since drainage ditches frequently penetrate mineral soils in these areas, the chemical composition of runoff showed increased pH, alkalinity, and concentrations of cations and sulphate. Levels of organic C in stream water decreased at drained sites. Changes in levels of some constituents in water depended on the type of peatland. For example, stream levels of N increased for drained fens, but decreased for drained bogs. Physical parameters and trace elements were not addressed in this study.

A study of clear-cutting and drainage on predominantly mineral soils in a group of small watersheds in Finland was reported by Ahtiainen (1988). Suspended solids in a drained area were, on average, 10 times the control value. Organic matter concentrations were high soon after drainage, but then decreased over time. Levels of different forms of P and N increased with drainage. Annual average water temperature increased after block cutting and ditching. Mean pH values decreased slightly after drainage, but later increased due to contact of water with mineral soil in the ditches. This study indicated that the major concern was with regard to suspended solids. Due to high sensitivity of the mineral soil to erosion in the study area, sediment loads were high not only after drainage but during any high-flow event.

Effects of drainage and clear-cutting on primary production in stream waters in the Finnish study described above (known as the Nurmes study) was also reported (Huttenen et al. 1988). Algal and bacterial primary production both increased considerably as a result of the forestry treatments. These effects could be measured three to four years after the treatments. However, restricted clear-cutting with a zone left uncut along water courses appeared to afford effective protection to the water. In the same study area, long- and short-term variation in particulate organic carbon, dissolved organic carbon, and total organic carbon were examined (Hovi 1988). The variations were related to weather conditions, although relatively high levels could be related to forestry measures undertaken in some peatland basins.

7.10 SUMMARY AND CONCLUSIONS

The effects of natural and developed peatland runoff waters on the quality of receiving water bodies were reviewed by comparing data found in the literature for natural and developed peatlands with water quality guidelines developed by the Canada Council of Resource and Environment Ministers (CCREM 1987) and Alberta Environment (1977). Constituents of peatland runoff waters in which concentrations exceeded or fell short of water quality guidelines were considered as having potential for noncompliance in the receiving waters as well. The possible effects are summarized in Table 25. Potential impacts from both natural and developed peatlands is indicated in the table. Possible noncompliance with water quality guidelines is indicated in Table 25 by an asterisk. As an example, a general conclusion from the literature is that the concentration of the first constituent listed (oxygen) will likely remain the same or decrease in receiving waters as a result of runoff from natural bogs or fens. However, fens or bogs developed for farming, mining, or other uses can potentially lower oxygen levels to below water quality guidelines in receiving waters.

Table 7.25. Summary of possible effects of peatland runoff on water quality parameters in downstream waters. (For abbreviations see p. 262.)

	Natural		Developed	
	Fen	Bog	Fen	Bog
Major Dissolved Substances and Related Properties				
Oxygen	S,D	S,D	D*	D*
BOD	I	I	I*	I*
COD	I	I	I*	I*
pH	S,D	D	S,D	D*
Acidity	S,I	I	S,I	I
Alkalinity	S,D	D	S,D	D
Calcium	S,D	D	S,D	D
Magnesium	S,D	D	S,D	D
Sodium	D	D	I	D
Potassium	D	D	D	D
Sulphur				
Sulphate	S	S	I	S
Sulphide	U	U	I	I
Specific Conductance	D	D	D	D
Hardness	D	D	D	D
TDS	D	D	D	D
Major Nutrients				
Total N	I	I	I*	I*
Nitrate-N + Nitrite-N	S	S	I	I
Nitrite-N	S	S	I	I
Ammonia-N	I	I	I*	I*
Total P	S,D	S	I*	I*
Total PO_4	S	S	I*	I*
Inorganic PO_4	S	S	I*	I*
Particulate Materials				
Suspended Solids	S	S	I*	I*
Turbidity	D	I	I*	I*

continued . . .

Table 7.25. Continued.

	Natural		Developed	
	Fen	Bog	Fen	Bog
	Minor and Trace Elements			
Aluminum	S	S,I	I*	I*
Antimony	U	U	U	U
Arsenic	I	I	I	I
Barium	D	D	D	D
Beryllium	D	D	D	D
Boron	D	D	D	D
Cadmium	U	U	U	U
Chromium	D	D	D	D
Cobalt	D	D	D	D
Copper	D	D	I	I
Halides				
Bromide	U	U	U	U
Chloride	S	S	I	I
Fluoride	U	U	U	U
Iron	S,I	S,I	I*	I*
Lead	U	U	U	U
Lithium	U	U	U	U
Manganese	I	I	I*	I*
Mercury	I	I	I*	I*
Molybdenum	D	D	D	D
Nickel	D	D	D	D
Selenium	D	D	D	D
Silica	D	D	D	D
Silver	U	U	U	U
Strontium	U	U	U	U
Thalium	U	U	U	U
Titanium	D	D	D	D
Tungsten	U	U	U	U
Uranium	U	U	U	U
Vanadium	D	D	D	D
Zinc	S,I	S,I	I*	U

continued . . .

Table 7.25. Concluded.

	Natural		Developed	
	Fen	Bog	Fen	Bog
		Organic Materials		
TOC	S,I	I	I	I
DOC	S,I	I	I	I
Humic acids	I	I	I	I
Fulvic acids	I	I*	I*	I*
Colour	I	I	I	I
Polyaromatic Hydrocarbons	U	I	U	I*
Phthalates	U	I	U	I
Methylmercaptan	U	U	U	U
Oil and grease	S*	S*	S*	S*
Phenolics	I	I	I	I
Resin acids	U	U	U	U
Cyanide	U	U	U	U
		Other Properties		
Temperature	S	S	I	I
Odour	S	S	S	S
Bacteria	S	S	S	S

Abbreviations:
- S = same concentrations in runoff and in receiving waters.
- I = increase - peatland runoff may increase or enrich levels of constituent in receiving waters.
- D = decrease - peatland runoff may decrease or dilute level of constituent in receiving waters.
- * = increase or decrease in level of constituent potentially results in noncompliance with water quality guidelines.
- U = unknown due to insufficient information.
- BOD = biochemical oxygen demand.
- COD = chemical oxygen demand.
- TDS = total dissolved solids.
- TOC = total organic carbon.
- DOC = dissolved organic carbon.

The following are comments on peatland water constituents and properties of greatest concern:

1. Oxygen can be depleted to below guideline levels because of high biochemical oxygen demand and chemical oxygen demand resulting from input of organic materials. There is very little information available about organic matter levels, forms, and interactions in peatland and runoff waters in Alberta.
2. Natural peatlands contribute to the acidity of downstream waters. Developed peatlands can produce considerable acidity in excess of that produced by peatlands in the natural state due to the oxidation of various compounds. Most studies indicate that the mineral-soil bottoms of drainage ditches neutralize the acidity. Downstream waters could be affected in areas where the underlying soil has low neutralizing capability, or where drained peat deposits are deep.
3. Input of N and P compounds to water bodies and contribution to eutrophication has been reported from various countries. The highest levels have been found where heavy fertilization is practised.
4. Of the minor and trace elements, Al, Fe, Mn, Hg, and Zn are of greatest concern in terms of exceeding water quality guideline levels due to development. Any studies of the above should also include Cd, Pb, and Cu as very little information is available about them.
5. Some organic compounds have been shown to exceed recommended guideline levels in other provinces. Information about the abundance of various organic chemicals such as polyaromatic hydrocarbons, phthalate esters, and phenolics in peatland and runoff waters is needed. Since many of these have industrial origins, operations such as peat mining may pose a greater concern than uses such as forestry or agriculture.

6. Monitoring and investigations of the nature and interactions of the various water quality parameters indicated above are required as very little information is currently available for Alberta, and indeed for peatlands in the rest of the country.

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8. SUMMARY OF RESEARCH RELATED TO PEATLAND HYDROLOGY AND HYDROCHEMISTRY IN ALBERTA

L.W. Turchenek

8.1 INTRODUCTION

This section summarizes work carried out within the province of Alberta on hydrology and hydrochemistry of peatlands and associated water bodies. Investigations which only indirectly pertain to peatlands but which can provide baseline data for further studies are reviewed in addition to research dealing directly with peatlands. There is generally a scarcity of data and research with respect to peatlands and associated waters not only in Alberta but in most other provinces as well. There nevertheless have been some important contributions made by researchers in Alberta. Some information is available in the form of environmental impact assessment reports. These are reviewed herein; however, it is likely that some reports were not found due to classified status, limited distribution, 'out of print' status, and other reasons.

8.2 ENVIRONMENT CANADA PROGRAMS

Surface water data of Environment Canada, Inland Waters Directorate, Water Resources Branch, is available in the form of a manual file and also as a computerized data base referred to as NAQUADAT (1985). The manual file contains information on over 400 water bodies in Alberta. The data have generally been gathered for larger water bodies. As such, they are not useful for relating water quality to peatland sources of various constituents, but they may be useful in broad, regional assessments of peatland development impacts on water quality.

8.3 ALBERTA ENVIRONMENT PROGRAMS

Various water quality research and monitoring activities are carried out by different agencies of Alberta Environment.

Surface water data are contained in the manual and computer (NAQUADAT) files referred to above. Other files include the river water quality surveys computer file and the lake water quality surveys computer file.

Like the Environment Canada programs, the Alberta Environment monitoring and research reports are specific to larger lakes and water courses in the province. They may be sources of data for activities in which water quality is related to peatlands on a broad, regional level.

8.4 ALBERTA FORESTRY, LANDS AND WILDLIFE PROGRAMS

Studies of specific water bodies for evaluations of fish habitat are carried out by the Fish and Wildlife Division, Alberta Forestry, Lands and Wildlife. The data are maintained as manual files and are available upon request for resource evaluation and planning purposes. The amount of water chemistry data differs with different streams and lakes, and comprehensive water quality data are available for relatively few of these. Nevertheless, these files may provide a source of data for future site specific and regional investigations of water quality in relation to peatlands.

8.5 REGIONAL STUDY OF WATER QUALITY IN THE ATHABASCA OIL SANDS AREA

Corkum (1985) reviewed various projects carried out for the Alberta Oil Sands Environmental Research Program (AOSERP), as well as routine water quality monitoring data, with the objectives of summarizing water quality constituents and examining relationships between these constituents and changes in land form, hydrology, and industrial development. The constituents examined were pH, total alkalinity, phenol alkalinity, total hardness, conductivity, turbidity, colour, Si, Ca, Mg, Na, K, F, HCO_3, CO_3, Cl, SO_4, O_2, total organic C, filterable residue, nonfilterable residue, total inorganic C, phenolics, oil and grease, total P, orthophosphate, total N, NH_3, $NO_3 + NO_2$, Al, Ag, B, Cd, Cr, Co, Cu, Fe, Pb, Mn, Hg, Ni, Se, Ag, V, and Zn. Constituents in the Athabasca River were examined with regard to longitudinal and seasonal changes and to effects of point-source effluents on water quality. Baseline data and relationships among the parameters were presented for east, west, and south drainages entering the Athabasca River, as well as of the Athabasca Delta drainage. Principal components analysis was used to help delineate sites with similar water quality characteristics. The resulting groupings could be related to the geology of the region.

Many of the creeks in the Athabasca oil sands area run through extensive peatland areas. Of the various water quality parameters, Corkum (1985) attributed only high values for colour and organic carbon to peatland sources. It is possible that various other water quality parameters are related to muskeg sources in this region. Among routine parameters, pH, conductivity, and the major ions appear related to regional geology. Values for colour and levels of organic carbon and phenolics were higher in areas with extensive peatland coverage than in areas with predominantly mineral soils. Relatively high levels of the various forms of N and P were found in most rivers and creeks of the region. Among the metals, noncompliance with the surface water quality objectives was noted for the potentially toxic metals Ag, Cd, Pb, Hg, and As. Other metals which exceeded the objectives included Fe, Mn, Zn, and Cu. Maximum values for B exceeded the Alberta Surface Water Quality Objectives in some streams. Maximum Al values were also relatively high in some instances.

No conclusions can be reached about relationships among water quality parameters and occurrence of peatlands, with the exception that colour and organic carbon levels most likely originate from muskeg areas. The levels oı N and P are of interest in that they appear to be highest in streams in high muskeg cover areas. Mercury is also of interest in view of its tendency to become more mobilized with increases in levels of dissolved organic matter.

Corkum (1985) noted instances where elevated levels of some water quality parameters could be related to anthropogenic activities such as oil sands mining and road building. However, most of the streams run through relatively undisturbed areas and natural sources account for the measured levels of constituents in waters of this region.

The water quality studies of streams in the Athabasca oil sands area have provided a database for secondary waterways with coverage that appears to be unmatched in other areas of the province. It is possible that further analysis of the original data could link some of the parameters to peatland occurrence. Water quality data could be examined in relation to kinds and extent of soils and geology, including peatlands, within individual drainage basins. Seasonal trends could be examined and inferences made about sources of water and dissolved constituents according to the method described by Schwartz and Milne-Home (1982a, 1982b). Such analyses were not attempted within the project reported herein; the intent is only to identify research and databases related to peatlands and associated waters, and to suggest additional research and analyses which could be carried out.

8.6 WATER CHEMISTRY STUDIES OF THE MUSKEG RIVER BASIN

Schwartz and Milne-Home (1982a) undertook a detailed, chemically based study of watersheds with extensive muskeg terrain using the Muskeg River area in the Athabasca oil sands area as the study area. The main factor found to influence the chemistry of the stream water at any given time was the relative quantity of water which enters the stream from any one of four well-defined sources: groundwater, muskeg, surface runoff, and direct precipitation. Creeks were at baseflow during the winter when contributions to streamflow from muskeg, surface runoff, and direct precipitation were negligible. Following spring snowmelt, the muskeg contribution to streamflow is larger as evidenced by greater similarity of stream water chemistry to muskeg water chemistry at this time. During snowmelt runoff and other storm runoff periods, water chemical properties such as ion concentration and specific conductance in streamflow were reduced to levels even lower than those attributable to muskeg water. Schwartz and Milne-Home (1982b) established the importance of muskeg control on stream discharge and chemistry when they carried out similar water chemistry studies on several creek basins in the oil sands area. The effect of draining and clearing land for the sands developments would be to generally increase major ion concentrations during summer and fall months. However, the impact of local disturbances in a larger basin would be diminished downstream as waters from affected areas are mixed with other waters. The authors concluded that the chemically based approaches for evaluating basins had provided an inexpensive technique for preliminary reconnaissance of surface water and groundwater settings, and that the techniques used should have applicability to other basins.

8.7 CHEMICAL AND BIOLOGICAL MONITORING OF MUSKEG DRAINAGE AT THE ALSANDS PROJECT SITE

Mayhood and Corkum (1981) and Mayhood et al. (1981) reported on studies carried out in 1980 to monitor the effects of muskeg drainage of the proposed Alsands oil sands extraction plant site on aquatic habitats and terrestrial vegetation, and to form the basis of a long-term program to monitor the effects of Alsands development on aquatic habitats in the Muskeg River over the life of the project. Numerous biological and water quality attributes were studied within the zones of potential impacts and in control areas. The most pronounced effects of muskeg drainage on the water quality of the Muskeg River were increases in suspended solids and turbidity especially as a result of a flood accident resulting from a breach in a ditch wall, and a temporary increase in nutrient concentrations immediately below the mine site drainage outfall caused by a fertilizer application to the ditched areas during seeding operations. Dissolved oxygen levels may have been decreased temporarily by decomposition of organic sediment contributed by the flood. However, aeration of the water was evident at the ditch outfall. Other water quality changes arising from the addition of mine site drainage were usually slight, when detectable at all.

8.8 EFFECTS OF NUTRIENT LEACHING FROM MUSKEG DRAINAGE AT ESSO RESOURCES COLD LAKE LEASE

Cross and McCart (1979) made preliminary predictions of the impact of muskeg drainage and increased phosphorus loadings on limnological conditions in a system of small lakes in the Cold Lake area. Existing hydrological information and an available empirical model were used to make the predictions. They considered that muskeg drainage would increase the phosphorus loading to the downstream lakes, but that the accompanying increase in water loading would mitigate this impact to some extent. Along with the increased phosphorus input, increases in organic matter would increase the oxygen demand in the lakes downstream of muskeg drainage areas. These lower levels of oxygen could then have detrimental effects on fish populations. Cross and McCart (1979) also concluded that the effects of muskeg drainage on lakes downstream would be roughly proportional to the volume of water drained and would probably only last as long as the water continues to be drained.

Baseline information on lakes and streams in the vicinity of Esso Resources Canada Limited's proposed heavy oil development near Cold Lake was reported by Aquatic Environments Limited (1979) and Cross (1979). As with the AOSERP baseline data, the information gathered can be useful in further analysis of water quality attributes in relation to peatlands. However, such analyses would be limited since mainly routine parameters were provided in the Cold Lake study; analyses of carbon, colour, and of minor and trace elements were not carried out. Water quality data for the Beaver River basin can also be found in Alberta Environment (1983).

8.9 BIOGEOCHEMICAL STUDY OF THE RED DEER RIVER

Baker et al. (1982) reported on a study of the biogeochemistry of the Red Deer River. The study was undertaken subsequent to observations that dissolved oxygen levels fell dangerously low during the winter months, indicating a deterioration in water quality. Contributions of organic substances by natural sources along tributaries were found to increase the chemical oxygen demand and the biochemical oxygen demand in the river. Four pathways by which organic substances may be moved from their sources to a surface watercourse were considered. The first, rainfall leaching of vegetation overhanging the river and then falling directly into the river, was considered insignificant because the extent of overhanging vegetation is very limited. The second route, transport through the unsaturated zone directly below soil surfaces, was also considered to be insignificant because the aqueous phase is not continuous thus severely limiting transport efficiency. The third pathway, overland flow, was considered to have the greatest impact on the flow regime of the Red Deer River. Its effect is virtually immediate, large volumes of water can be moved in this manner, and contact with various sources of organic substances is strong. The fourth route, downward percolation through soil to the groundwater and later discharge to the river, was considered to account for a small percentage of river flow. Sedges and grasses generally appeared to be the most likely sources of organic materials, based on similarities of amino acid ratios in source materials and water. In leaching experiments, organic soils were found to release the greatest amount of organic material to the leaching water as compared to other mineral soils. Organic soils occupy only a small portion of the river basin studied, but they nevertheless may contribute significantly to the organic carbon load of the Red Deer River because of the quantities of organic materials they can release.

8.10 HYDROCHEMISTRY OF PEATLANDS IN THE SAULTEAUX RIVER AREA

Hydrochemical research was carried out as part of a project dealing with drainage of a peatland to improve forest growth in the Saulteaux River area, east of Lesser Slave Lake (Toth 1985). The objectives of the research were to (a) determine the baseline chemical composition of the groundwater and surface water prior to installation of a drainage ditch network, (b) monitor the change in the chemical composition of the water in the area after the start of drainage operations, and (c) determine the change in water chemical composition resulting from release of reduced sulphides and organic compounds by artificial oxidation of peat.

The composition of the surface water was found to be of the Ca^{2+}-Mg^{2+}-HCO_3^- type with total dissolved solids <200 $mg{\cdot}L^{-1}$. Shallow groundwater, that is, water a few metres below the surface in the peat deposit, had TDS values some 200 to 400 $mg{\cdot}L^{-1}$ higher than the surface water. The trace metals Mn, Pb, Cu, and Zn were detected in significant concentrations in both groundwater and surface water. The concentrations ranged from <0.005 to 0.075 $mg{\cdot}L^{-1}$ for Cu, from 0.0005 to 0.012 $mg{\cdot}L^{-1}$ for Pb, and from 0.01 to 0.12 $mg{\cdot}L^{-1}$ for Zn. Levels of Mn ranged from <0.02 to over 0.05 $mg{\cdot}L^{-1}$ in groundwater, and from <0.02 to 8.2 $mg{\cdot}L^{-1}$ in the surface water. Toth (1985) suggested that the trace metal distributions result from the control on solubility by sulphide ion for Cu, Pb, and Zn. Manganese, however, is strongly absorbed on organic matter and is fairly soluble under reducing conditions. The trace metals did not occur at levels considered to be toxic, but levels of Mn were objectionably high based on water quality objectives.

Peat oxidation experiments using peat-water suspensions showed that the response of six different peat samples to exposure to oxygen was variable. In all samples, there was a decrease in pH by about one unit. In some samples, there were increases in Ca, SO_4, NO_3, and total organic carbon, indicating that the sulphide and organic matter were being oxidized. Of the trace metals, only Cu was found at significant levels in the aqueous part of the suspension.

Water samples were collected on two dates, in July and September, after the peatland drainage ditches had been excavated in late winter of 1985 (Alberta Research Council, unpublished information). The highest Pb concentration in surface water was 0.05 $mg{\cdot}L^{-1}$. Copper levels were mainly at about 0.02 $mg{\cdot}L^{-1}$, Zn ranged up 0.30 $mg \cdot L^{-1}$, and Mn levels as high as 2.5 $mg \cdot L^{-1}$ were found. Iron concentrations as high as 6.40 $mg \cdot L^{-1}$ were found in surface peat waters. The chemistry of runoff water in the main collector ditch of the drainage system is of particular interest. The pH of the water was 7.9 and 8.1 on the two sample dates. Some of the ditches within the peatland were dug through mineral soil. This effect of contact with mineral soil on increasing pH (as well as alkalinity) is consistent with observations from other regions (see Chapter 7, Section 7.3.3). Nitrate levels as high as 0.4 $mg \cdot L^{-1}$ were found in the ditch. Trace metal levels were: Cu = 0.005 to 0.02 $mg \cdot L^{-1}$, Pb = 0.004 to 0.007 $mg \cdot L^{-1}$, Zn = 0.01 to 0.09 $mg \cdot L^{-1}$, Fe = 0.5 $mg \cdot L^{-1}$, and Mn = 0.2 $mg \cdot L^{-1}$. Of these, Mn and Fe consistently exceeded the Alberta surface water quality objectives, while Zn exceeded the objectives on one of these occasions.

Results for Hg were reported as being less than 0.005 $mg \cdot L^{-1}$ which presumably is the detection limit. Thus, it is not possible to ascertain whether the Alberta water quality objective of 0.0001 $mg \cdot L^{-1}$ was exceeded. Data for water from the Saulteaux River showed that, with the exception of Fe, all the above constituents had been diluted to below the water quality objectives. Levels of Cu were slightly higher in the river water then in the collector ditch water; thus, the peatland drainage waters have a dilution effect on Cu in this situation.

8.11 WETLAND DRAINAGE AND IMPROVEMENT PROGRAM

This research program, carried out jointly by Forestry Canada and the Alberta Forest Service, was outlined by Hillman (1987) and is described in Chapter 10. Of particular relevance to water quality issues are plans to determine the effects of drainage and resulting peat decomposition on the organic chemical water quality and to investigate differences between bog and fen drainage. Some results have been reported by Hillman (1988) and are summarized in Chapter 7, Section 7.3.1.3. Peatland drainage for forestry has been shown to have various effects on water quality in other countries.

It is possible that extensive areas of peatlands could be drained for improved forest growth in the future. Therefore, it may be beneficial for agencies involved in water quality issues to monitor progress and results of water quality studies within this project and possibly to provide input to the project in the form of communicating specific concerns, advising on sampling and analysis, holding seminars and workshops, and in other ways.

8.12 PEATLAND ECOLOGY STUDIES IN THE BOREAL WETLAND REGION

A comprehensive study by Forestry Canada of the dynamics of peatlands in the Prairie Provinces to serve as background information for reconnaissance surveys has been undertaken by Zoltai and Johnson (1983). The research involved description and sampling of peat sections, vegetation, and hydrology at numerous sites in Alberta, Saskatchewan and Manitoba. Laboratory studies included determination of physical properties (moisture content, bulk density, ash content), chemical analyses (Al, Ca, Cu, Fe, K, Mg, Mn, Na, Ni, P, Pb, S, and Zn), and macrofossil analysis. Radiocarbon age, calorific value, and pyrophosphate index of decomposition were determined on selected samples. Surface water samples were also taken for chemical analysis. Publications to date include those of Zoltai and Johnson (1985, 1987) and the National Wetlands Working Group (1988).

The Forestry Canada study provides a valuable, broadly based source of baseline information on peat and peatland water chemistry and other properties. The information will be useful in the analysis of environmental aspects of peat and peat water properties and in evaluating developmental impacts on these properties according to types or classes (according to the Canadian Wetland Classification System) and geographic distribution of peatlands. Such capability will be required in any future activities involving either modelling or conventional assessments of peatland development impacts in specific geographic areas.

8.13 PEATLAND RESEARCH IN RELATION TO SOIL CLASSIFICATION AND MAPPING

Research has been conducted for a number of years at Alberta Research Council into chemical and physical properties of peat, developing criteria for Organic soil classification, and developing field and remote sensing methods for inventory. One report on a peatland inventory pilot project has been prepared (Turchenek et al. 1984), but no other information has been published. One aspect of the research involved collection of surface water samples for analysis of major constituents. Results of these analyses are presented elsewhere in this report (Chapter 6) and represent a small, baseline data set for a relatively dense set of sample points in a specific area of the province (consisting of two 625 km^2 study areas near the town of Athabasca). The water analyses, along with chemical and physical data for Organic soils, assist in characterizing different classes of peatlands. As noted in Section 8.12, such information will be helpful in routine assessments and modelling of impacts of peatland development on an areal landscape basis.

8.14 PEATLAND ECOLOGY STUDIES

A number of peatland ecology studies have been carried out since the mid-1970s at the University of Alberta, Department of Botany. Many of the studies had the objective of relating the occurrence of various peatland moss species to different environmental gradients, particularly nutrient and ionic gradients. Much of this work was referred to in Chapters 3 and 6 of this report. Analysis for major dissolved constituents in peatland surface water was required in most of these studies. The data, which can be found in theses or scientific journals, are another source of baseline information for peatlands in this province. These include Kubiw (1987), Nicholson (1987), Chee (1988), and Chee and Vitt (1989). One of the projects involved studies of the effects of acid deposition on mosses of rich fens (Rochefort 1988; Rochefort and Vitt 1989). Some of this work is reviewed in Chapter 9 of this report.

8.15 SUMMARY

Research and information sources regarding the nature of waters associated with peatlands in Alberta can be categorized as follows: (1) ongoing water surveys (Alberta Environment; Environment Canada; Alberta Forestry, Lands and Wildlife); (2) environmental impact assessments of energy projects (Athabasca oil sands area; Cold Lake region) and peatland drainage for forestry (Forestry Canada, Alberta Forest Service); (3) specific water course studies (Red Deer River); (4) peatland ecology studies (University of Alberta; Forestry Canada); and (5) studies related to land inventory and land use evaluation (Alberta Research Council; Forestry Canada; Agriculture Canada). There is limited information about peatlands and associated waters in Alberta, but current projects are adding significantly to the knowledge and information base.

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9. ACID DEPOSITION AND ITS EFFECTS ON PEATLANDS

L. Rochefort

9.1 INTRODUCTION

Anthropogenic acid deposition is a major environmental concern in North America, northern Europe, and other industrialized regions of the world. A major proportion of the total peatland area in the world also occurs in these regions. Widespread disruption of peatland ecosystems could have serious and large-scale consequences for freshwater lakes and streams.

Under natural conditions, precipitation (including both rain water and snowfall) is characterized by a pH of 5.6 (Charlson and Rodhe 1982). This pH is a result of atmospheric carbon dioxide (CO_2), which dissolves in water vapour to form carbonic acid (H_2CO_3). Carbonic acid, being a weak acid, dissociates only slightly in distilled water and would not be expected to produce a pH less than 5.6 (Likens et al. 1979). Detailed chemical analyses done by Galloway et al. (1976, 1984) revealed that acidity of rainfall is caused by the strong mineral acids H_2SO_4 and HNO_3. The authors also demonstrated that weak acids and Bronsted acids have a minimal influence on the pH (free acidity) of acid precipitation. Sulphate and nitrate are the primary anions in acidic precipitation, although chloride anions may occasionally be an important component of the total acidity in precipitation (Yue et al. 1976). By definition, acidic precipitation is rain or snow with a hydrogen ion concentration greater than 2.5 $\mu eq \cdot L^{-1}$, or a pH <5.6 (Evans 1982).

Problems arising from acid precipitation occur when: (1) there is a sufficiently large source of pollutant emission; (2) there are ecosystems or 'receptors' sensitive to the action of the pollution; and (3) climatological conditions transport the pollutant to sensitive areas (Taylor 1981).

Acid deposition arises from the emission of sulphur oxides (SO_x) and nitrogen oxides (NO_x) associated with the smelting of sulphide metal ores, the combustion of fossil fuels and other industrial processes. In Canada, two-thirds of the SO_x emissions are from several large primary smelting operations; most of the remaining SO_x emissions are derived from combustion of fossil fuels for power generation, commercial and industrial activities, and home heating. Transportation is responsible for two-thirds of NO_x emissions (Summers and Whelpdale 1976). Alberta is the largest producer of industrial acid emissions in Western Canada. Sanderson (1984) reported that the SO_2 emissions in 1981 totalled 559 691 t. The breakdown among sources was as follows: gas plants - 44.6%; oil sands plants - 25.3%; power plants - 17.8%; sour gas production facilities - 5.3%; flaring gas plants - 2.5%; and fertilizer plants, refineries, pulp and paper plants, and heavy oil recovery plants - 4.8%. Sanderson (1984) also reported that there are 377 700 t of NO_x produced annually in Alberta, making a total of just over 900 000 t of acid-forming emissions. In the following year, the total SO_2 and NOx emissions were lower at 488 297 and 353 511 t respectively (Sandhu and Blower 1986). Emission levels are expected to decrease further by the year 2000 as a result of tighter controls imposed by provincial and federal governments. Total emissions of SO_2 are expected to decrease to about 410 000 t while NOx levels are projected to decrease to about 290 000 t annually. Thus it would appear that there should be no major acid deposition problem in Alberta when considered on the basis of average deposition rates across the province. There could, however, be problems with local or regional transport of pollutants.

Once emitted into the air, SO_x and NO_x may undergo photochemical or photo-oxidation reactions in the gaseous and/or liquid phases.

Sulphur Dioxide Chemistry (Brosset 1973)

$$H_2SO_4 \xrightarrow[H_2O]{} H^+ + HSO_4^- \xrightarrow[H_2O]{} 2H^+ + SO_4^{2-}$$

concentrated acid

$$SO_2 \xrightarrow[H_2O]{} H_2SO_3 \xrightarrow[H_2O]{} H^+ + HSO_3^- \xrightarrow[H_2O]{} 2H^+ + SO_3^{2-}$$

$$HSO_3^- \longrightarrow 2H^+ + SO_4^{2-} \qquad SO_3^{2-} \longrightarrow 2H^+ + SO_4^{2-}$$

diluted acid

Nitrogen Oxide Chemistry

The conversion of NO_x to HNO_3 takes place in a series of complicated reactions during which nitrogen oxides undergo various oxidation stages to finally produce HNO_3 (Legge et al. 1980).

reactions during combustion

$$N_2 + O_2 \longrightarrow 2NO$$
$$2NO + O_2 \longrightarrow 2NO_2$$

atmospheric reactions near the emission source

$$2NO + O_2 \longrightarrow 2NO_2$$

reactions in the ambient polluted atmosphere

$NO_2 + h\nu \longrightarrow O(^3P) + NO$ (P is the quantum level)

$O(^3P) + O_2 + M \longrightarrow O_3 + M$ (M is other molecule; NO_2, SO_2)

$$NO + O_3 \longrightarrow NO_2 + O_2 + \text{energy}$$

$$O_3 + NO_2 \longrightarrow NO_3 + O_2$$

$$NO_3 + h\nu \longrightarrow NO_3 + O(^3P)$$
$$\longrightarrow NO + O_2$$

$$NO_3 + NO_2 \xrightarrow[M]{} N_2O_5$$

nitric and nitrous acids formation

$$HO_2 + NO_2 \longrightarrow HONO + O_2$$
$$HONO + h\nu \xrightarrow[M]{} HO^{\bullet} + NO$$
$$NO + NO_2 + H_2O \longrightarrow 2HONO$$
$$N_2O_5 + H_2O \longrightarrow 2HNO_3$$
$$NO_3 + H_2O \longrightarrow HNO_3 \text{ (+ other products)}$$
$$OH^{\bullet} + NO_2 + M \longrightarrow HONO_2 + M$$

Atmospheric factors such as temperature, air pressure, humidity, wind velocity, light intensity (UV), availability of aerosols, and water vapour control these atmospheric chemical reactions and the composition of the ultimate deposition (Glime et al. 1984). The transformed emissions are subsequently removed from the atmosphere and transferred to the surfaces of vegetation by dry and wet deposition processes. Dry deposition occurs continuously and includes absorption and adsorption of gases, and the impaction or gravitational settling of fine aerosol and coarse dust particles (Altshuller 1976). Wet deposition occurs intermittently and includes the transfer of dissolved and suspended materials in raindrops, snowflakes, hail, dew, and fog (Evans 1984). Depending on the atmospheric conditions at the time of release, these gases and particles may be transported over many hundreds of kilometres. Therefore, acid precipitation is no longer localized, but spreads regionally and becomes an environmental problem in areas removed from industrial centres.

Scandinavia and northeastern North America are currently receiving acid precipitation with a mean pH of 4.0 and acid rain events with pH as low as 3.0 (Bolin 1971; Likens et al. 1972). Fog events can have a pH of less than 1.5 or 2. The effects of this precipitation constitute a problem of increasing importance (Likens and Bormann 1974; Work Group 1 (MOI) 1983; US EPA 1984). In Europe, from 1955 to 1974, the pH values of rain dropped from approximately 6.5 to slightly above 4.0 (Oden 1976). In North America, the northern Atlantic coast and lower Great Lakes basin have the highest wet-deposition rates for NO_3^- and SO_4^{2-}, and the average pH value of rainfall is 4.0. The rest of North America rarely receives precipitation having a pH value higher than 5.0 except for the northern plains and Western Canada (Munger and Eisenreich 1983). Over the past decade, deposition of acidic substances has been the subject of much debate. Acid deposition has been recognized as having a great ecological impact in both terrestrial (e.g., Abrahamsen 1980; Hutchinson and Havas 1980; Overein et al. 1980) and aquatic systems (Harvey et al. 1981; Schindler et al. 1980a, 1980b; Schindler and Turner 1982; Schindler et al. 1985, 1986). Unfortunately, little research has been conducted on the ecological effects of acid deposition upon peatlands and our understanding of their vulnerability is limited (Gorham et al. 1984).

The concept of acidification of peatlands is complex because peatlands are naturally acidic. The processes involved in natural and anthropogenic acidification have to be distinguished in order to assess the ecological impacts of air pollution on peatlands. The purpose of this chapter is to review the literature on the effects of deposition of acidic and acid-forming substances on peatland ecosystems. The chapter is divided into two parts: Section 9.2 describes natural acidification processes occurring in peatlands, including biogeochemical interactions and plant community changes along gradients of acidity; Section 9.3 summarizes the effects of anthropogenic acidification on (1) peatland vegetation, (2) water chemistry, and (3) wetland ecosystems.

9.2 NATURAL ACIDIFICATION OF PEATLANDS

9.2.1 Biogeochemical Interactions

The acidic environment of many mire habitats is attributable to several biogeochemical processes (Figure 9.1). This section reviews the different processes that produce acidity in peatlands. Vegetation as a producer of acidity will be examined

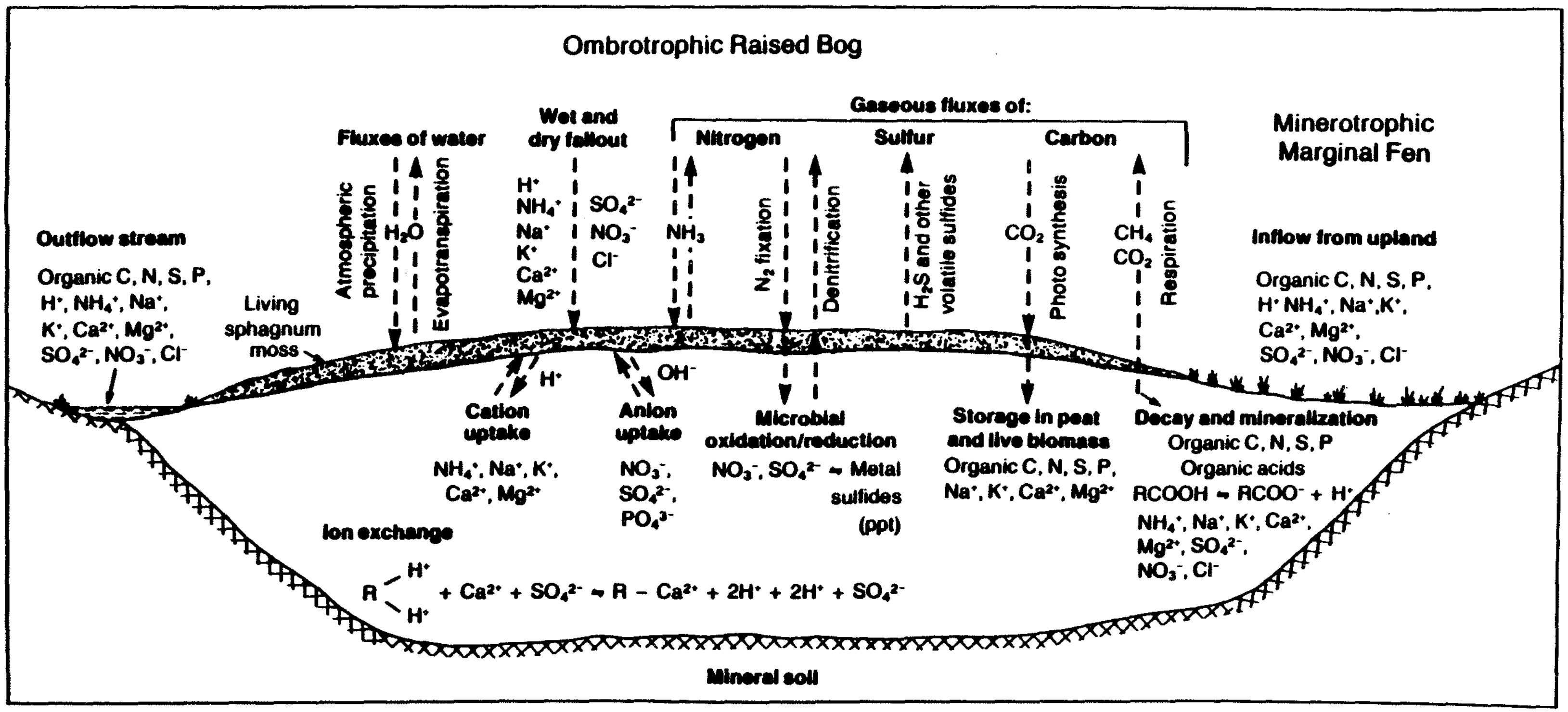


Figure 9.1. Ecosystem processes and the pathways of chemical inputs and outputs in an idealized peatland. (Source: Gorham et al. 1984).

first, followed by a discussion of biogeochemical interactions as causes of acidity.

9.2.1.1 Cation-exchange. *Sphagnum* species are prominent wetland plants with the capability to direct succession by acidification (Andrus 1986). The general restriction of *Sphagnum* to nutrient poor and acidic sites is partly due to *Sphagnum* playing a role in creating these conditions. Bellamy and Rieley (1967) give an example where the acidifying action of the hummock-forming *Sphagnum* creates a much more acidic environment than in the surrounding water. It has long been known that *Sphagnum* can modify surrounding water (Skene 1915) by making it more acidic. Williams and Thompson (1936) and Ramaut (1954) reported that cations from solution were taken up by the plants and exchanged with H^+. Later Clymo (1963, 1964) indicated a strong correlation between the cation-exchange capacity (CEC) of *Sphagnum* species and their content of unesterified polyuronic acid (see also Theander 1954, Knight et al. 1961 and Spearing 1972). Plant CEC and uronic acids are related to the dryness of the microhabitat; species growing highest above the water level on hummocks have the highest uronic acid contents and cation-exchange capacity (Clymo 1987; Vitt et al. 1975b). Because uronic acids are found in the holocellulose fraction of the cell wall of *Sphagnum* (Craigie and Maas 1966), most of the cation exchange properties persist after death. In general, *Sphagna* have higher exchange capacities than other plants of oligotrophic habitats, including true mosses (Brown and Bate 1972; Clymo 1963).

If neutral water is sprayed over a *Sphagnum* plant, dead or alive, the water becomes more acid. This is explained by the fact that uronic acids occur in the cell wall of *Sphagnum* in free acid form (i.e., -COOH) (Clymo 1967). Hence, cations from the environment (Ca^{2+}, Mg^{2+}, K^+, Na^+, etc.) replace the H^+ ions from the exchange sites and these produce a decrease in water pH as they are slowly leached by rain and flowing water. Indeed, Clymo hypothesized that the ability of *Sphagnum* to produce acidity depends on plant growth rate and the production of new -COOH groups. With realistic values of rainfall, concentration of cations in rain, and *Sphagnum* growth rates, Clymo (1967) found that a mean pH of about 4.2 could be maintained in an unpolluted atmosphere. In an example of this theory, calculations for a Moor House peatland (Clymo 1984) showed that *Sphagnum* contributed about half of the measured acidity. The rest would have had to come from other sources.

9.2.1.2 Polygalacturonic acid. The reduction of environmental pH by *Sphagna* is considered to be an adaptive metabolic (internal) process rather than a passive chemical process (cation exchange) over which the moss has little control (Kilham 1982). Kilham suggested that the dissociation of polygalacturonic acid (PGA) inside the moss, not the production of hydrogen ions by cation exchange, is of primary importance in determining ambient pH. Further, the more likely regulating mechanism of pH is the PGA actively produced by the plants in response to a need for acidification. This hypothesis has been challenged and is discussed by Andrus (1986).

9.2.1.3 Net biological uptake of nutrient ions. Nutrient uptake (Ca^{2+}, K^+, NH_4^+, NO_3^-, etc.) can affect the acidification processes of a bog ecosystem (Kilham (1982). Owing to the principle of electroneutrality, charge balance must be maintained between plants and their environment (Reuss 1977). Thus, when a plant takes up cations in excess of anions, other cations (usually H^+) are released into the environment. Similarly the same mechanism occurs with anion nutrients and anions such as hydroxide (OH^-) are produced (Reuss 1977). Nilsson et al. (1982) called this process 'cation excess', which they define as the excess of cation uptake over anion uptake.

In practice the nutrient ion balance is calculated directly from the mass budget for the bog ecosystem as a whole. Therefore, if the sum of the cations exceeds the sum of anions, the acidity of the system will increase because H^+ production will overcome OH^- production. On the other hand the system will become more basic if

OH^- production exceeds H^+ production (Kilham 1982). Reuss (1977), describing the soil-plant system in general, suggested that fluxes of H^+ from plant uptake processes are much greater than rainfall H^+ inputs, thus complicating interpretation of rainfall effects. Further investigations are needed in peatlands to evaluate the importance of this process.

9.2.1.4 Carbonic acid. Villeret (1951) claimed that high concentrations of CO_2 found in bog water (carbonic acid) were the principal chemical factor responsible for mire acidity. But experimental evidence (Clymo 1987; Gorham 1956) has discounted this theory. Clymo (1987) showed that CO_2 in water is a secondary, not a primary, source of acidity. There were also detailed physico-chemical objections that high CO_2 concentration in water would maintain a low pH.

9.2.1.5 Oxidation and reduction of sulphur and nitrogen compounds. Another possible source of acidity is the supply and interconversion (redox processes) of sulphur compounds. As mentioned by Clymo (1984) and Hemond (1980, 1983), there is active production of H_2S at and below the water table in peat bogs. A generalized equation for sulphate reduction by bacteria is as follows (Nriagu and Hem 1978):

$$SO_4^{2-} + 2(CH_2O) \rightleftharpoons H_2S + 2HCO_3^-$$

This H_2S may be oxidized, after upward diffusion or a drawdown of the water table, to SO_4^{2-} with companion of $2H^+$. Gorham (1956, 1967) was the first to suggest this as an explanation of relatively acid water with high SO_4^{2-} concentration in pools at Moor House during dry weather.

Kilham (1982) also described the bacterial oxidation of pyrite and elemental sulphur as the producer of acidity in bog ecosystems. Production of sulphuric acids would occur according to the following generalized reactions (Wetzel 1975):

$$FeS_2 + 15/4O_2 + 7/2H_2O \rightleftharpoons Fe(OH)_3 + 2SO_4^{2-} + 4H^+$$
$$2S^{\circ} + 3O_2 + 2H_2O \rightleftharpoons 2H_2SO_4$$

Soil nitrogen (N) transformations involve production and consumption of H^+ ions (Reuss 1977). The oxidation of ammonium to nitrate by aerobic chemautotrophic bacteria results in the production of acid:

$$NH_4^+ + 3/2O_2 \xrightarrow[\text{Nitrosamonas}]{} 2H^+ + NO_2^- + H_2O$$

$$NO_2^- + 1/2\ O_2 \xrightarrow[\text{Nitrobacter}]{} NO_3^-$$

Assuming subsequent plant uptake of NO_3^-, part of the acidity produced would be neutralized. Therefore, ammonium contained in rainfall is another component to consider when studying the effects of acidification on peatlands.

9.2.1.6 Organic acids. Two processes of organic acid production may be distinguished: the release of organic acids by excretion from living plants (Ramaut 1955a,1955b), and the production of humic and fulvic acids by decomposition of vegetation (McKnight et al. 1985; Perdue and Lytle 1983). Organic acids are generated by incomplete oxidation of decomposing organic matter as indicated below:

$$(CH_2O)n + O_2 + H_2O \rightleftharpoons RCOOH$$
$$RCOOH \rightleftharpoons H^+ + RCOO^-$$

Relatively little acid is excreted by growing *Sphagnum* (Clymo 1967) but large amounts may be produced during decomposition (Clymo 1984). Organic acids can make significant contributions to water acidity in given swamps and bogs with pH as low as 3.0 to 3.4 (Moore and Bellamy 1974; Oliver et al. 1983; Veery 1975). Recent work has shown that the production of organic acids was the dominant source of acidity for a large array of bogs examined in North America, Ireland, and England (Gorham et al. 1984; Hemond 1980; Urban et al. 1987). Hemond (1980) indicated that production of weak organic acids is within subsurface interstitial waters, where anaerobic conditions are prevalent.

The chemistry of organic acids in surface waters is different from that of mineral acids (Jones et al. 1986). Two important differences are:

1. Organic acids can be important pH buffers for waters in the pH 4.5 to 5.5 range; and
2. Organic ligands chelate metals such as Al and as a consequence increase their solubility.

The property of chelation of these organic acids can protect peatland ecosystems by substantially reducing the toxicity of metals such as Al.

9.2.2 Succession

In humid climates where precipitation exceeds evaporation and where decomposition is slower than production, processes of peat formation allow development of raised bogs. Slowly the peat surface is isolated from groundwater influences, and changes in water chemistry and vegetation occur. As a consequence, all the natural acidification processes described above increase the acidity of peatlands. Gorham et al. (1987) enumerated those changes as the following:

> As fens are converted into bogs pH declines sharply because of alteration in local hydrology. Removal of hydrologic inputs to a peatland from adjacent or underlying mineral soils cuts off the supply of bases that neutralize polyuronic acids produced by *Sphagnum* (Clymo 1987) and the complex, yellow-brown organic acids produced by decomposition of plant remains (Urban et al. 1987). Moreover, retardation of flow consequent upon removal of such hydrologic inputs allows organic acids to reach higher concentrations before being flushed from the peatland; surface waters of acid bogs are often distinctly more tea-coloured and higher in dissolved organic carbon (DOC) than water of circumneutral fens (Glaser et al. 1981).

To date, there are no records of the actual change in pH as a fen is transformed into a bog, although the complete transformation of 1 m^2

of extremely rich fen to bog has been proven possible (Bellamy and Rieley 1967).

Once initiated, the transformation of poor fen to acid bog is believed to be rapid as indicated by a bimodal pH frequency distribution for the vast peatlands of northern Minnesota, northern Swedish peatlands, and central Alberta peatlands (Figure 9.2). To explain this bimodal distribution, Gorham et al. (1987) suggested:

> As declining minerotrophic inputs of alkalinity are titrated away by organic acids from the decay process, acidity shifts quickly from a pH around 6 to a pH around 4, so that sites with intermediate pH values are relatively scarce.

The rapid shift in pH from between 5 and 6 to about 4 can be seen in the relationship between pH and alkalinity in peatland waters (Figure 9.3). From these observations, Gorham et al. (1984) considered poor fens as being the peatland ecosystems most sensitive to acid deposition.

9.2.3 Plant Community Succession Along Acidification Gradients

Spatial analysis of peatland flora in northern and western Alberta (Horton et al. 1979; Slack et al. 1980; Vitt et al. 1975a), revealed marked changes in the nature of plant communities along natural gradients of acidity in peatlands. In that region, indicators of typical rich fen species include: *Tomenthypnum nitens*, *Scorpidium scorpioides*, *Drepanocladus revolvens*, *Sphagnum warnstorfii*, *Campylium stellatum*, *Utricularia intermedia*, *Calliergon trifarium*, *Tofieldia glutinosa*, *Drosera anglica*, and *Muhlenbergia glomerata*. In transitional fens, the mires are more likely to be dominated by *Drepanocladus vernicosus*, *D. adnuncus*, *D. polycarpus*, *Brachythecium mildeanum*, *Potentilla palustris*, *Carex diandra*, and *C. lasiocarpa*. In poor fen conditions, many of these plants disappear and are replaced by species such as *Tomenthypnum falcifolium*, *Drepanocladus exannulatus*, *Sphagnum jensonii*, *S. angustifolium*, and *Rubus chamaemorus*. Finally, in truly ombrotrophic conditions species like *Sphagnum fuscum* and *S. nemoreum* dominate the peatland.

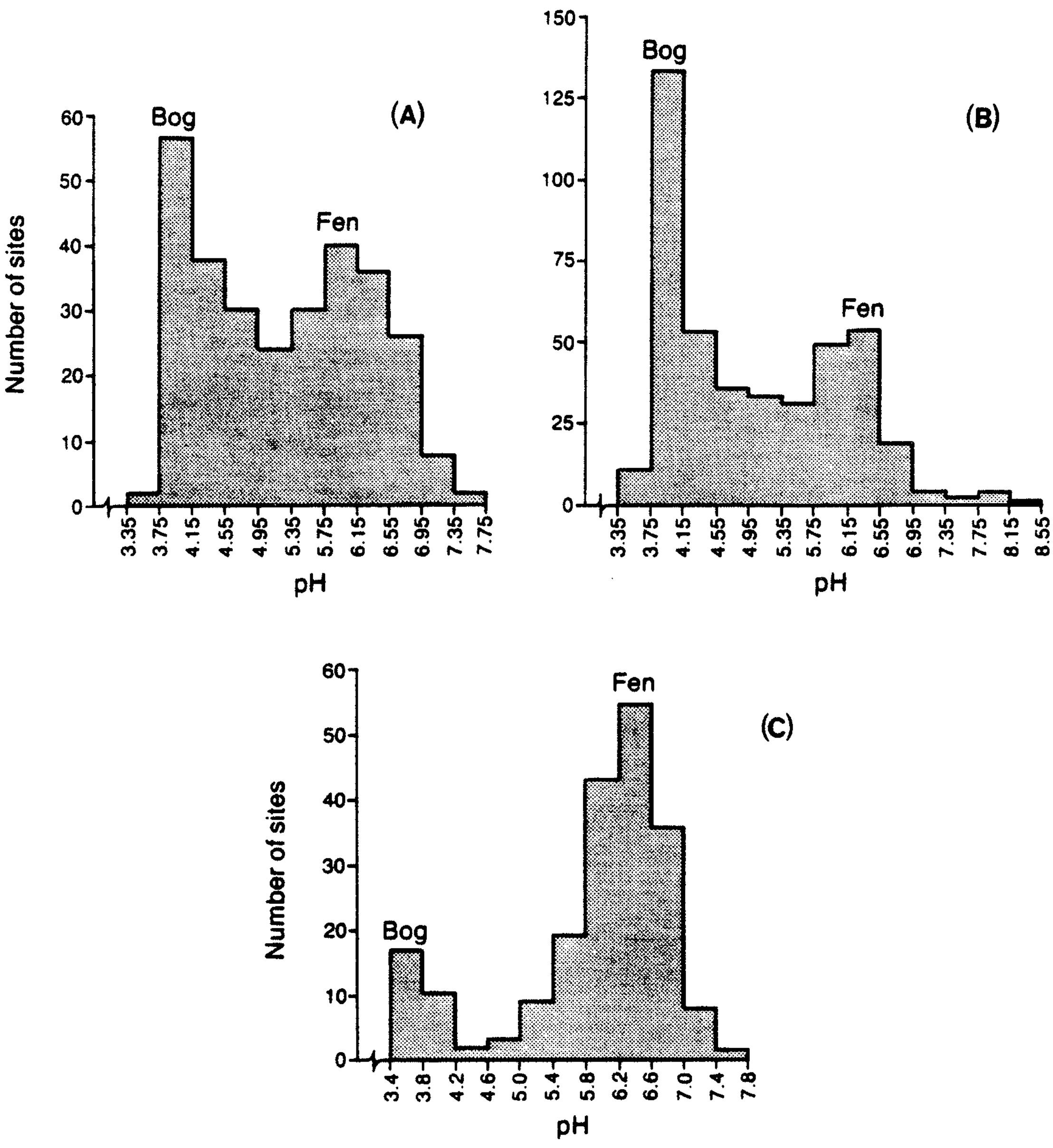


Figure 9.2. Frequency distribution of pH in surface waters collected from different plant communities of peatlands in (A) northern Minnesota, (B) northern Sweden, and (C) north-central Alberta. ((A) and (B) adapted from Gorham et al. 1984 and (C) from Turchenek et al. 1984).

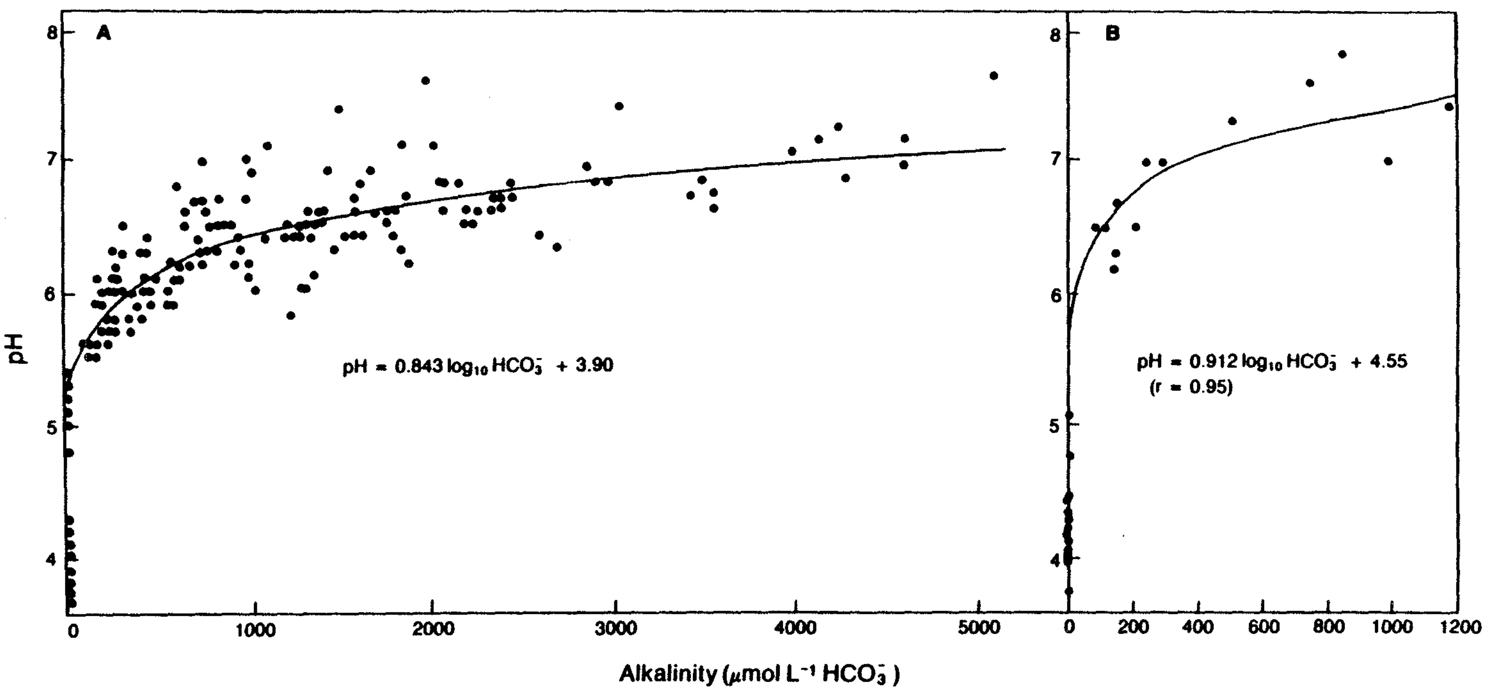


Figure 9.3. Relationship between pH and alkalinity in surface waters from fen and bog sites in (A) north-central Alberta and (B) the English Lake District. ((A) adapted from Turchenek et al. 1984 and (B) adapted from Gorham et al. 1984).

In similar studies on peatlands in northern Minnesota, Gorham et al. (1987) showed that rich fen species (e.g., *Carex leptalea*, *Triglochin maritima*, *Scorpidium scorpioides*, and *Sphagnum warnstorfii*) will be replaced by species characteristic of intermediate conditions of pH and calcium concentration (e.g., *Eriophorum tenellum*, *Potentilla palustris*, *Calliergon stramineum*, *Sphagnum* Section *Subsecunda*). Eventually the latter species will be displaced by typical bog species (e.g., *Eriophorum spissum*, *Carex pauciflora*, *Sphagnum capillifolium* = *S. nemoreum*). Hence, as fens are transformed into bogs, species richness declines.

Chronological changes in plant communities are also observed in peat cores. Stratigraphic profiles in a peatland record chemical changes to some degree. New developments in quantitative analyses of bryophyte stratigraphy in peat deposits permit reconstruction in some detail of the transformation of circumneutral fen into acid bog (Janssens 1983; Gorham et al. 1987; Janssens and Glaser 1986; Janssens 1989).

9.3 ANTHROPOGENIC ACIDIFICATION

Acid precipitation has not been mentioned yet as a source of acidity in bog water. Sulphuric mineral acids were determined as the primary acid form in wetlands near Sudbury, Ontario (Glooschenko and Stevens 1986) and in Belgium (Vangenechten 1981). Gorham et al. (1985) showed that under severe acid deposition in northern England, the acidity of bog waters was dominated by strong mineral acids. The concentration of sulphuric acid deposited from the atmosphere was higher than the coloured organic acids. Earlier, Gorham (1958) had noticed that acidity of bog surface waters increased as one approached industrial centres in Britain. What really is the effect of acid precipitation, a relatively new acid component, on the dynamics of peatlands?.

The purpose of this section is to review the literature dealing with possible effects of anthropogenic acidic substances on peatlands. The first part focuses on the effects of acidic deposition on vegetation - especially bryophytes. The second part deals with chemical changes with regard to sulphur and nitrogen as affected by acid rain. Finally, the third part examines acid deposition effects at the ecosystem level.

9.3.1 Bryophyte Sensitivity to Air Pollution

There is some evidence that the dominant vegetation of peatlands, the bryophytes, may be more vulnerable to acid precipitation than vascular plants. The leaves of bryophytes are one cell thick, have no stomates, and have almost no protective cuticle (Proctor 1979). Because of this direct contact with the atmosphere, most terrestrial bryophytes are classified as being poikilohydric in that they absorb required nutrients (Flint and Gregory 1969) and moisture from the atmosphere. They have also been shown to have a high cation exchange capacity in all parts of the plant (Spearing 1972; Brown and Buck 1978a, 1978b, 1979; Buck and Brown 1978). Most mosses have poorly developed internal conducting tissue and external ectohydric capillary movement of liquid occurs along the stems and leaves. Whorled and overlapping leaf arrangements, as well as thick cell walls, are morphological adaptations for precipitation interception and moisture conservation. These morphological characteristics, highly evolved for direct response to the atmosphere in order to directly absorb nutrients and moisture, result in high susceptibility to acid precipitation injury (Glime et al. 1984) .

Lichens were first recognized and used as bio-indicators of atmospheric pollution in France and Britain (Grindon 1859; Macmillan 1861; Nylander 1866). During the late 1960s a very strong correlation between the distribution of lichens and levels of air pollution was established (for a review, see LeBlanc and Rao 1974). As the concept of bio-indicator species evolved, mosses were used successfully as indicators of air quality. Subsequently, bryophytes were used more frequently for environmental studies (Gilbert 1970a, 1970b; Kauppi and Mikkonen 1980; Stefan and Rudolph 1979; Sundstrom and Hallgren 1973). For reviews of the use of mosses and lichens as bio-indicators see LeBlanc and Rao (1974), Nash (1976) and Richardson and Nieboer (1981). In Canada, bio-monitoring studies using bryophytes have been conducted by LeBlanc and DeSloover (1970), LeBlanc et al. (1972a, 1972b) and LeBlanc et al. (1974). In Alberta, studies of bryophytes and lichens as bio-indicators were conducted by Addisson and Puckett (1980), Case (1980, 1984), Kennedy et al. (1985), and Skorepa and Vitt (1976). Recently, direct evidence of acid rain effects on lichens has appeared in the literature (Fritz-Sheridan 1985; Gilbert 1986; Lechowicz 1982). With time, the techniques of bio-monitoring have become more refined. Examples include: using moss bags (Temple et al. 1981); live tissue analyses to determine metallic and nonmetallic elements (Pakarinen 1981); and, in situ sampling of ombrotrophic mosses and lichens to avoid soil contamination (Glooschenko et al. 1981; Schell 1986).

The physiological responses of bryophytes sensitive to SO_2 and other air pollutants has been studied (Winner and Bewley 1978a, 1978b; Winner and Koch 1982), but similar responses to acid precipitation has not been studied extensively. The next subsections will review what is known about wet acid deposition effects on peatland vegetation. Since little information is available on the subject, it is pertinent to first present the findings of experimental acidifications performed on mosses in the boreal forest (Section 9.3.1.1). The second subsection (9.3.1.2) will describe an actual experimental acidification of a whole *Sphagnum*-dominated peatland. This is the only study dealing directly with the effects of acid deposition on wetland ecosystems in North America. A research group in Britain has also been investigating the direct effects of air pollution and associated acid deposition upon peatlands in northern England. Their studies will be related in the third subsection (9.3.1.3). Finally, the last subsection (9.3.1.4) will report on the occurrence of *Sphagnum* mosses in response to acidification of open water. So far, no research group has investigated the effects of acidic dry deposition on peatland vegetation due to technical restrictions in quantifying dry deposition.

9.3.1.1 Acid deposition on boreal mosses. Hutchinson et al. (1987) sprayed simulated acid rain (2:1 molar ratio of sulphuric to nitric acid) on boreal forest feather moss (*Pleurozium schreberi*) and lichens (*Cladina* spp.) over a period of five growing seasons. The feather moss community was most deleteriously affected (moss coverage, growth, chlorophyll content) by sprays of pH 2.5 and 3.0, but was also affected by sprays of pH 3.5; lichens were less sensitive. In laboratory experiments, sprays containing sulphuric acid were more detrimental than nitric acid or water sprays of pH 5.6. While ambient rains of pH 4.0 to 4.5 may not be harmful to the boreal mosses, it is apparent that occasional extreme acid rain events may be deleterious.

Glime et al. (1984, 1986) studied laboratory and field responses of five boreal bryophyte species (*Dicranum polysetum*, *Hylocomium splendens*, *Pleurozium schreberi*, *Ptilium crista-castrensis*, and *Rhytidiadelphus triquetrus*) to acidic wet deposition. Total chlorophyll concentrations decreased in relation to an increase in H^+ concentration of acid solution. Once again, sulphuric acid treatments were more toxic to the bryophytes than nitric acid treatments. Further, the additions of HNO_3 increased chlorophyll if the pH was not extremely low. These results were not surprising as nitrogen is a limiting nutrient. In short-term experiments, their data suggested that episodes of acid rain can adversely affect chlorophyll content of mosses. At a pH of 2.5, moss chlorophyll was reduced in the 2 minutes following moist conditions. However, the authors gave no indication of the ability of the moss to recover from these short episodes. In long-term experiments, plants sprayed at pH 2.5 to 3.5 indicated signs of chlorosis and the lower stem parts became necrotic, and pH 2.5 caused cessation of growth.

9.3.1.2 Experimental acidification of a peatland. Experimental acidification of a *Sphagnum*-dominated peatland at Experimental Lakes Area (Ontario) began in 1983 (Bayley et al. 1986). The peatland is classified as a poor fen (Vitt and Bayley 1984). As discussed earlier, the poor fen is considered the most sensitive type of peatland in regard to acid precipitation (Gorham et al. 1984).

The acidification was carried out by irrigating the fen with solutions that simulate the acidic precipitation of eastern Canada and United States; i.e., pH 4 (Munger and Eisenreich 1983). The spray was composed of 50% nitric acid - 50% sulphuric acid (molar ratio) based on the projection of Brimblecombe and Stedman (1982). Once a month, during the growing season (May to October), 2.7 ha of the fen were irrigated with acid plus lake water using a pipe distribution network to deliver water to sprinkler heads. The acid was sprayed on the fen at pH 3 to reduce the volume of lake water applied to the fen and to keep the experimental disturbance of the hydrologic and cation budgets as low as possible. However, the acid loading to the site annually was equivalent to reducing the mean pH of precipitation from 4.9 to about 4, a value typical of northeastern American rainfall. The unacidified portion of the site served as a control and received an equal volume of water without acid. Further details about the experimental design are found in Bayley et al. (1986).

Three dominant *Sphagnum* species were monitored for decomposition and growth. *S. angustifolium*, *S. fuscum*, and *S. magellanicum* were the plant materials studied. Initial years of acidification resulted in *Sphagnum* fertilization and peat accumulation (Bayley et al. (1986). Similar results were obtained in a study of *Sphagnum angustifolium*, *S. fuscum*, *S. magellanicum*, *Tomenthypnum nitens and Scorpidium scorpioides* by Rochefort (1987). In a rich fen in central Alberta, *Tomenthypnum nitens and Scorpidium scorpioides* were treated under natural conditions in 1986 with simulated acid rain prepared in a 1:1 mixture of sulphuric and nitric acids at pH 3.5. Significant increases in growth and chlorophyll b content in *Tomenthypnum nitens* were observed in the acidic treatments, but growth and chlorophyll content of *Scorpidium scorpioides* were unaffected. In a poor fen in northwestern Ontario, decomposition of the three *Sphagnum* species indicated above was not affected by 3 years of simulated acid rain, but production was promoted (Rochefort 1987; Rochefort and Vitt 1988).

9.3.1.3 British investigations. British studies have provided evidence for the modification of mires by atmospheric pollutants. *Sphagnum* species are now largely absent from the blanket bog vegetation of the southern Pennines where in the past, they were the dominant peat-formers (Moss 1913; Pearsall 1950; Tallis 1964). The disappearance of *Sphagnum* plants in that area is correlated with the atmospheric pollution caused by the Industrial Revolution of the last 200 years. Today only S. *recurvum*, which has a wide ecological amplitude, is widespread and abundant. *Sphagnum imbricatum* occurs in abundance as sub-fossil remains in peat mires throughout the British Isles but is rare today (Green 1968). The exact conditions under which it thrived in the past remain an enigma.
Ferguson et al. (1978) and Ferguson and Lee (1979, 1980) have shown that the same *Sphagnum* species studied by Tallis (1964) are susceptible to both acid rain and sulphur pollutants. These workers measured a range of concentrations of HSO_3^-, SO_2, and SO_4^{2-} that caused a diminution in chlorophyll content, inhibition of photosynthetic carbon fixation, inhibition of growth and, in some cases, death of the mosses. Low concentrations of these chemicals stimulated growth. They ascertained that the SO_2 fallout and its derivatives could explain the disappearance of *Sphagnum* species from the mires.

Transplants of *Sphagnum* from a site away from industrial influence to the southern Pennines barely survived or died (Ferguson and Lee 1983a). Since mean annual concentrations of SO_2 in the southern Pennines have fallen in the past decades (Ferguson and Lee 1983b), SO_2 toxicity alone is unlikely to account for the paucity of *Sphagnum* or for the failure of transplants. Ferguson and Lee

(1983a) concluded that the detrimental effects were the result of an interaction between present atmospheric nitrogen and sulphur pollutants (components of acid rain) with the legacies of previous pollution episodes which are accumulated in the surface peats. With the increasing importance of the concentrations of nitrate in both air and rain in UK (Salmon et al. 1978, cited by Press et al. 1986), the research group turned their interest to increased atmospheric nitrogen supply. They first demonstrated that *Sphagnum* species are adapted to nitrate-nitrogen and presented evidence suggesting that ammonium inhibits the development of nitrate reductase enzyme (Press and Lee 1982). However, low concentrations of both ions (NH_4^+ and NO_3^-) are not detrimental. In a subsequent study, Woodin et al. (1985) demonstrated a close coupling of the *Sphagnum fuscum* plants with their atmospheric nitrogen supply at a remote site in subarctic Sweden, using the substrate inducible enzyme nitrate reductase. Nitrate was utilized principally by the *Sphagnum* carpet but not by the higher plants rooted in the peat. This close relationship between the 'natural' deposition of an ion in precipitation and the physiology of an ombrotrophic bryophyte species further emphasized the likely susceptibility of *Sphagnum* species to perturbation of the atmospheric environment by gaseous pollutants or their solution products (Woodin et al. 1985).

Ferguson et al. (1984) transplanted five *Sphagnum* species from an unpolluted site in North Wales to a polluted site in the southern Pennines. Tissue analyses showed changes in element concentrations, particularly total nitrogen, within a few months. In another project, however, concentrations of nitrate and ammonium within the range measured in southern Pennine bulk deposition failed to stimulate and even reduced the growth of *Sphagnum cuspidatum* in laboratory experiments (Press et al. 1986; Lee et al. 1987). German workers examined the influence of NH_4^+-N and NO_3-N on *Sphagnum magellanicum* cultivated under defined conditions in phytotrons (Rudolph and Voigt 1985). They established the nitrate and ammonium concentrations most favourable to the plants (100 μM NO_3^-; 95 μM NH_4^+)

as well as those levels which were deleterious to growth and metabolism of *Sphagnum magellanicum*. In a review of the responses of ombrotrophic mires to acidic deposition, Lee et al. (1987) pointed out the increasing importance of the present-day nitrogen supply from the atmosphere. Transplants of *Sphagnum* mosses, in a nitrogen-rich atmosphere in the southern Pennines, showed poor growth, rapid and massive accumulation of nitrogen in their tissue, and marked change in nitrogen enzyme activity. These results suggest that nitrogen deposition as well as sulphur pollutants can affect the growth and metabolism of ombrotrophic *Sphagnum* species and hence the ecology of mires. In conclusion, Lee et al. (1987) emphasized that for further studies it will be important to consider interactions between sulphate, nitrate, and ammonium ions on the growth of *Sphagnum* species and the ecology of ombrotrophic mires.

9.3.1.4 European investigations. In some lakes of Sweden, acidification has led to progressive 'oligotrophication' (Haines 1981; Cowling and Linthurst 1981; Cowling 1982), where macrophyte communities dominated by *Lobelia* and *Isoetes* have regressed and been replaced by *Sphagnum*-dominated communities during the last few decades (Grahn et al. 1974; Grahn 1986). Grahn (1977) showed a correlation between pH and *Sphagnum* occurrence. He stated that the *Sphagnum* appearance in lakes was correlated with the deposition of airborne acid substances. The increased growth of *Sphagnum* is probably a response to improved habitat conditions, but it is quite possible that if the mat continues to grow, the acidifying properties of the *Sphagnum* could take over and dominate. This phenomenon of occupation by dense *Sphagnum* mats on bottom areas has further consequences for the lake ecosystem including reduced nutrient cycling and lowered aquatic productivity. According to Grahn et al. (1974), several ions, most of which are important for biological production, are bound to the moss tissue, due to its strong ion exchange capacity, and thereby withdrawn from other organisms. Furthermore, the life conditions for many members of the bottom fauna deteriorate, because the moss is very poor substratum for these organisms. Takeover by *Sphagnum* has been reported once in North America at Lake Golden in New York State by Hendrey and Vertucci (1980). Similarly, mats of *Drepanocladus fluitans* and *Leptodictyum riparium* (acidophile bryophytes) have been observed to cover lake bottoms (Gorham and Gordon 1963).

In the Netherlands, there has been a dramatic decline during the last 30 years in some macrophyte communities in heathland vegetation (Roelofs 1983) and oligotrophic soft waters (Shuurkes et al. 1986). The observed changes in macrophytes appear to be caused by airborne sulphur and nitrogen deposition (Roelofs et al. 1984). Roelofs (1986) concluded that: (1) acidification of oligotrophic, poorly buffered heathland soils as a result of acid precipitation leads to a decline in the number of plant species; (2) acidification of water above a carbonate-free sediment leads to disappearance of all submerged macrophytes due to lack of carbon dioxide; and (3) acidification of a water body above a sediment containing little carbonate, as a result of ammonium-containing precipitation, leads to suppression of the isoetid plant community by luxuriant growth of *Juncus bulbosus* and/or *Sphagnum* species as a result of increased carbon dioxide and ammonium levels in the water layer.

A study of old (ca 1920) and recent (1978) diatom assemblages from 16 Dutch moorland pools showed that acidification of humic poor moorland pools reduces diatom diversity (Van Dam et al. 1981).

In Belgian bog lakes, acid sulphur-rich deposition (wet and dry) lowered the pH and increased the sulphate concentrations (Vangenechten and Vandergorght 1980; Vangenechten et al. 1984). Furthermore, the water of those moorland pools was unfavourable for most freshwater fish, and algal communities were impoverished.

9.3.2 Acid Deposition Effect on the Biogeochemistry of Peatlands

Surface bog waters from Manitoba to Newfoundland have lower concentrations of nitrate and sulphate than those in atmospheric precipitation despite the concentrating effect of evapotranspiration (Urban et al. 1987). Urban et al. (1987) also noticed a subtle shift in organic versus mineral acidity produced, with the importance of mineral acidity increasing eastward along a transect. These findings indicate clearly that acid rain components are interacting with peatland ecosystems. Nitrate and sulphate relationships within peatlands are documented in the following section. Peatlands as potential sources of acidity for aquatic ecosystems are discussed in a subsequent section.

9.3.2.1 Nitrogen dynamics. Nitrogen has been characterized as playing a dual role in the ecology of *Sphagnum* bogs (Urban and Bayley 1986). Nitrogen is often considered as a limiting nutrient in many North American bogs (Moizuk and Livingstone 1966; Tilton 1978, cited by Urban and Bayley 1986; Watt and Heinselman 1965). On the other hand, the supply of nitrogen from precipitation in Britain is known to damage *Sphagnum* and probably prevents many bryophyte species from recolonizing mires (Lee et al. 1987). Nitrate is efficiently taken up within peatlands (Moore 1978; Hemond 1983), except for the polluted British sites (Lee et al. 1987). The data of Bayley et al. (1986) also showed that peatlands are an efficient sink for nitrate in an experimentally acidified peatland. Nitrate throughout the year was undetectable in surface water and outputs of nitrate were only 0.4% of inputs. The experimentally added nitric acid at the Experimental Lakes Area (ELA) increased NO_3^- in the surface water from <0.5 $\mu eq \cdot L^{-1}$ to 7-71 $\mu eq \cdot L^{-1}$ during and just after irrigation (Figure 9.4). Twelve to 24 hours after irrigation, the NO_3-N concentration in both areas returned to background concentrations. The major mechanism of uptake is not known.

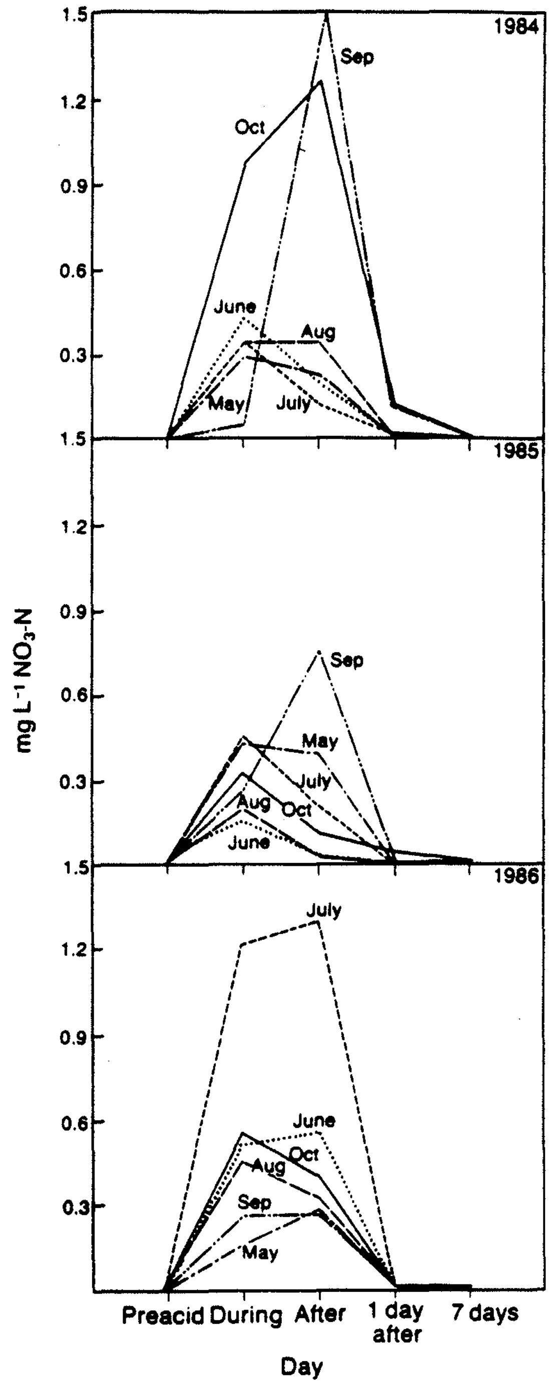


Figure 9.4. Concentration of NO_3-N in surface water of an oligotrophic experimental area before and after acidification experiments. Samples collected prior to acidification (Preacid), 2 hours after the start of acidification (During), just after (After), 1 day after and 7 days after acidification. (Source: S.E. Bayley, University of Alberta, 1987, personal communication).

Urban and Bayley (1986) proposed three possible mechanisms of nitrogen transformation in peatlands: denitrification, assimilation by plants, and dissimilatory reduction to ammonium. Bayley et al. (1987) observed little increase in ammonia in their water chemistry, indicating that the process of dissimilatory NO_3^- reduction is probably negligible. In contrast, Hemond (1983) suggested that dissimilatory reduction to ammonium may be important in a Massachusetts bog. No direct measurement of this process has yet been made in peatlands. Background rates of denitrification are very low at Marcell bog (Minnesota) and the ELA site (Urban and Bayley 1986) and no denitrification response to the acid application at ELA has been detected. These later results suggest that plant uptake is probably the major sink for NO_3 deposited by rainfall.

Vascular plants in acidic habitats, chiefly *Ericaceae*, have a restricted ability to utilize nitrate (Havill et al. 1974). *Sphagnum* species have been shown to utilize nitrogen in the NO_3^- form. In a field experiment, Bayley et al. (19867) demonstrated that the nitrate in an acid spray was removed during the time the water flowed from the top of the *Sphagnum* mat to a funnel beneath the mat; i.e., within 10 cm (see also Urban and Bayley 1986). These field results were confirmed by laboratory experiments with nitrate uptake. Three *Sphagnum* species were soaked in acid spray medium (668 μeq) in glass jars (Bayley et al. 1987). Within 20 hours, 100% of the nitrate from the water was removed (Figure 9.5). Similar results were obtained with brown stems (i.e., with non-living plant material), and therefore passive ion exchange was hypothesized as an important mechanism controlling this uptake.

The uptake of NO_3^- by vascular plants results in the release or exchange of OH^- ion (Reuss 1977), suggesting that H^+ from nitric acid in rainfall would be neutralized by the subsequent uptake of NO_3^-. Studies of NO_3^- uptake with *Sphagnum* are needed to evaluate if the latter process also occurs in peat. At present, the above studies suggest that the deposition of anthropogenic nitrate (H^+ - NO_3^-) is fertilizing North American peat bogs. This is in sharp

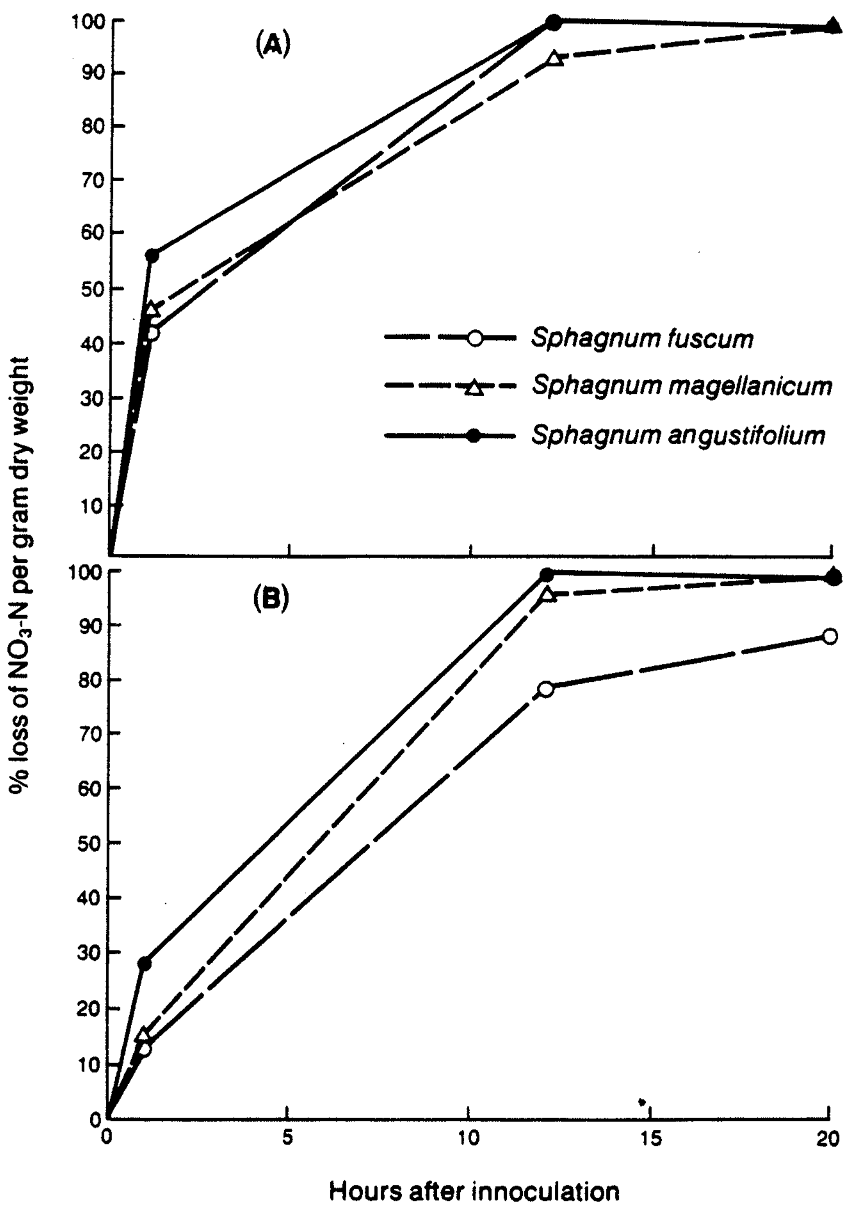


Figure 9.5. Removal of NO_3-N from (A) water containing live green *Sphagnum* and (B) water containing brown stems of *Sphagnum*. (Source: Bayley et al. 1987).

contrast to the situation in England where nitric acid fallout is severely detrimental to the British mires. More investigations are needed to determine the threshold of toxicity at which nitrate changes from being beneficial to detrimental in peatland ecosystems.

Ammonium is also an important pollutant in Canada (Barrie and Sirois 1982). Ammonium deposition seems to be largely retained by most catchments including wetlands but the effects of ammonium on the chemistry of peatland are poorly understood (Jones et al. 1986).

9.3.2.2 Sulphur dynamics. Sulphate retention within North American bogs ranges from 60 to 90% on an annual basis (Bayley et al. 1987; Urban et al. 1987). A decrease in SO_4 concentrations as water passes through a wetland has been noticed in other studies (Braekke 1981a; Kerekes et al. 1985; Rippon et al. 1980; Vitt and Bayley 1984; Wieder and Lang 1982, 1984). For example, Vitt and Bayley (1984) indicated that concentrations of Ca, Mg, SO_4, NO_3-N, and alkalinity were significantly lower upon leaving an oligotrophic fen than when entering it, while hydrogen ion concentration was significantly higher (and pH lower) when leaving the fen. Sulphate is more mobile than nitrate and penetrates more deeply into the peat. For example, SO_4^{2-} concentrations returned to background level only 7 to 14 days after an acidification event at ELA (Figure 9.6). This greater mobility of sulphuric acid may, by different processes, result in changes or damage to peatlands. First, sulphuric acid toxicity to vegetation has been recognized by Ferguson et al. (1978), Glime et al. (1984) and Hutchinson et al. (1987). Secondly, alkalinity also may be generated through sulphate reduction. The anaerobic reduction of sulphate to sulphide (H_2S) requires low oxygen levels and a fixed carbon energy source, as it is carried out by heterotrophic anaerobic organisms (Reuss 1977). Such conditions are found in peatlands. Consequently, high concentrations of sulphide compounds may accumulate and produce alkalinity as H^+ ions are consumed. Conversely Hemond (1980) pointed out that an increase in sulphate reduction could either lead to a significantly increased

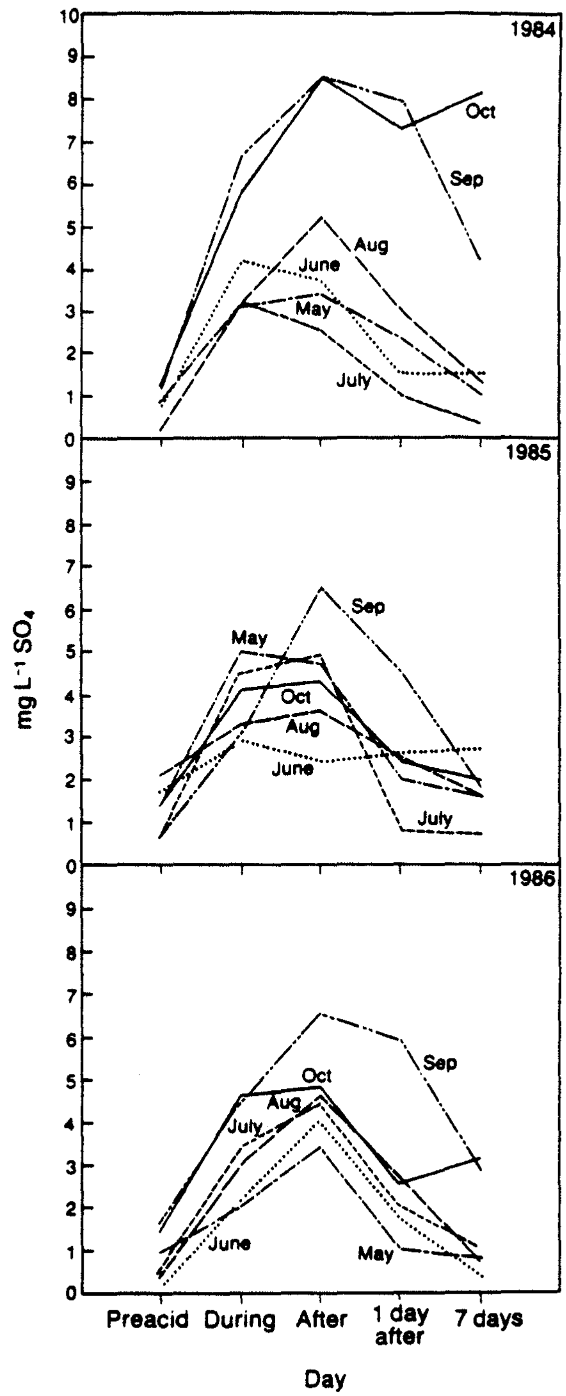


Figure 9.6. Concentration of sulphate in an oligotrophic pool before and after acidification experiments. Samples collected prior to acidification (Preacid), 2 hours after the start of acidification (During), just after (After), 1 day after and 7 days after acidification. (Source: Bayley, University of Alberta, 1987, personal communication).

rate of peat oxidation, which perhaps could not be sustained indefinitely, or could cause an accumulation of H_2S in the water and reach toxic levels. Experimental acidification of the ELA peatland with H_2SO_4 increased the annual retention (Bayley et al. 1987). Thirdly, accumulation of sulphide may also produce an acid pulse either after periods of drought (Bayley et al. 1987; Bayley and Schindler 1987) or seasonal drawdowns (Gorham 1956). Aeration of anoxic soils may result in rapid acidification due to H^+ production during the oxidation of S^{2-} to SO^{2-}. If heavy rain follows, some sulphuric acid is exported. Fourthly, during periods of sulphate flushing, concurrent stripping of other cations may occur (Braekke 1981b, 1981c). This happens when sulphide is reoxidized as mentioned above, or during snowmelt with direct input of a significant quantity of sulphuric acid. The H^+ ions produced replace basic cations on the organic matter complexes (cation exchanger), and these bases may be leached from the peat in association with the SO_4^{2-} anion. Bayley et al. (1987) described a study demonstrating that the sulphate released in an autumn sulphate pulse was accompanied by the release of Ca and Mg.

Sulphate retention is dependent on seasonal changes in hydrology. Jones et al. (1986) suggested that the sulphate retention process has at least a partially permanent role as a sink of acidity since all peat deposits contain sulphur.

Laboratory experiments have shown that sulphate uptake is faster but less complete than nitrate uptake. After 20 hours, 79 to 82% of the sulfate was removed from acid solutions (Bayley et al. 1987). Thus, sulphate uptake is also a source of alkalinity.

In summary, it is clear that sulphate reduction and nitrate uptake have increased considerably in North American wetlands since the period before acid rain. As pointed out by Hemond (1980), these processes have prevented the mineral acidification of bogs which would otherwise have occurred. However, he recognized that there is no mechanism for the protection of above ground biota against mineral acid precipitation, and there is also a limit to the extent to which the system can be theoretically protected. From the discussion above, it is clear that more investigations are needed to establish the degree to which nutrient uptake and microbial reduction can mitigate acidification processes (Gorham et al. 1984).

9.3.2.3 Peatlands as potential sources of acidity in lakes. Many peatlands in North America produce and export organic acids causing streams and lakes to become tea-coloured (DOC > 10 $mg \cdot L^{-1}$) as mentioned by Gorham et al. (1987). Such streams and lakes often lie on noncalcareous substrata, such as the igneous and metamorphic rocks of the Precambrian Shield, and have little buffering capacity (calcium about 50 $\mu eq \cdot L^{-1}$). Gorham et al. (1985) have called attention to the fact that 'brown' acidic water can affect streams and lakes:

> Even in cases where their bicarbonate buffering capacity can accommodate a natural acid input from bog drainage, such lake waters may be especially susceptible to acid deposition from urban/industrial sources of air pollution, because further acid input can readily drive the pH down to levels toxic to the biota.

In Halifax County, Nova Scotia, lake acidification is due more to inputs of coloured organic acids from bogs than to acid deposition from the atmosphere (Gorham et al. 1986). But, as stated by Kerekes et al. (1985), even if the organic acids are the main contributors to the acidity of these coloured waters, anthropogenic sulphur further increases the acidity of these waters, particularly at times of high discharge and snowmelt (LaZerte and Dillon 1984).

There is also evidence that sulphate acid pulses from wetlands as discussed earlier, can cause fish kills (Brown 1980, citing various reports). It is unknown whether wetland acid sulphate pulses have increased in number or magnitude with increased acid rain. No accompanying release of heavy metals or trace metals (Al, Cu, Fe, Mn, Zn) from these acid pulses has been reported.

9.3.3 Peatland Ecosystem Responses to Acidification

The sensitivity of peatlands to acid deposition remains a subject of controversy. Evidence points out that acidic deposition would accelerate the acidification process of natural peatlands. Gorham et al. (1987) presented a conceptual model showing the possible effects of anthropogenic acid deposition on a poor-fen (Figure 9.7) and stated:

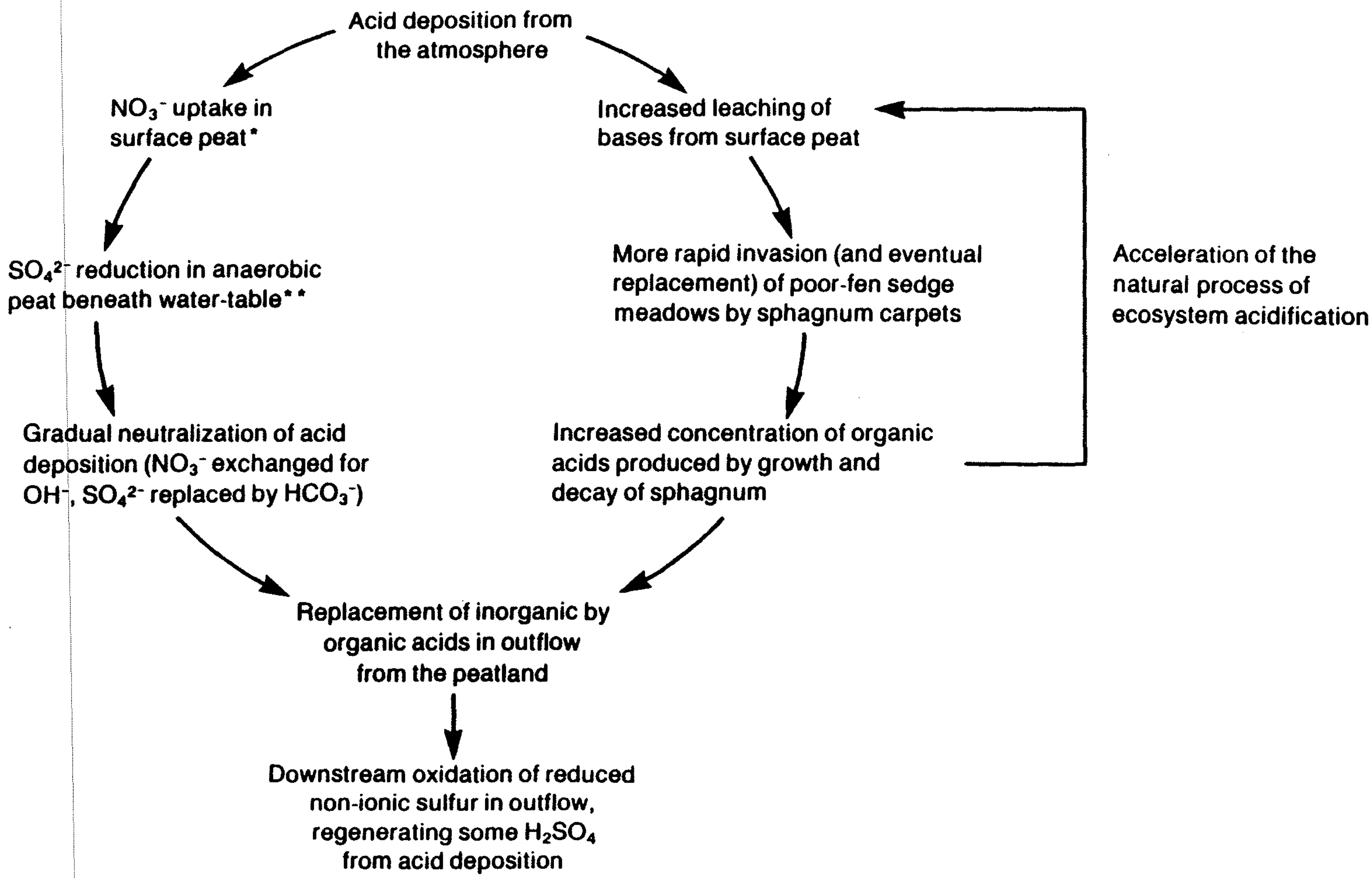
> Even in peatlands where uptake and reduction lower nitrate and sulphate concentrations well below those in atmospheric precipitation, acid deposition can perhaps leach surface peats above the water-table (in fens vulnerable to acidification) to the point that *Sphagnum* invasion is either initiated or accelerated. If that happens it is likely to cause further autogenic acidification as *Sphagnum* generates polyuronic acids, peat rises above the groundwater-table, minerotrophic inputs to the peat surface are cut off, and the slowing of water flow allows the build up of soluble, coloured organic acids produced during decomposition.

In conjunction with studying the acid deposition loading in a region, dry fallout (particularly dustfall) has to be considered in evaluating its potential contribution as a buffering factor.

Increased atmospheric deposition of strong acids may have an effect on nutrient availability in peatlands. Oligotrophic peat has a low degree of base saturation, and such cations as Ca^{2+}, Mg^{2+}, K^{+}, and NH_4^{+} are available from the adsorbed phase rather than the water solution (Gorham 1967; Clymo 1984). An increase in H^{+} ions will replace the cations on the peat exchange complex and nutrients will be more difficult to obtain for plant growth.

Peat plateaus and palsa bogs influenced by permafrost are components of Northern America that may be affected more rapidly by acidification as the frozen ground restricts the volume of peat accessible to acid deposition. Consequently, microbial reduction and sources of minerotrophy are reduced, and the neutralizing capacity of the system is diminished (Gorham et al. 1984).

Peatlands in discharge areas (i.e., receiving runoff water from surrounding watersheds), will have some capacity to buffer acid deposition. Recharge areas, as suggested by Gorham et al. (1984) are



*plus some uptake of SO_4^{2-} **plus some reduction of NO_3^-

Figure 9.7 A conceptual model of the chemical and biological effects of acid deposition upon a fen vulnerable to acidification. (Source: Gorham et al. 1987).

likely to be much more vulnerable. It is well acknowledged that bogs are sensitive systems with regard to pollutants (e.g., Glooschenko and Capobianco 1978; Pakarinen and Tolonen 1976, 1977; Pakarinen 1978).

9.4 SUMMARY AND CONCLUDING REMARKS

At present, there is uncertainty concerning the sensitivity of peatlands in relation to acid deposition. Research in the UK has indicated that *Sphagnum* spp. are affected by ambient levels of acid precipitation. Investigations in other countries, including Canada, have indicated that nitrate and sulphate deposition enhances growth of some bryophytes, at least in the short term. Neutralization of the mineral acids by plant uptake and microbial reduction appear to occur, although this neutralization may be limited, and more research is required to establish the degree to which these mechanisms can mitigate acidification processes.

The acid precipitation problem in recent years has stimulated research in peatland chemistry although our understanding remains primarily descriptive. The processes that play the most prominent roles in controlling alkalinity and acidity in peatlands are: (1) sulphate, nitrate, ammonium, and metals retention; (2) metal complexation and cation exchange; (3) production of organic acids via decomposition; and (4) assimilation (biological uptake). Careful examination of the acidity/alkalinity budgets of peatlands is needed to identify sources and sinks of acidity and to enable the relative rating of the importance of acid deposition in peatland acidification.

As suggested in Gorham et al. (1984), in future studies involving separation of natural from anthropogenic acidification processes in peatlands, it will be necessary to examine the balance between strong mineral acids and organic acids, including the polygalacturonic acids synthesized by *Sphagnum* (Clymo 1963; Spearing 1972) and the complex 'humic' substances (Leenheer 1981; Thurman and Malcolm 1981, 1983) resulting from organic decay and excretion.

There is little information about wetland acidification effects on wetland biota. As mentioned by Gorham et al. (1984), the concept of indicator species within plant or animal communities should be developed to assist in determining the degree of natural or anthropogenic acidification of a given peatland. They suggest studies of acidification responses that focus on plants and animals with narrow pH ranges.

An understanding of the effects of acid deposition will be dependent on the understanding of sulphur and nitrogen behaviour in peatland ecosystems. Development of basic models of biogeochemical cycling of sulphur and nitrogen is required to help prediction of the effects of sulphates and nitrates in acid deposition. Furthermore, the interactions of dry deposition with peatland vegetation need to be elucidated and included in biogeochemical models.

Research on acid deposition effects on peatlands should probably focus on peatland types that are considered sensitive to acidification. For example, Gorham et al. (1984) recognized poor fens as being highly sensitive to acidification with their very low alkalinities, pH values of about 4.5 to 6.0 and wholly organic peats (little input of silt). Similarly, Anderson (1986) classified poor fens, poor swamps, and bogs as being wetland types in Canada most vulnerable to acidification.

In Alberta, information on the amount and locations of potentially sensitive wetland systems is available in a report and map of the sensitivity of soils to acidic inputs (Holowaychuk and Fessenden 1988). However, refinement of the sensitivity classification and mapping, based on further research on acidification and neutralization mechanisms, is needed. The sensitivity ratings are based on general principles of peat soil chemistry and do not take into consideration the acid neutralizing capabilities of the peatland vegetation and of the bicarbonates and organic acids in surface and interstitial waters of the peat. Peat consists of about 10% organic matter and 90% water when considered on a volumetric basis instead of a weight basis. Consequently, peat soils should probably be thought of as organic matrix-ambient water systems (Holowaychuk et al. 1986). Because of their constitution and hydrology, peatlands as terrain components are transitional between terrestrial and aquatic systems. Hence it is necessary to consider all components of peatlands together (i.e., holistically) in assessing their vulnerability to acidic deposition. The examination of vegetation and soil components together is even more important in peatlands than in upland ecosystems because the soil, and its properties, are derived almost totally from the vegetation.

In Alberta, particular attention needs to be given to peatlands in relatively high emission and deposition areas. Examples include the large peatland systems in the Athabasca oil sands region, particularly those that overlie the poorly buffered Canadian Shield. Acid deposition effects in alpine and subalpine areas should also be investigated since ecosystems in these areas are recognized as being fragile and climatologically severe (Hidy et al. 1986).[1]

9.5 REFERENCES

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[1] Editor's note: Also see the recent report and recommendations of Wood (1989).

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10. FORESTRY AND ITS EFFECTS ON PEATLANDS AND ASSOCIATED WATERS

M.E. Pigot

10.1 INTRODUCTION

The concern that greater amounts of timber land may be slated for agricultural or recreational use has prompted an interest in forest peatland drainage in Alberta. The forested peatland resource, excluding privately or federally owned lands, has been estimated at 25 661 km^2; open peatlands are estimated at 13 880 km^2 (letter dated October 6, 1986 from C.A. Dermott, Director, Timber Management Branch, Alberta Forest Service 1986).

This section will discuss the justification for drainage, the system design, and the results obtained, using literature primarily from Fennoscandia and the USSR. Further, this review will consider plans of both federal and provincial departments to drain and possibly harvest forested peatlands in Alberta.

10.2 DRAINAGE - JUSTIFICATION

Peatlands with their associated high water tables result in anaerobic substrates that are the major factor thought to limit tree growth (Heikurainen 1968; Kuusela 1974; Manson and Miller 1955; Munro 1984; Pyatetsky 1974; Thompson 1974). Numerous investigations have shown that root development and penetration into the soil are significantly reduced by high soil moisture (Boggie 1972; Heinselman 1963; Lees 1972; Munro 1984; Rowe 1955). Low tree growth rates are correlated with both low substrate oxygen content and redox potentials, yielding a low overall site quality of undrained and unmanaged peatlands (Bay 1967; Pierce 1953). Above-ground net primary productivity has been found to be well below the average of many forest stands growing on mineral soil (Grigal et al. 1985; Vasander 1982; Paavilainen 1980; Reader and Stewart 1972; Van Cleve 1981). Factors associated with the low productivity include low substrate biological activity and nutrient availability (Pyatetsky 1974; Kozlovskaya 1974; Smoljak 1974), because potential mineralization of organically bound elements is hampered by the low oxygen levels (Malcolm and Cuttle 1983). Hence, tree roots tend to be confined to the aerobic layer, but deeper rooting will occur when water tables are lowered.

10.3 DRAINAGE METHODS AND DESIGN

Drainage is the standard method used to improve tree growth and survival. The objective behind draining peatlands is to control and maintain the water level through a series of ditches. The drainage design (Figure 10.1) consists of a main ditch cut through the centre of the stand - usually the area of deepest peat. Contour ditches which drain into the main channel are dug at a small angle to contour lines to ensure maximum drainage. Trap ditches are placed along the perimeter of the stand to prevent external moisture from entering the area. A settling pond at the low end of the main ditch prevents excessive sedimentation or nutrient loading of the local waterways (Aitolahti 1974; Manson & Miller 1955; Paivanen 1984b).

Optimum spacing and depth of ditches have been discussed in papers by Aitolahti (1974), Braekke (1974, 1983), Heikurainen (1983a, 1983b), Lees (1972), Meshechok (1968), O'Carroll et al. (1981), Paivanen (1984a), Savill et al. (1974), and Vompersky (1974). The peat material and its hydrologic characteristics, especially hydraulic conductivity, influence the structure of the system (Boelter 1974). Models exist based on the spring minimum drainage rate where groundwater levels can be drawn down to 20 cm so that a soil temperature of approximately 5°C can be achieved throughout the rooting depth at the onset of root growth (Pyatetsky 1974). The more common method follows Heikurainen (1983a), where the equation to obtain the desired drainage ditch spacing (S) is:

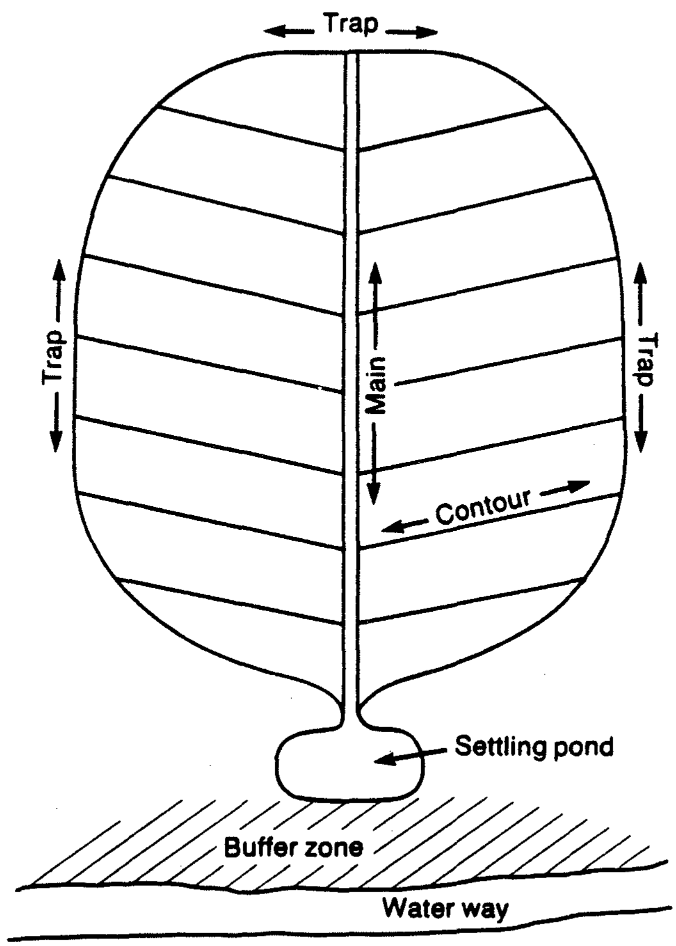

Figure 10.1. Peatland forest drainage design.

$$S = 2(h-n)k/p$$

where S = optimum spacing (m)

n = drainage norm (m) - defined as the distance from the peat surface to the groundwater table measured over the growing season

h = depth of ditches (m)

p = precipitation during the growing season (mm/24 hours)

k = permeability of soil (mm/24 hours; Darcy's formula).

Modelling has shown that ditch depth (m), $\log_e$ (ditch distance (m)), normal rainfall (mm), permeability (mm/24 hours) and water table level (cm) on June 1 can explain 76% ($P<0.001$) of the drainage norm for June to September (Braekke 1983). Permeability will vary between bog and fen according to von Post's humification index (H). When values are greater than 8 ($H>8$), permeability of *Sphagnum* peat (bog) will be less than 20 mm/24 hours. Conversely, sedge peat (fen) at the same value will be less than 200 mm/24 hours (Braekke 1983).

Studies have shown that a distance of approximately 30 to 50 cm to the water table should be maintained for optimal growth of either established or newly planted trees (Boggie 1972; Braekke 1974; Meshechok 1968; Salmgren 1974; Savill et al. 1974). Since the majority of roots occur in the upper 15 cm, obtaining a depth to the water table of more than 50 cm will not have an effect on the thickness of the root layer (Pyatetsky 1974). To attain the maximum depth of the water table and soil oxygenation may take from 2 to 3 years, with water and aerobic levels returning to the surface after 5 to 6 years if ditches are poorly maintained (Lahde 1974).

10.4 DRAINAGE EFFECTS

10.4.1 Hydrology and Soil Physical Properties

Water table drawdown due to drainage produces aerobic conditions which increase biological activity and nutrient cycling, and alter the soil structure. Invertebrate composition, abundance, and biomass increase significantly and cause a more rapid decomposition of the surface organic layers (Kozlovskaya 1974; Okruszko 1968; Smoljak 1974). Excreta produced by the soil animals is rich in mobile N and the microorganisms present in the excreta further decompose organic matter through biochemical reactions. In addition, there are increases in the mineralization of N (Savill et al. 1974), Fe, P, and S, as well as the amount of humus compounds and substances which can be hydrolized (Okruszko 1968). As the water table is lowered, thermal conductivity of the soil is diminished resulting in deeper frost and a longer period of freezing temperatures than on undrained peatlands (Heikurainen and Kenttamies 1978; Heikurainen 1964; Swanson 1986). Increased organic matter decomposition results in the alteration of soil structure, increasing bulk density, and decreasing porosity, water saturation, and hydraulic conductivity (Paivanen 1984c). Such soil structure changes reduce the loss of stored water because of the greater soil matric suction.

In some peatlands, drainage may only lower the water table slightly, because surface drainage may be equal to or less than the volume of subsurface water recharge (Heikurainen 1964). Effects on runoff depend on ditch depth, spacing, peat physical properties, the presence/absence of vegetation, the source of water to the peatland, and changes in topography and physical properties (Verry and Boelter 1978). Seuna (1974) found that drainage of an open bog produced increased spring and summer minimums. Even during dry periods flow continued. Decreased evapotranspiration was presumed to be the cause of this increase because bog vegetation would die due to a change in habitat. Forested peatlands showed different results such that tree stands continued and probably increased transpiration as well as interception due to improved growth (Boggie 1972; Heikurainen 1964; Seuna 1974). Consequently, woodless bogs would have a greater runoff during the autumn than forested bogs (Vompersky 1974; see also Chapter 5).

Differences in runoff between drained and undrained bogs may be partially a function of time (Heikurainen 1980; Manson and Miller 1955; Verry and Boelter 1978). Regardless of site location, a rapid elevation of groundwater in the spring occurs, but it recedes during summer and remains low during fall and winter. A slower rate of snowmelt and an available water storage capacity on a drained peatland results in a longer but lower peak spring runoff than on an undrained area. Similarly, in drained stands, longer but lower peaks in the summer volume of surface runoff have been explained by an increased interception of precipitation by trees and a higher storage capacity. Conversely, undrained fens show much more stable runoff over time because of the continual water inflow from exterior areas (Verry and Boelter 1978). However, drainage is expected to alter this stability due to surface subsidence and decomposition.

10.4.2 Water Quality[1] Suspended solids alter the runoff water quality during the excavation of the ditches with the quantity of solids increasing with runoff volume (Clausen 1980; Kenttamies and Laine 1984). For example, the excavation of a 500 m ditch length resulted in the removal of approximately 500 kg soil during maximum spring runoff but only 1 kg or less during winter (Sallantaus and Patila 1983). Solids decrease following the completion of ditch construction, however, their presence in the runoff may lead to an increased biological oxygen demand and affect downstream waters. Dissolved organic matter also increases, depending on runoff amounts (Kenttamies and Laine 1984). Water colour becomes darker due to the increased release of humic and fulvic acids (Clausen 1980), reducing the availability of light and hence photosynthesis by downstream aquatic plants. Bog runoff pH may decrease with increased release of weak organic acids caused by greater humification of bog *Sphagnum* (Hemond 1980). Conversely, pH values of fen runoff may increase where ditches pass through minerotrophic *Carex* layers, although values have been found to decrease during high runoff on sites in Finland, due probably to acid precipitation (Sallantaus and Patila 1983). Further, fens have higher specific conductivity and Ca and Mg values than bogs (Clausen 1980). Aqueous concentrations of total P and Fe often increase with runoff, except when oxic conditions co-precipitate these ions (Pashkevitch 1984). Other studies have shown that the amount of P in the water only increases following aerial fertilization (Kenttamies 1981; Kenttamies and Laine 1984). An increase in Al and Na has also been observed in runoff (Clausen 1980; Paavalainen 1980). An important design consideration is that the quantity of released minerals is believed to be negatively correlated with ditch spacing (Konstantinov et al. 1983).

Over-draining can reduce the number of substrate organisms (Kozlovskaya 1974). Yefremova (1984) found that high drainage volume led to greater subsidence, a lower redox potential, complete decomposition of organic compounds, leaching of humic substances, flocculation of colloids, and reduction of nitrates to nitrites and ammonia as the breakdown of the peat structure increased. Further, the amount of humic substances has been positively correlated and the amount of carbon has been inversely related to the structure of the peat. Recharge of soil moisture becomes mainly dependent on precipitation when over-drainage reduces capillarity and surface layers can no longer replenish moisture lost through evaporation (Lindholm and Markula 1984). As the peat loses its ability to hold water, it becomes more dense and less porous which causes subsidence. Unchecked, the peat will become irreversibly dry and cracked (Mulqueen 1986). Hence, over-drainage will produce conditions unfavourable to plant growth and will alter the runoff water quality.

[1] Refer to Chapter 6 for an in-depth review of water quality.

10.5 EFFECTS OF DRAINAGE ON SITE PRODUCTIVITY

In most cases drainage of forested peatlands improves tree growth (Aitolahti 1974; Boelter 1974; Holman 1964; Istomin and Tarakhanov 1974; Medvedyeva 1974; Payandeh 1973a; Vasander 1982) (Figure 10.2). Height and age at the time of drainage are often limiting factors to revival. The youngest, most vigorous trees show the best response; trees taller than 12 to 14 m, with diameter at breast height greater than 15 to 19 cm, show insignificant improvements (Heikurainen 1964; Istomin and Tarahanov 1974; Payandeh 1973a). Growth is also dependent on site fertility, the success of drainage, the climate of the area, and the peatland type (Heikurainen 1963). Hence, fens tend to be moderately more productive than bogs (Vasander 1982), and similarly, perched bogs are more productive than raised bogs because of the greater nutrient input (Grigal et al. 1985). Because thickness of the undecomposed *Sphagnum* layer is negatively correlated with the site index (Heinselman 1963), drainage should improve tree growth as this method eventually reduces the thickness of *Sphagnum* mats.

Root growth commences when the soil temperature becomes greater than 5°C (Pyatetsky 1974) and will therefore be restricted to the surface layers in areas of deep frost or permafrost. Further, depending on the drainage efficiency, downward movement may be limited by soil texture, structure, excess moisture, and poor aeration. This shallow rooting leads to wind-throw in thin stands unless windbreaks are included in the stand structure (Heikurainen 1964). However, drained stands are expected to have less wind-throw due to superior rooting structures, compared to undrained peatlands.

After drainage, although the total tree biomass increases, plant material has been found to differ in both structure and biomass allocation (Medvedyeva 1974). Under boggy conditions, the majority of biomass is composed of shoots and needles or leaves, whereas after drainage, stem wood production is predominant. However, Wang et al. (1985) found that young spruce and tamarack growing under fen conditions in north-central Alberta showed a decrease in wood relative density and tracheid length with an associated increase in radial and volume growth.

10.6 SILVICULTURE IN DRAINED PEATLANDS

Silvicultural practices usually recommend the application of fertilizer for improved growth on bogs (Table 10.1) (Carlyle and Malcolm 1986a; Vasander 1982), although fens may yield a less favourable response especially if newly drained (Paavilainen 1980; Valk 1980, 1983). In general, P and K (Carlyle and Malcolm 1986b; Heikurainen 1983b; Meshechok 1968) and often N are limiting nutrients to growth on bogs (Boelter 1974; Paavilainen 1974, 1980; Smoljak and Ipatiev 1983). Application of P is generally low since organic soils cannot retain high levels. Retention of fertilizer P is positively correlated to the mineral fraction in the soil, especially Fe and Al. The low availability of these elements in peat means that excess P cannot be adsorbed and is therefore easily leached (Ahti 1984). Similarly, readily mobile K will not be retained by peat soils (Malcolm and Cuttle 1983; Martin 1980). An estimated 37% of the 50 kg P ha^{-1} applied to a lodgepole pine site was leached 7 years after application, with an additional 28% believed to be unavailable to plants (Malcolm and Cuttle 1983). Long-term P release has been observed, showing that leached amounts are highest immediately after application and can be found in the runoff up to 10 years following fertilization (Ahti 1984). Overdosage of K and P has been found to increase acidity in the surface layers hindering tree development (Kenttamies and Laine 1984; Smoljak and Ipatiev 1983). Karsisto (1974) and Paavilainen (1980) suggested fertilization 3 to 5 years after drainage to allow the trees to recover and nutrient mineralization to occur. Improved growth due to fertilization stabilizes drainage, since interception and transpiration are greater due to increased needle size (Paivanen 1974).

Maintenance of drainage ditches ensures optimum water levels. Studies show that site quality deteriorates over time and hence tree growth is negatively affected if ditches are allowed to

Table 10.1. Biomass and element content of fertilized and unfertilized 7-year-old lodgepole pine.[a]

	Biomass ($t \cdot ha^{-1}$)	N ($kg \cdot ha^{-1}$)	P ($kg \cdot ha^{-1}$)	K ($kg \cdot ha^{-1}$)
Above-ground	20.8[b]	131.1	12.5	47.1
	(0.6)	(7.3)	(0.4)	(1.6)
Whole-tree	26.9	143.0	14.2	53.6
	(0.7)	(9.1)	(0.5)	(1.8)

[a] Source: Carlyle and Malcolm (1986b).
[b] Fertilized is non bracketed; unfertilized in brackets.

accumulate silt or vegetation (Aitolahti 1974; Heikurainen 1964, 1968; Istomin and Tarakhanov 1974; Lahde 1974).

Harvesting results in a rise in the water table and increased runoff because throughfall increases and transpiration decreases (Paivanen 1974, 1980). In areas of permafrost activity, removal of vegetation may cause permafrost aggradation and deep frost cracking due to the absence of an insulating snow cover associated with windswept conditions (Payette et al. 1986). Zoltai and Tarnocai (1971) found that dense stands caused similar effects, inhibiting accumulation of snow. Hence, stand thinning promotes the active soil layer whereas clear-cutting may increase permafrost. It should be noted that clear-cut harvesting of undrained peatland forest increases water level fluctuations in fens and leads to mineralization and release of P (Knighton and Stiegler 1980; Verry 1980). Conversely, release of P from bogs is more dependent on daily precipitation and evapotranspiration and is available due to the lower mineral fraction and hence lower adsorption ability of the peat (Knighton and Stiegler 1980). Since cutting is most often carried out during winter, methods of harvest will determine the amount of organic matter and debris present in spring runoff.

Afforestation of open peatlands and reforestation of harvested areas are necessary since natural invasion is too slow and might promote inferior species (Heikurainen 1968). Site preparation includes burning to reduce competition (Boelter 1974), mulching with excavated peat from the ditches and furrows (Braekke 1974; Heikurainen 1964; Lees 1972; Savill et al. 1974; Thompson 1974), and establishing nurse crops to reduce wind loss and freezing of the seedlings (Heikurainen 1968). Point fertilization is often used during planting (Mesechok 1968) to supplement the nutrients released through oxidation of the exposed peat. Application of fertilizer before planting may lead to leaching especially of P and K, with the possibility of large amounts of K increasing acidification of the peat as it replaces H^+ ions on the exchange sites (Malcolm and Cuttle 1983). Seedlings generally favour the moist conditions of the peat (Rowe 1955) but the colder temperature of the peat, especially during the spring, may inhibit growth (Heikurainen 1964).

10.7 DRAINAGE ECONOMICS

The actual cost of turning peatland into productive forest should be a major consideration in site selection. The majority of the cost is from actual construction of the drainage system, although other costs include ditch maintenance, stand thinning, afforestation, and fertilization (Keltikangas and Seppala 1974). Design must compromise between biological and economical spacing for optimum drainage. The latter is usually wider to compensate for trees lost along the ditch lines and the digging cost (Braekke 1983). Profitability of drainage is affected by the site quality and potential for future development (Paivanen 1984b). In a black spruce peatland in Ontario, approximately 30 years were required for the increased value of a drained stand to offset the cost of site improvements (Payandeh 1973b). For future management investments, the optimum rotation period to harvest will be influenced by variable costs of drainage, silviculture, and stumpage. Estimates of costs are higher in the northern latitudes since the colder climate will decrease the tree growth increment and will lengthen the harvest rotation (Heikurainen 1968; Keltikangas and Seppala 1974).

10.8 PEATLAND FORESTRY IN ALBERTA

Research and practical application of peatland drainage for forestry purposes are still in the preliminary stages in Alberta. A Canadian review identifies that the majority of research investigations are less than 5 years towards completion (Table 10.2), and thus for forestry, there is a lack of long-term data (Hillman 1987). Unfortunately, some studies, such as the Athabasca forest project (Nash 1983), were not statistically designed and tend to be descriptive and confined to tree growth. More recent drainage research has been designed to determine optimum ditch networks, the effects on the physical and chemical properties of peat, and species response to silvicultural applications.

The basic Alberta design strategy borrows heavily from the Fennoscandian experience; however, results may be different due to climatic variation, and the predominance of fen as opposed to bog conditions. Yearly ground water fluctuations are more pronounced in bogs and hence stands are adapted to drawdown moreso than on fens where annual water levels are fairly constant (Verry and Boelter 1978). Severe drainage of fen sites may cause mortality when stands are subjected to a change in soil moisture (R. Rothwell, Department of Forest Science, University of Alberta, 1986, personal communication), and an alteration in the nutrient balance (Makitalo 1985). Even in the more southerly areas, permafrost aggradation may occur due to the drying of surface layers and a reduced thermal conductivity of the soil (Swanson 1986). Alteration of soil temperature may prove detrimental to seedlings used in afforestation or reforestation. Further, the Forest Ecological Classification system which is used in Ontario for site selection and drainage design is not applicable in Alberta. The high occurrence of forest fire due to a drier, hotter growing season makes development of one system difficult, and therefore, site selection was based on the following: the Wetlands Classification System produced by the National Wetlands Working Group, the physiognomic classification for

Table 10.2. Wetlands drainage for forestry projects in Alberta.[a]

Legal Description	Drained Area (ha)	Total Length of Ditches (km)		Year Started	Lead Agencies
		Main	Lateral		
Athabasca Forest					
1. Sec 23, 24, 25, and 26 Tp 87, R 9, W 4 Mer	76	2.12	-	1975	AFS[b]
Peace River Forest					
2. Kimiwan: Sec 16, 17, 20, and 21 Tp 80, R 17, W 5 Mer	172	4.05	71	1984	AFS
3. Kimiwan II: Sec 22, 26, and 27 Tp 79, R 19, W 5 Mer	456	4.45	115	1985	AFS
4. Manning: Sec 12 Tp 95, R 1, W 6 Mer	105	3.77	40	1984	AFS
5. McLennan 28: Sec 28 Tp 79, R 19, W 5 Mer	85	2.61	28	1985	CFS[c]/AFS
Slave Lake Forest					
6. Goose River: Sec 14, 15, and 23 Tp 68, R 19, W 5 Mer	131	2.73	37	1985	CFS/AFS
7. Saulteaux River: Sec 8, 9, 16, and 17 Tp 71, R 2, W 5 Mer	50	2.69	16	1981	AFS/U of A[d]
Whitecourt Forest					
8. Wolf Creek: Sec 19 and 30 Tp 51, R 14, W 5 Mer	72	1.53	19	1985	CFS/AFS

[a] Source: Hillman (1987).
[b] AFS = Alberta Forest Service.
[c] CFS = Canadian Forestry Service.
[d] U of A = University of Alberta.

peatland vegetation developed by Johnson and Zoltai, and Jeglum's guidelines for trophic states (Hillman 1987).

Most recent studies in Alberta have concentrated on the effect of drainage on the growth and rooting depth of tamarack and spruce (Lieffers and Rothwell 1986, 1987a; Dang 1988) and on soil temperature (Lieffers and Rothwell 1987b); effects of climate on tree growth in fens have also been studied (Dang and Lieffers 1989). Dang (1988) found that above-ground tree growth was positively correlated to the depth of water table as well as minimun air temperatures during growing season. However, improved growth may only be noticeable after 3 to 6 years following drainage and it will take from 13 to 19 years for tree species to achieve stable growing conditions. Possible reasons for the slow improvement may be due to the small root system and leaf area of trees on undrained peatlands. Photosynthetic products may be initially too low to improve growth. Since vertical reach of roots is determined by the seasonal maximum height of the water table (Dang 1988) causing shallow root systems, initial drawdown of water levels may produce water stress. Conversely, heavy precipitation and a resulting increase in water table may produce root dieback. Although roots will penetrate deeper once the water table is lowered, they will be frozen longer due to the deeper, longer-lasting frost associated with drained peatlands (Lieffers and Rothwell 1987a). Hence growth will be hindered (Lieffers and Rothwell 1987b) and dieback may occur. However, the benefit of deeper root penetration is one of greater tree stability and since the roots are mychorrhizal, they may prove to be a source of mineral nutrition (Lieffers and Rothwell (1987a). Generally, medium, coarse, and very coarse roots are restricted to the surface 30 cm (Lieffers and Rothwell 1987b). This layer tends to warm more slowly on drained than undrained sites but will eventually reach higher (3 to 4°C) temperatures than undrained sites. In the long term, this produces an earlier bud flush in tamarack. However, should temperatures become too high during growing season, tree growth will decrease due to greater respiration and water stress caused by warm dry conditions (Dang 1988).

Of the eight areas under research in Alberta, only Wolf Creek will have baseline data available for future comparison with drainage treatment and control plots (Hillman 1988). Wolf Creek is one of the three sites chosen for the Wetland Drainage and Improvement Program - part of the Canada/Alberta Resource Development Agreement being undertaken by the Forestry Canada and the Alberta Forest Service (Hillman 1987, 1988). This program is designed to produce cost-effective and environmentally sound forest drainage methods specific to Alberta. Major concerns being considered are: (1) the effect of drainage and resulting decomposition/mineralization on the organic chemical water quality, (2) the differences between fen and bog drainage, (3) microclimatic variations due to the loss of surface water, (4) tree revival, survival, and quality especially in northern areas, (5) variation in tree responses among Alberta, Canada, and international investigations, and (6) the ability to model research results. Although models that simulate optimum forest drainage systems for bog sites are available in the literature (Elpatievsky 1974; Paivanen 1984a; Payandeh 1987; Salmgren 1974), these models may not be applicable in Alberta.

10.9 CONCLUSIONS

The availability of literature on the drainage of peatlands and the subsequent site amelioration for forestry is extensive. However, the applicability of this published research to the Alberta situation is often hindered by the varied peatland types and climate. Investigations outside Alberta have shown that drainage and various silvicultural methods improve tree growth, but at the same time affect off-site water quality by increasing the suspended solids, nutrient loading and the colour of runoff. Over-drainage has been shown to cause irreversible drying and cracking of the peat, creating an unusable medium for plant growth and survival. Because Alberta is in the preliminary stages of peatland forestry research, knowledge of whether this resource will be economically and environmentally feasible is still uncertain.

10.10 SUMMARY

1. Drainage generates tree growth by improving aerobic conditions which increase biological activity and nutrient cycling, and improve soil structure.
2. Drainage increases spring and summer minimum runoff; evapotranspiration may increase slightly due to improved tree growth and hence increased transpiration and interception.
3. Drainage alters water quality by increasing suspended solids, dissolved organic matter, water colour, and nutrient loading.
4. Over-drainage increases the breakdown of the peat structure to the point where the peat will eventually lose its ability to hold water, and become dry and cracked.
5. Silvicultural practices can alter water and soil quality; fertilizer P and K are easily leached and if over-applied may increase the acidity of surface layers.
6. Harvesting of timber increases runoff and produces a rise in the water table. In areas of permafrost, removal of vegetation may cause aggradation and deep frost cracking due to loss of an insulating snow cover resulting from windswept conditions.
7. Applicability of published data to Alberta peatland forestry may be hindered by the different and varied peatland types and climate of the province as compared to other provinces and countries.

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11. PEATLAND AGRICULTURE AND ITS EFFECTS ON HYDROLOGY AND WATER QUALITY

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11.1 INTRODUCTION

Pressures to expand the agricultural land base in Alberta are resulting in increasing areas of peatlands being cleared and used for farming. Peatlands have long been used for the production of a wide variety of agricultural products in Europe. In Canada, the variety of crops grown is not large due mainly to adverse climatic conditions. Peatlands are nevertheless being increasingly used for agriculture. With drainage and clearing of peatlands for farming, significant changes in the regional hydrologic balance and the composition of downstream waters can occur.

Peatland agriculture in Alberta, and the impacts on hydrology and quality of waters associated with peatlands, are discussed in this chapter. Only peatlands used for farming in situ are discussed. The use of peat moss extracted from bogs as a growing medium in greenhouses or as a soil amendment is not included in the discussion. However, the process of extracting peat moss from virgin peatlands is discussed in Chapter 11.

Agricultural practices on both mineral and organic soils can be categorized as follows: (1) horticultural, (2) orchards and vineyards, (3) cropland, (4) improved pasture and forage crops, and (5) rough grazing and rangeland (Leeson 1969). Orchards and vineyards on peatlands are uncommon in Canada. Horticultural use of peatlands is mainly limited to areas with relatively warm climates such as those of southern Ontario and southern British Columbia. The major practices on peatlands in areas of cooler climates (i.e., relatively few growing degree-days and short frost-free period) are improved pasture and forage crops, cropland (coarse grains and oilseeds in Alberta), and rough grazing and rangeland.

11.2 DEVELOPMENT OF PEATLANDS FOR AGRICULTURE IN ALBERTA

Shallow organic soils occurred extensively in the central and northwestern areas of the province prior to settlement and agricultural development. Removal of these peats by burning to convert land to agricultural production was a common practice. The landscape now consists of highly productive mineral soils. Some deeper organic soils were broken and cleared for farming, but much of this land appears to have been abandoned and has reverted to native vegetation such as willow scrubland (observations made by the author and by staff of the Alberta Soil Survey). With larger and more versatile machinery for clearing and drainage, and with improved management practices, these peatlands are again seen as having potential for productive agriculture.

The following description of land preparation is largely derived from personal communication with R. Sherstabetoff, Organic Soil Specialist, Alberta Agriculture, Edmonton (1986) and from Sherstabetoff (1987). Clearing and drainage are generally the first procedures required in developing peatlands in agriculture. Surface vegetation is usually removed by bulldozing into windrows during the winter. The windrows are subsequently burnt and the residues spread over adjacent soil surfaces and incorporated into the soil in later tillage operations. Ditching is done in the spring but this can vary according to when and where the need arises; generally peatlands do not require extensive drainage. Shallow surface ditches are excavated to control spring melt and runoff as well as to remove excess water from depressions and lower slope or basal watershed situations. Excavation is performed with available farm equipment since the purchase or rental of special ditchers would be uneconomic. Further information on preparation of organic soils for farming can be found in Stewart (1977).

Areas selected for drainage are within a farmer's holdings or on leased crown land adjacent to those holdings, and hence, usually accessible by road. Associated road ditches appear to have been effective in handling excess runoff from field drainage as well as reducing the number of ditches required to drain a given area. Field runoff funneled into the road ditches will eventually enter local waterways. Water level control within drainage systems is limited and may cause problems as described in following sections.

Various methods are used in developing peatlands for farming in Alberta. There are three general practices followed by different farmers upon clearing and draining peatlands. In one of these, the soil surface is ploughed and cultivated to prepare a seed bed. The peat soil is used without any further modification of the surface. About 1 or 1 1/2 years are required before crops can be planted when the organic soil is prepared in this way for farming.

Another practice, though somewhat uncommon, consists of restructuring organic soils according to the 'black culture' method used in Germany. It involves deep ploughing peat to bring mineral subsoil to the surface. A variation of this consists of spreading peat onto adjacent mineral soil and of spreading mineral surface soil onto peat surfaces. This latter technique is similar to the another method practised in Germany known as 'sand-cover cultivation' (Kuntze 1980).

A third practice consists of removing the entire peat deposit by bulldozing down to mineral soil, windrowing, burning the peat, and spreading the residues. This procedure is a variation of an older method in which peat was burned in situ. Ditching to remove excess water is not needed unless some of the land acts as a receiving basin for snowmelt or runoff waters. Two to three years are required before a crop can be sown.

11.3 FARMING PRACTICES ON ORGANIC SOILS IN ALBERTA

Information about farming practices on organic soils in Alberta was obtained by personal communication with R. Sherstabetoff, Organic Soil Specialist, Alberta Agriculture, Edmonton (1986) and from Sherstabetoff (1987). Farmed peatlands in Alberta are used mainly for pasture, forage crops and annual field crops. Barley is the major grain while canola is the major oilseed crop grown. Use of organic soils mainly for hay and pasture has been advocated in the past, but the production of annual crops appears to have increased in recent years. Because of problems with early fall frosts, early-maturing varieties of the annual crops are required. The main forage grasses grown are timothy, reed canary, meadow foxtail, and orchard grass; Alsike clover is the main legume grown.

Liming of organic soils is generally not practiced. Forage and hay crops may or may not be fertilized. In more intensively managed pastures, fertilizers are applied at rates of up to 45 to 62 $kg \cdot ha^{-1}$ N (50 to 70 $lb \cdot acre^{-1}$), 89 to 107 $kg \cdot ha^{-1}$ P (100 to 120 $lb \cdot acre^{-1}$) and 89 to 107 $kg \cdot ha^{-1}$ K (100 to 120 $lb \cdot acre^{-1}$). Good responses to fertilizers are obtained with forage, pasture, or annual crops. For grain crops, Cu application is necessary to avoid poor heading and low yields. The Cu is usually applied as a foliar spray but experimentation with soil incorporation of $CuSO_4$ (bluestone) or other suitable forms of Cu is also being carried out.

In addition to fertilization, organic soils must be managed to control weeds, erosion, and excess water. Weeds are controlled as on mineral soils, although high organic matter contents preclude the use of herbicides such as Avadex. As indicated previously, few farmers practice water table control. Control is especially difficult near roadways where ditches are somewhat deeper than required for recommended water table levels. Consequently, wind erosion can be a major problem with desiccated peat surfaces. Practices to reduce erosion include incorporation of crop residues into the soil in the fall, avoiding disturbance in the spring, and use of cover and pasture crops after harvest of the primary crop.

11.4 EFFECTS OF PEATLAND AGRICULTURE ON SOILS AND RUNOFF WATERS

11.4.1 Post-Development Effects

The effects of clearing and drainage include subsidence due to dewatering and to compaction by machinery, reduction in evapotranspiration due to removal of all vegetation, and oxygenation of the surface peat layers resulting from lowering the water table and mixing the surface layers. It is recommended that clearing and draining organic soils not intended for immediate agricultural use should not be undertaken because of the increase in fire hazard and the rate of decomposition (Lucas 1982). However, it may require a year following drainage before a seedbed can be developed. A number of changes can occur during this period. The amount of peatland runoff may increase due to reduced evapotranspiration. Levels of particulate and dissolved organics in runoff waters may increase due to physical disturbance of the peatland surface and to accelerated decomposition. Lastly, increased levels of oxidation products resulting from aeration may occur in the runoff water. In particular, acidity may rise as a result of oxidation of hydrogen sulphide (H_2S) to H_2SO_4.

11.4.2 Impacts of Continued Farming and Management Practices

It is well documented that continued agricultural use of peatlands usually results in subsidence, that is, the lowering of surface elevation. The rate of subsidence is influenced by various factors including: (1) biological oxidation, (2) height of the water table, (3) character of the organic material, (4) compaction, (5) burning, (6) wind erosion, (7) water erosion, (8) shrinkage and dehydration, (9) geological subsidence, and (10) cropping system (Lucas 1982). The subsidence is accompanied by changes in physical properties such as a decrease in content of large pores and an increase in bulk density.

Although macropore space is lost due to subsidence, total porosity diminishes only slightly. Hence, microbial degradation of peat above the water table and oxidation reactions such as those listed in Table 4.1, Chapter 4 continue to occur.

Pilspanen and Lahdesmaki (1983) have indicated that the main effects of draining and subsequent drying of peat are: (1) a slight drop in pH, (2) a lowering of the aerobism limit, and (3) an increase in overall biological activity. Nitrogen compounds are mobilized into forms usable by plants in drier peats. Aerobic microorganisms liberate free amino acids and amides from peat proteins. Production of ammonia and urea is stimulated, as is nitrification. Most of the N in runoff waters of drained and cultivated peatlands in Ontario (Miller 1979) and in Michigan (Erickson and Ellis 1971) was found to be in the nitrate form. Clausen (1980), however, found that most N in runoff was in organic or ammonia form. Clausen (1980), in a study of quality of runoff from natural and disturbed Minnesota peatlands, found that the disturbed peatlands had higher concentrations of suspended sediment, acidity, K, Fe, Al, and Na. Disturbed peatlands, including one under cultivation, also had darker runoff than undisturbed areas. Seasonally, spring minimum values were found for pH, alkalinity, specific conductance, Cu, Mg, colour, and various forms of N. Conversely, suspended sediment levels were highest in spring. Average total P concentrations, and total-, nitrate-, ammonia-, and organic-N forms, were highest in disturbed (drained and cultivated) peatlands.

Some of the effects of peatland farming on plant nutrients in organic soils in Germany have been reviewed by Kuntze (1984). In low-moor (or fen) soils, the N content is relatively high and the C/N ratio is sufficiently narrow for vigorous microbial activity to occur. The rate of N mineralization commonly exceeds the N levels required by crops and some of it ends up in the soil leachate. The mineralization rate is dependent on temperature; hence, it may be expected that rates would be lower in Alberta as compared to those in Germany. There is evidence that the N is not necessarily lost to drainage waters but that a considerable proportion of it is denitrified (Kuntze 1984). In high-moor (or bog) soils, N levels are low and the C/N ratio is high. Such soils need to be limed to increase pH, but accelerated decomposition results from this. Over time, the surface soil is enriched in N and becomes more similar to fen soil. Thus, native N as well as any N added through fertilizer is retained by the soil. As long as the soil is quite acidic, nitrification is low and leaching to the subsoil and groundwater is negligible.

Drainage and cultivation of peat in Ontario, Minnesota, and Florida have been reported to increase phosphate concentrations in runoff (Hortenstine and Forbes 1972; Nicholls and MacCrimmon 1974; Clausen 1980). Phosphorus may be released as a result of mineralization of fen soils. If liming is practiced, phosphates are quickly immobilized and transformed into calcium phosphate which is only slightly soluble (Kuntze 1984). This phosphate, nevertheless, seems available to plants and fertilization isn't required. In bog soils, precipitation of phosphate is not possible due to lack of free Ca, Al, and Fe ions. Some organic P may accumulate near the surface with microbial decomposition, but some P may also be leached. The release of P into the drainage water increases with the duration and intensity of agricultural use.

Farmed fen soils are usually deficient in potassium and fertilization is necessary for successful crop growth. In soils rich in lime, applied K is unable to displace Ca on exchange sites. In bog soils which are limed, K is easily displaced by Ca. Unless it is used by the growing crop, such K is subject to leaching and may enter drainage waters (Kuntze 1984).

Methods to mitigate nutrient losses and conserve nutrients as well as the peat soil material have been suggested by Kuntze (1984). Nitrogen losses from grasslands have been found to be 10 times lower than those in cultivated fen soils. Sand-cover cultivation, or covering peat soil with a layer of sand, is commonly practiced in Germany. This practice has been found to slow down decomposition and thus conserve the nutrient content. Raising and lowering the water table, thus varying aeration and the redox potential, has also been found to reduce nitrification. Phosphorus fertilizer losses from peatlands depend on timing of application. Increasing the pH by liming reduces P losses, but at the same time stimulates decomposition. Application of Fe to bind phosphate in the soil has also been found to be successful in reducing losses.

The leaching out of K into drainage waters is ecologically less hazardous than the loss of N or P. Concerns about conserving K are mainly economic; that is, it needs to be applied at seeding time or spread out over the growing season so that it is used by plants rather than lost to drainage waters.

For all three elements above, sand-mix and sand-cover cultivation reduces the loss of nutrients to levels similar to those of mineral soils. Without sand application P losses from bog soils can be 10 to 100 times greater than for mineral soils (Kuntze 1984). However, in conjunction with using peatlands primarily as permanent grassland for hay or pasture, sand incorporation could considerably reduce nutrient losses to groundwater. Kuntze (1984) noted that stock density on pasture should be limited to two livestock units per ha or loss of P by leaching may occur.

The transport of nutrients through a wetland in New York was studied by Peverley (1982). The study area consisted of cultivated organic soils and wetlands managed for waterfowl downstream from the cultivated soils. Nutrient concentrations were found to be highest in sites adjacent to cultivated peatlands. The soluble N, P, and K concentrations were elevated compared to most rural streams. Levels of $(NO_3^- + NO_2^-)$-N were highest during fall and winter flows, and lowest during late summer. Ammonium-N was usually <10% of dissolved inorganic N. Reactive-P levels were highest in summer and fall. Concentrations of dissolved unreactive P were usually less than half those of reactive P and followed the same seasonal trends. Potassium and Ca concentrations were highest in the fall, when the streamflow levels were lowest, indicating a simple dilution pattern. However, the highest total loading (i.e., total export of nutrients in runoff) was associated with highest flows, in winter and spring, for all elements. Dissolved-C levels were highest in the fall (the lowest flow period), although over half of this was inorganic C; the pH and alkalinity of the system were high due to underlying calcareous materials. About 40% of the C was organic, mainly in the form of fulvic and humic substances.

Drainage and cultivation of marshes in Wisconsin had effects on water composition similar to the studies described above. Nitrate-N, ammonia-N, organic-N, soluble orthophosphate, Ca^{2+}, Mg^{2+}, Na^{+}, K, alkalinity, and SO_4^{2-} were found to be substantially higher in tile drainage waters of cultivated marshes as compared to natural or drained, but non-cultivated, marshes (Lee et al. 1975). Fertilizers were considered to have contributed to this chemical discharge from cultivated marshes. The drainage waters from both uncultivated and cultivated marshes were considered to reduce water quality in receiving waters by increasing water hardness, alkalinity, and colour and by contributing to the eutrophication of these waters by the discharge of N and P. The greatest impact among these, however, was from the cultivated marshes.

There have been few reports about the levels of trace elements in drainage waters of cultivated peat soils. Pashkevitch (1984) reported some observations of impacts from draining peatlands for agriculture in the Polessie region of Byelorussia. Drainage resulted in a sharp increases in Fe^{2+} concentrations in drained fens, especially those in floodplain topographic situations. As aeration in drained peat soils improves, oxidation conditions result in a decrease of Fe in drainage waters. Contents of Mn, HCO_3^-, NO_3^-, and SO_4^{2-} also were observed to increase with a reduction in water level. The elements N, S, K, P, Pb, Zn, Cu, and Sb were noted to be relatively high in river waters, the catchment areas of which were extensively drained (Pashkevitch 1984, citing other research in the USSR).

From the previous description of peat farming practices in Alberta, and from the review of impacts in other regions, the following general impacts on runoff waters may be expected to occur:

1. Major nutrients such as N and P will be mobilized, but positive responses of crops to fertilization suggest that crops take up most of these nutrients during the

growing season. Over-fertilization would result in increased levels of N and P in runoff waters.

2. Soil surfaces will become well aerated as a result of drainage and tillage operations. The pH of runoff waters may decrease due to S oxidation, particularly after draining and clearing, and during early years of farming the peat soils.
3. Levels of other soluble base cations and of heavy metals will likely increase.
4. Dissolved organic carbon levels may increase.
5. Particulate organic carbon levels will increase during operations which drastically disturb the soil, (i.e., initial clearing and drainage operations; wind erosion episodes).

Most of these effects would likely be substantially decreased where mixing or overlaying of mineral soil has occurred. The same holds for lands from which peat has been stripped. However, it is probable that burning peat concentrates a number of elements including heavy metals such as Pb, Cd, Cu, Ni, V. and others. These could strongly impact on surface waters should substantial amounts of burn residues be moved by surface runoff into local streams and lakes. However, documentation of this kind of effect has not been found in the literature.

11.5 EFFECTS OF PEATLAND AGRICULTURE ON HYDROLOGY

The effects of farming peat soils on peatland hydrology are essentially the same as those described for the effects of drainage in Chapter 5. Drainage and development for agriculture increase storage capacity, regulate flow into peatland outlets over a longer period of time, and increase the baseflow. Increased moisture-holding capacity accounts for delayed flow of runoff from a drained area as compared to that from an undrained area (Lundin 1975; Burke 1968). Consequently, flow characteristics are much more uniform from drained areas. and hydrographs do not show the sharp peaks exhibited by undrained areas (Heikurainen 1979; Eggelsmann 1975). In a study of several river basins in Byelorussia, Klueva (1975) found that after agricultural development of peatlands, minimum summer daily runoff increased from 30 to 150%. During the winter, the minimum flow increased in half of the basins studied.

Evapotranspiration from marshes after removing vegetation has been shown to be reduced by 40 to 50% (Bulavko 1971). When the marshes are reclaimed and planted with crops, evaporation will increase slightly but it will remain 10 to 15% below its original value. As a consequence, the annual runoff will approach its original value (Bulavko and Drozd 1975; Zubets and Murashko 1975).

It was noted previously that stripping and burning peat has been practiced in some localities in Western Canada. In these situations the underlying mineral soil is farmed, and hydrologic responses will likely become similar to those of surrounding non-organic mineral soils.

11.6 CURRENT AND FORECAST EXTENT OF PEATLAND AGRICULTURE IN ALBERTA

Data on the areal extent of peatlands being used for agriculture in Alberta are not available and are difficult to determine. The areas of organic soils in the agricultural parts of the province have been determined from the SIDMAP database and are presented in Table 11.1. SIDMAP refers to the Soil Inventory Database for Management and Planning, a computer database which contains land resource and management data by quarter section (160-acre cells) for areas of Alberta covered by reconnaissance soil surveys. These areas include most of the agricultural areas of the province and only some of the forested areas. Except for the Peace River region, most of northern Alberta is not included in the database (Hiley et al. 1986). Various kinds of management information (crop type, yield, etc.) can be derived from SIDMAP, but studies of management practices on organic soils have not been carried out to date.

Table 11.1. Areas of Organic soils within municipalities in Alberta.

Location[b]	Quarter Sections[c]	Land Area in Municipality (ha)	Organic Area (ha)	Organic Area (%)
County 1	702	556 220	37 111	6.7
County 3	800	301 227	41 996	14.0
County 7	1161	216 363	64 485	29.9
County 9	8	348 163	433	0.1
County 10	937	336 050	50 173	15.0
County 11	1544	218 065	84 051	38.6
County 12	3566	450 540	194 661	43.3
County 13	1121	267 032	63 278	22.5
County 14	140	284 414	7697	2.7
County 17	283	376 184	16 320	4.3
County 19	888	353 267	49 986	14.2
County 20	62	148 012	3579	2.4
County 21	1	268 402	59	0.02
County 23	307	412 711	16 543	4.0
County 24	49	551 716	2858	0.5
County 25	663	351 165	35 681	10.2
County 28	1871	296 466	104 196	35.7
County 30	117	249 288	5548	2.2
County 31	1649	326 346	89 522	27.5
I.D. 5	1	20 715	35	0.2
M.D. 99	4184	891 527	231 713	12.3
I.D. 13	55		2847	
I.D. 14	11 659	2 359 880	599 736	25.5
I.D. 15	4838	825 123	271 266	33.0
I.D. 16	13 213	3 393 760	647 344	19.1
I.D. 17	5677	7 123 674	314 176	4.4
I.D. 18	5714	7 259 477	316 215	4.4
I.D. 19	636	287 517	36 455	12.7
I.D. 20	545	590 145	29 050	4.9
I.D. 21	1003	756 971	53 149	7.0
I.D. 22	3086	741 659	167 726	22.7
I.D. 23	12 910	8 420 452	709 281	8.4
M.D. 31	14	381 088	475	0.1
M.D. 87	253	239 781	13 995	5.9
M.D. 90	316	2 065 558	17 234	0.8
M.D. 92	1430	318 841	77 331	24.3
M.D. 130	503	284 014	28 141	9.9
M.D. 133	26	69 452	1488	2.1
M.D. 135	30	88 867	1635	1.8
M.D. 136	96	140 305	5073	3.6

[a] Data provided by R. Wehrhahn and L. Marciak, Alberta Agriculture, Edmonton, 1986; derived from SIDMAP database (Hiley et al. 1986).

[b] See Figure 10.1 for map of municipalities in Alberta.

[c] Number of quarter sections where Organic soils are either dominant (40-100% of area) or significant (20-40% of area); quarter section is based on 145 acres (59 ha) to account for roads, dwellings, etc; excludes all Indian Reserves, Metis Colonies, Military Reserves, Provincial Parks and urban lands.

Figure 11.1. Municipality
map of Alberta.

The data in Table 11.1 show that the area of organic soils in some municipalities is quite high, with the largest percentage occurring in County 12 (Athabasca). Municipalities with large areas of organic soils are those occurring along the agriculture-forest fringe. It follows that these municipalities also have the greatest potential for pollution of surface waters through agricultural use of peatlands. There could also be many situations where the surface waters of a county may be affected by peatland clearing and farming practices in neighbouring counties. However, there do not appear to have been any studies carried out on effects of peatland agriculture on water quality in these areas. The actual impacts of farming organic soils on water quality in these areas cannot, therefore, be evaluated. The only conclusion which can be made is that a number of the effects recognized in other parts of Canada and in other countries are likely to occur in Alberta's farmed peatland areas as well.

The extent or degree of impacts of peat farming on water quality in a region would likely be proportional to the area of peatland farmed. This again is difficult to evaluate due to the lack of land use data as noted previously. An analysis of one county with extensive peatland areas may provide a general indication of peat farming impacts. The County of Barrhead (No. 11) has an estimated 84 259 ha of peatlands accounting for 38.6% of the county area (Table 11.1). The proportion of peatlands suitable for agricultural development in this county has been estimated at 20% (M. McLenaghan, Alberta Department of Forests, Lands and Wildlife, Public Lands Division, Barrhead; 1987, personal communication). Thus, about 16 850 ha, or about 287 quarter sections, of organic soils may eventually be farmed in the County of Barrhead. Information about the actual acreages of peatland currently being farmed in the County of Barrhead is not available. An estimate provided by M. McLenaghan (personal communication, 1987) is that 3000 to 4000 acres (1200 to 1600 ha) of organic soils are currently being farmed. This represents less than 10% of the area potentially suitable for agriculture. The conclusion that can be drawn from this is that if there are any water quality impacts due to farming organic soils at present, these impacts could be considerably greater in the future as up to 10 times the current area of organic soils could be brought into agricultural production. If these figures are used to extrapolate to other municipalities listed in Table 11.1, a crude estimate of the total area of peatland that could be used for farming is 1 million ha. As such, agricultural uses of peatlands potentially have large and widespread impacts on regional hydrology and quality of associated waters.

Some factors can mitigate the impact of agricultural peatland development on water quality. The 20% or so of peatlands within agricultural areas of the province regarded as being suitable for farming consist predominantly of shallow organic soils. Impacts of initial clearing, draining, and breaking activities on these soils would be similar to those on deep organic soils. However, the bottoms of drainage ditches would commonly reach mineral soil, and the plough layer would include mineral subsoil within a few years. The contact of drainage waters with mineral soil would result in increased buffering capacity and reduce the acidifying effects of peatland waters. This would be the main effect of contact with mineral subsoil, accompanied by secondary influences that the changed pH could have on reactions and forms of trace elements and humic materials in the water.

11.7 SUMMARY

Improved pasture, forage crops, grain crops, and rough grazing and rangeland are the main forms of agriculture practised on peatlands in Alberta. Reports from other provinces and countries indicate that the main effects of drainage and development are subsidence and oxygenation of the surface layers, and increased levels of particulate and dissolved organics, as well as oxidation products in the runoff waters.

The rate of subsidence is influenced by biological oxidation, height of water table, character of organic material, compaction, burning, wind and water erosion, shrinkage and dehydration, and the cropping system used. Nutrient loading in the runoff waters may occur partially because of the mobilization of minerals, but would more likely occur because of fertilization. Organic carbon levels and suspended matter will increase in drainage waters mainly during operations such as clearing and ditching which drastically disturb the soil.

The SIDMAP database can be used to determine areas of organic soils within municipalities. However, data on the areal extent of peatlands currently being used for agriculture in Alberta are not available. The total area of organic soils suitable for farming is possibly in the area of 1 million hectares. In general, it appears that the proportion of peatlands that is suitable for agriculture and that is currently being farmed is quite low, probably less than 10%. If a somewhat larger proportion of the potentially arable peatlands were brought into production, the effects on hydrology and water quality could be significant.

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12. EFFECTS OF MINING, LINEAR DEVELOPMENTS, AND OTHER ACTIVITIES ON PEATLANDS AND ASSOCIATED WATERS

L.W. Turchenek

12.1 HORTICULTURAL PEAT MOSS AND FUEL PEAT MINING

12.1.1 Horticultural Peat Moss Operations in Alberta

About 50 000 t of horticultural peat moss were produced annually during the mid-1980s in Alberta. In 1985, this accounted for 8.5% of the total Canadian production and was valued at more than $10 million (Prud'homme 1985). The peat moss is produced in central Alberta at production plants near Seba Beach, Evansburg, Smoky Lake, and Alpen Siding. Although there are few producers, three different methods are used in extracting the peat and these have potentially different impacts on downstream aquatic environments. One of these is a wet mining method whereby peat is extracted in situ with a backhoe mounted on a barge. The peat is then pressed, dried in natural-gas-fired rotary driers, and baled for market (Hood 1985). Since prior drainage isn't required and peat waters are substantially returned to the peat extraction site, the impact on downstream waters is minimal.

The other two methods are similar except for the peat collection and drying processes. In the so called 'haku' method, the peat on a cleared and levelled bog surface is milled, left to dry, windrowed, loaded onto light trailers, and stockpiled or transported to a plant for further drying and packaging. In the vacuum harvester method, on which the majority of peat production in Alberta is based, milled peat is left to dry in situ until suitable for packaging; it is collected by vacuum and either baled immediately or stockpiled for future processing. A feature common to these two methods is that bog preparation involves clearing and drainage operations to obtain a relatively dry surface. Ditch runoff from the drained bogs can therefore have an impact on receiving streams or other water bodies.

12.1.2 Potential Impacts on Hydrology and Water Quality

Few studies on the effects of peat mining on water quality have been carried out. Some of these studies pertain to mining of peat to be used as fuel, but extraction methodologies for this and for horticultural peat are much the same. Some studies were reviewed in Chapter 7 and data are presented in various tables. The following are summaries of representative reports which demonstrate effects of mining on peat quality.

Clausen (1980) reported results of 2 years of water quality investigations of the runoff from four peatland watersheds. Two of the peatlands were drained, one under cultivation and one being mined, while the other two are undisturbed. Both the cultivated and mined peatlands had higher levels of colour, suspended sediment, acidity, K, Fe, Al, and Na. For both disturbed and undisturbed peatlands, spring minimum values occurred for pH, alkalinity, specific conductance, Ca, Mg, N, and colour. Maximum values of suspended sediment occurred during the spring.

Clausen and Brooks (1983a, 1983b) conducted a hydrochemical study of Minnesota peatlands, initially to define the water quality of streams draining peatlands prior to development, and subsequently to investigate impacts of peat mining on water quality. A multiple watershed approach was used whereby the frequency distributions of water quality constituents were used to detect whether runoff from a mined bog differed from that of 15 unmined (control) bogs. Peat mining increased water temperature, suspended sediment, specific conductance, and concentrations of acidity, Fe, Na, and N species, although drinking water standards were not exceeded. The elevated levels of some nutrients could contribute to eutrophication of downstream waters; one of the most important nutrients in this regard is P but its levels did not increase as a result of mining.

Largin et al. (1976), in a study of the effects of mining and drainage in the USSR, found increases in pH, Ca, Mg, HCO_3, SO_4, and humic and fulvic acid concentrations for bogs, and increases in pH, Mg, SO_4, NO_2, and humic and fulvic acids for fens. Chlorine and

NO_3 were reduced after mining the bogs while concentrations of Ca and HCO_3 decreased after mining fens. The pH, Fe, Ca, Mg, and humic and fulvic acid concentrations increased as time since drainage increased. These increases were considered to result from increased aeration. Seasonally, summer maximum concentrations were observed for Fe, Ca, Mg, and humic and fulvic acids; fall maximums occurred for HCO_3; and spring minimums in various parameters occurred due to snowmelt dilution.

Sallantaus and Patila (1983) monitored the water quality of runoff from four artificial peatland watersheds which were drained totally for peat mining. A peatland area in the natural state was used as a reference basin for water quality. There was evidence that drainage could increase pH values in runoff water up to several pH units in situations where drainage ditches encounter minerotrophic sedge peat layers under ombrotrophic *Sphagnum* peat. During peak flows there was less increase in pH. In one of the peatlands, the average concentration of P was 10 times and the specific discharge (120 kg P km^2) 20 times that of the reference peatland during the first 12 months after drainage. There were no distinct differences in the concentrations of dissolved organic matter, but particulate organic matter content was clearly higher than the usually negligible values in natural peatlands. However, specific discharge of particulate organic matter was less than that of dissolved organic matter. Most of the particulate organic matter loading occurred during the ditch excavation period. The investigators suggested that discharge of suspended solids could be considerably reduced by avoiding peak flow periods.

Korpijaakko and Pheeney (1976) investigated the effects that mined peatlands might have on the adjacent aquatic environment. Drainage channel samples were compared with those from the natural streams into which the waters were developed both above and below the point of discharge. Water analyses consisted of pH and light transmittance. Samples of bottom sediments were identified and their quantities determined. The pH values ranged from 3.96 to 4.20 on the bogs proper and up to 5.45 in natural drainage channels on the bogs. The pH values of the river waters, some of which were near neutral (6.2 to 7.0) and others somewhat acidic (4.2 to 5.2), did not appear to be affected by the bog runoff waters. Peat sediment did not seem to increase in adjacent streams. Light transmittance was higher in drainage ditches than in surrounding streams or in stagnant bog water. This suggested that efficient filtering had occurred during passage of the waters from the peat profile to the drainage ditch. The authors provided several recommendations for conducting mining operations without disrupting the ecological balance of the surrounding environment. The major one regarding peat sediments was that drains should not open directly into surrounding streams, but drainage water should be forced to pass through catchment areas were sediment would be filtered and water would percolate through peat and discharge into ditches.

Osborne (1983) reported preliminary results of water quality monitoring of a drained and mined bog in Newfoundland. In addition to bog discharge samples, water upstream and downstream from the discharge point was sampled. Levels of turbidity, suspended solids, Fe, and Mg had increased in downstream waters as a result of discharge from the mined bog. The levels of these constituents exceeded Environment Canada water quality guidelines during some, but not all, months of the summer sampling period.

Washburn Gillis and Associates Ltd. (1983) reported on analyses of peat and waters from three bogs from Newfoundland, Ontario, and British Columbia. Conventional water quality parameters, heavy metals, and organics were measured. Except for Fe and Al, the concentrations of all parameters were at acceptable levels with respect to water quality guidelines which have been established to protect aquatic life and ensure that water may be safely used for drinking water, livestock, and irrigation. These data were obtained for bogs which had been drained in preparation for mining. However, an operating peat dewatering plant discharging into the original bog was found to have a high chemical oxygen demand in its recycle pond waters, and Fe and Al exceeded water quality guidelines. In several samples of bog waters, phthalate esters were found in concentrations which exceeded water quality guidelines to protect aquatic life from chronic toxic impacts.

Suspended sediment in stream waters was found to increase considerably due to peat mining activites in Newfoundland (Panu 1988). Of other water quality parameters examined in this study, colour, turbidity, Mn, and Fe were most commonly found to increase in streams receiving drainage waters as compared to upstream levels. However, recommended guideliness were exceeded in both situations. It was concluded that each outflow ditch from a fuel peat operation should be equipped with a sedimentation basin to control peat loading of receiving waters. It was recommended that excessive releases of Mn and Fe should be investigated further.

12.2 LINEAR DEVELOPMENTS

In much of northern Canada, linear developments such as seismic lines, pipelines, roads, and power transmission lines must be built through muskeg. Built up drilling and well pads are features associated with roads constructed in oil- and gas-producing areas.

The least drastic of these linear disturbances are seismic lines, or cut-lines, since they usually involve only the surface removal of vegetation. Pipeline rights-of-way are more severe due to physical disturbance resulting from excavation of the peat. In muskeg areas, seismic lines are cut mainly during the winter and the resulting physical change in the peat deposit would be minimal. The main effect could be a slight alteration of the hydrologic cycle whereby removal of tree and shrub cover could result in reduced evapotranspiration and increased runoff.

Effects of disturbances along hydroelectrical transmission corridors through peatlands have been investigated mainly in relation to impacts on vegetation. Magnussen and Stewart (1987) in northern Manitoba and Grigal (1985) in Minnesota noted that vegetation in a right-of-way did not change markedly in floristic composition. In the Manitoba study, water chemistry indicated a slight enrichment in the right-of-way environment. Thawing of permafrost and subsequent erosion had occurred only in areas of heavy traffic. However, use of the tree/shrub herbicide picloram in the right-of-way led to disappearance of non-targeted species such as *Sphagnum fuscum*.

Cut-lines and pipeline rights-of-way running through perennially frozen peatlands can result in more severe and long-lasting effects. The northern third of Alberta lies within the discontinuous permafrost zone (Brown 1977). Bogs in this area are commonly permanently frozen while fens, for the most part, thaw during the summer. Removal of vegetation results in greater insolation of the peatland surface and resultant thawing of ice. However, the thaw may not penetrate to a great depth. Thawing of peat in a pipeline right-of-way resulting in a slightly deeper active layer has been reported along gas pipelines in northern Alberta (Reid 1977). The water remains in place, however, and the hydrologic impact is thus localized.

Roads built through muskeg can have more damaging effects than cutlines or pipelines. They can directly disturb regional surface drainage and local vegetation. This can lead to indirect disturbance of the soil thermal regime, soil composition, chemical content of surface water, vegetation, and wildlife on a regional basis, as the ecosystem adjusts to a new equilibrium. When surficial flows of water perpendicular to a length of roadbed are greater than the ability of the roadbed to transmit them and the capacity of the upstream organic terrain to absorb them, ponding of water occurs in the upstream muskeg. This ponding can divert nutrient-rich water from fens or mineral terrain onto ombrotrophic bog terrain. Nutrients in this water alter the bog regime, destroying its characteristic vegetative cover and soil composition. In muskegs with permafrost, continued ponding of surface water will degrade underlying permafrost, resulting in development of thermokarst terrain. Several techniques are used to mitigate the effect of environmental impacts of roads built through peatlands, but some of these are inadequate or lose their effectiveness over time. As a consequence, drowning of peatland vegetation occurs frequently along roads (Pomeroy 1985).

The environmental effects of pipelines, roadways, and other developments have frequently been described in environmental impact assessments and related research reports concerning proposed and existing corridors for oil and gas transmission in the north (e.g., the Mackenzie Valley and the Arctic Gas pipeline routes). Comprehensive hydrologic balance and water quality studies have not been found, however, and only qualitative statements can be made about potential impacts. Roads and ditches can result in diversion of water flow, as noted above. This can result in changes in vegetation communities and, therefore, in wildlife habitat. Water quality parameters may be locally affected, but impacts on downstream waters are likely to be minimal due to dilution by other uncontaminated water sources.

Roads can also affect peatlands and water bodies through contamination by road dust. Spatt and Miller (1981) found that the growth of *Sphagnum* spp. was impaired due to road dust and road-related construction on the Alaska Pipeline haul road. Changes in vegetation communities resulting from dust accumulation, including salts applied to reduce slippery conditions and to hold down dust, have been noted along roads in Sweden (Backman et al. 1980). Backman et al. (1980) noted that the chloride content of water in small ditches was found to increase rapidly during occasional snowmelts in the winter. However, the salts were diluted to minimal levels during spring thaw. In swampy and marshy areas, the effects on vegetation and local hydrology due to ponding by roads far outweighed those due to deposited road dust.

A variety of elements can be deposited from airborne road dust on nearby peatlands. In New Brunswick, Santelmann and Gorham (1988) showed that concentrations of Al, Cr, Fe, La, Ni, Sc, Sr, and V in *Sphagnum* mosses decreased logarithmically away from a gravel road to a distance of 200 m. Concentrations of other elements were related to other sources; Na and Cl were related to deposition from precipitation enriched with sea spray while As, Cd, Pb, and Zn were attributed to air pollution.

12.3 Drastic Disturbances

Drastic disturbances are another category of anthropogenic activities which can have deleterious effects on peatlands and their associated waters. The main type of development within this category in Alberta is surface mining, and the major activity of this kind within areas of peatlands is open-pit mining of oil sands. This method of oil sands extraction involves the stripping of overburden in order to expose the oil-bearing formations. In a fairly substantial portion of the potentially mineable area in the Athabasca oil sands region, the overburden is characterized by a surface peat deposit. This peat deposit is removed and stockpiled with the intention of using it for reclamation purposes at a later time. Prior to removal of the peat and stockpiling, the peatland needs to be drained. Thus, there are potential impacts due to the initial drainage procedures and subsequent impacts resulting from the actual mining operations. Special protective measures are often required when a surface mining operation can have an impact on the environment beyond the boundaries of the mine. In the case of Syncrude Canada Limited near Fort McMurray, the part of Beaver Creek downstream from the mine boundary was diverted via other drainage ways to the Athabasca River (Syncrude Canada Ltd. 1973). Well planned measures such as these, together with continued monitoring of effects on ecosystems, can effectively minimize the impact of such developments.

Longer-term impacts of muskeg drainage should be considered in situations where several years lapse between drainage operations and development of the mine. The impact on hydrology and water quality of the Muskeg River was studied after a peatland was drained from the proposed mine site of Alsands Energy Limited, also in the Athabasca oil sands area. The site was drained in early 1980 and monitoring studies reported were mainly carried out later that same year. However, the site has not been developed for mining to date and the results of the earlier studies may or may not be currently applicable. These studies indicated that the effects on the Muskeg River would be minor for the most part. Delamore (1981) concluded, on the basis of theoretical considerations, that clearing and ditching of the Alsands 5-year mine site would have increased rainfall runoff from the area by about 20%. Because the mine site accounts for somewhat less than 1% of the Muskeg River drainage basin area, it was concluded that the impact of only a 20% increase in runoff should have had an insignificant impact on the river.

With regard to water quality, a considerable amount of data on the Muskeg River and its tributaries had been collected through the Alberta Oil Sands Environmental Research Program and was reviewed by Mayhood and Corkum (1981). Using these data for baseline information, Mayhood et al. (1981) concluded from the post-drainage monitoring studies that water quality changes arising from the addition of mine-site drainage water were usually slight, when they were detectable at all. The most pronounced effects of drainage on water quality of the Muskeg River were increases in suspended solids and turbidity. The most pronounced changes in these parameters occurred during a sudden flood discharge of heavily-silted water from a small lake into the Muskeg River which resulted from a failure of a ditch wall. Short-term increases in N and P concentrations occurred in mine-site drainage runoff when fertilizers were applied during reclamation of ditched areas. Thus, the impact of peatland drainage on water quality appears to be minimal except during unusual events such as those above. In the Alsands situation, it is also important to consider that peatlands occupied only a portion of the mine area. Drainage ditches may also have run through mineral soil areas and the resulting interaction of peat water with inorganic material could alter water chemistry. In particular, acidity and pH could be affected through reaction with neutralizing and buffering materials.

12.4 SUMMARY

Horticultural peat moss and fuel peat extraction, linear developments and drastic disturbances were reviewed in this chapter. Potential impacts of peat mining on drainage water attributes include higher levels of colour, suspended sediment, water temperature, acidity, N, K, Fe, Al, and Na. Linear developments such as seismic lines, pipelines, roads, power transmission lines, and drilling well pads can all affect peatlands to various extents depending on degree and magnitude of the disturbance. Effects include disturbance of surface drainage, surface vegetation, and soil thermal regime. Ponding of surface water can degrade permafrost and divert nutrient-rich waters into other neighbouring ecosystems such as nutrient-poor bogs. Major disturbances such as surface mining of oil sands require engineering measures to minimize impacts on surrounding wetlands and associated waters. Drainage of peatlands in the oil sands region prior to mining activities can have impacts such as increased turbidity, suspended solids and concentrations of N and P. However, the overall impacts in these situations are minimal where the drained area accounts for a small proportion of the area of the drainage basin.

Human influences on peatlands reviewed in this and preceding chapters were those that have potential for widespread, regional impacts. Other activities or events that can have local effects on peatlands were not reviewed. These include oil and brine spills, chemical spills, and other occurrences of this nature. Information on these is available in the literature, although some of it may be in unpublished reports and, therefore, difficult to locate.

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13. RECOMMENDATIONS FOR RESEARCH AND MANAGEMENT OF PEATLANDS AND ASSOCIATED WATERS

L.W. Turchenek

13.1 INTRODUCTION

Concern about the impacts of anthropogenic activities on the quantity and quality of waters associated with peatlands is apparent from the world literature. However, there have been only a few studies indicating which water properties are affected and to what extent, although they reveal some consistency in the kinds of water attributes affected. Furthermore, it has been recognized that runoff waters of natural, undisturbed peatlands can have negative impacts on the quality of downstream waters. In Alberta, there is a limited amount of baseline data and of research results by which impacts of runoff from both natural and developed peatlands on hydrology and hydrochemistry of downstream waters can be evaluated.

As a consequence of the paucity of information regarding Alberta's peatlands and associated waters, several recommendations are presented for enlarging the database so that some preliminary observations and conclusions can be made about the effects of both natural and developed peatlands on water quality and quantity. This first group of recommendations consists essentially of suggestions for data gathering and survey programs.

A second group of recommendations is made with regard to research into the dynamics of water and its constituents. These address concerns such as quantification of the water balance equation for peatland areas and investigation of the forms, interactions and movement of specific water constituents such as trace elements.

A third group of recommendations concerns the development of tools for evaluation of processes in peatland systems for identification of further research needs and for management purposes. This area mainly involves modelling of systems and processes. Modelling was not addressed in the literature review, but it was considered important to make some suggestions for application of this powerful tool in water management and research.

13.2 RECOMMENDATIONS FOR ACQUISITION OF BASELINE DATA ON PEATLANDS AND ASSOCIATED WATERS

Recommendation No. 1. **Identify peatland drainage basins with current or planned developments that can affect hydrology and hydrochemistry of associated waters.**

The purpose of this activity is to determine which drainage basins are or can be significantly affected by peatland development, and to assist in establishment of priorities for inventories, research, monitoring, or other activities related to peatlands and their development. This is an inventory type of activity whereby drainage basins in which peatlands being drained, farmed, used for forestry, mined for peat moss, receiving 'high' acid deposition loads, or modified in other ways are identified. This activity could be accomplished by map overlay techniques, either manually or by use of computer geographic information systems. Information requirements include drainage basin delineation, land disposition and use, and location and extent of peatlands. Map information about peatlands can be obtained from soil surveys, forest cover maps, and other sources as indicated in Chapter 2. This map information is currently available and of a generalized nature. These recommended activities are a prerequisite to carrying out the recommendations for more detailed peatland inventory and data collection as indicated below.

Recommendations No. 2. **Define the water quality of peatland surface and runoff waters, and examine the impacts of existing developments on water quality.**

There are few data in Alberta about the chemistry of waters within peatlands and of waters in streams and lakes draining peatlands. Routine water quality monitoring is carried out only for major watercourses and lakes. Small lakes and streams in headwater areas of drainage basins can be important as aquatic habitat and as sources of drinking water for wildlife and livestock. Some constituents of water may not comply with water quality guidelines in these small water bodies. The purpose of this recommended activity is to obtain baseline data to define the quality of water within peatlands, in peatland runoff and in water bodies downstream from peatlands. A suggested approach to obtaining this information is that of Clausen and Brooks (1983a) which basically involves a water quality survey of a large number of peatlands and drainage waters. Data can also be collected during the course of peatland surveys recommended above. Priority areas for investigation should be those areas of concern as identified in Recommendation No. 1.

Examining the effects of anthropogenic activities in peatlands on the quality of downstream waters requires a 'monitoring' as opposed to 'survey' methodology. The multiple watershed approach applied in Minnesota (Clausen and Brooks 1983b) is recommended. In this approach, a number of similarly sized peatland drainage basins, some natural and some developed, are selected for study and comparison. Sampling and measurement of water quality parameters is then carried out regularly over a period of one to several years. The results of such studies should be suitable for extrapolation to other peatlands and should provide quantitative information on relationships among factors which can be used as model inputs.

In the case of conducting environmental impact assessments of planned developments in peatlands, the multiple or paired watershed approaches are commonly used. This involves calibrating watersheds, carrying out development activities (leaving one or more basins as controls) and following up with monitoring and statistical analysis. In Alberta, it has been suggested that further work in determining the impacts of peatland drainage for forestry be carried out using this type of approach. Such work is part of the Wetland Drainage and Improvement Program of Forestry Canada and the Alberta

Forest Service (S. Takyi, Forest Research Branch, Alberta Forestry, Lands and Wildlife, 1989, personal communication). The same approach can be applied to agricultural utilization of peatlands, experimental acidification studies, and other investigations.

Recommendation No. 3. Develop a database of the distribution, extent, types, and characteristics of peat and peatlands in Alberta.

Reliable estimates of the kinds, areas, and distributions of peatlands are required in order to predict any natural or developed peatland influences of water quality and quantity. This information is obtained by conducting inventories, with priority given to those areas with existing or potential agricultural, forestry, peat extraction, or other uses of peatlands. It would be preferable that the boundaries of inventory projects coincide with natural drainage basin boundaries rather than municipal, land survey, or other political boundaries, which artificially divide drainage networks. Inventories carried out in this manner would facilitate the application of water balance or water quality models in evaluations of peatland water resources. The most useful inventory scales are relatively high, ranging from 1:20 000 to 1:50 000, but 1:100 000 scale inventory may be suitable for many types of management decisions and land use and impact evaluations. Appropriate methodologies for inventories using remote sensing and field survey methods have been well documented.

The peatland inventory should include a sampling and characterization component. Little information is currently available about the chemical and physical properties of peat. In particular, the concentrations of toxic elements and compounds, and their variations with depth and peat type, are important because some of these have been shown to be released in appreciable quantities from developed peatlands. Such data would enable estimations to be made of the total quantities of substances which could be released to surface waters as a consequence of development. Inclusion of properties important in acid-base buffering would enable more reliable evaluations of susceptibility to acid deposition. Data on the physical properties of peat deposits are required for calculations of water balance and for input into hydrologic models.

13.3 RECOMMENDATIONS FOR RESEARCH ON THE DYNAMICS OF PEATLAND WATERS AND THEIR CONSTITUENTS

Recommendation No. 4. **Determine the forms, interactions, and movement of toxic substances in natural and developed peatlands and their runoff waters.**

Various aspects of the chemistry of minor and trace elements and of toxic organic compounds are identified as an information gap not only in Alberta but on a global scale as well. Elements of particular interest include aluminum, mercury, iron, manganese, and zinc. Lead, cadmium, tin, and other elements should also be included in such investigations; they have not been shown to be released in significant quantities from peatlands, but in many studies reviewed, the detection limits were higher than water quality guidelines. Aspects of elements requiring study include mechanisms of release from the peat matrix to water, binding by dissolved organic matter, chemical forms of the elements, and their fate in downstream waters. Such information is required for examination of bioavailability and toxicity in water quality studies. Similar recommendations and approaches have been presented by Shotyk (1986).

Recommendation No. 5. **Define the role of organic substances in peatland drainage waters in relation to acidification and fate of trace and nutrient elements.**

Recent research has shown that humic materials depress primary productivity in lakes due to binding of certain essential nutrients rendering them unavailable to organisms. Likewise, toxicities of elements are not necessarily related to their total levels in solution as most or all of some elements can exist in relatively harmless, complexed forms. The nature and mechanisms of formation of the complexes need to be known in more detail, as does their ultimate fate which can involve different sedimentation mechanisms or eventual flushing out of the system. The acid-base chemistry of humic substances is also important in terms of providing buffering to external inputs of acidic and acid-forming substances. A summary of research recommendations related to organic materials in surface waters and to acid deposition was developed by Jones et al. (1986). Suggested studies for improving the understanding of 'brown waters' in wetlands and surface waters included determinations of complexation constants for aluminum and other elements, improved speciation models, seasonal and spatial variations in dissolved organic carbon (DOC) characteristics, mobility of DOC, variation in DOC with stream and lake order and morphometry, and role of organic substances in ion balance. It is suggested that Jones et al. (1986) be referred to for detailed recommendations. Because this is a very large area of investigation, priorities need to be established for any research carried out in Alberta. Likely priority areas would include interactions of DOC with toxic elements since these have been shown to be problematic in other areas. As with other recommendations, application of existing models using currently available data would be helpful in identifying specific knowledge gaps and research needs.

Recommendation No. 6. Determine the effects of acidic and acid-forming deposition on peatlands and associated waters.

Deficiencies in information and knowledge and some recommendations for action with regard to the impacts of acid deposition on peatlands and associated waters were outlined in Chapter 9 of this report. Other specific recommendations can be found in Wood (1989). The lack of information about impacts of acid deposition on wetlands has been indicated in reports of various acid deposition projects including the Alberta Government/Industry Acid Deposition Research Program. A major consideration in Western Canada relates to the need for evaluating the contribution and effects of dry acid-forming deposition on peatland systems, as the work to date has dealt mainly with wet acid deposition.

Recommendation No. 7. Quantify the hydrologic cycle for peatlands in Alberta.

Water balance studies of watersheds containing peatlands are required to attain understanding of the influences of peatlands on water yield, hydrologic responses to storm events, and various other relationships. The anthropogenic uses of peatlands can alter watershed hydrology. To quantify such effects, comparison studies of natural and developed peatlands are required. Although information is available from studies in other regions, the hydrologic responses documented should be verified for Alberta conditions. The information obtained should be suitable for extrapolation to other peatlands and should provide data for quantitative modelling of hydrologic responses to development. A hydrologic model using appropriate relationships and factors for Alberta conditions is also necessary for quantifying total loads of constituents in water quality studies.

13.4 RECOMMENDATIONS FOR DEVELOPMENT OF TOOLS TO ASSIST RESEARCH AND MANAGEMENT OF WATER RESOURCES IN PEATLAND AREAS.

Recommendation No. 8. Develop and apply water quality and hydrologic models for evaluating influences of peatlands on water quality, for determining research needs, and for management purposes.

The influences of peatlands on water quality could be effectively and efficiently investigated through the use of models, but currently used models should be reviewed to determine if they include, or could easily include, these unique influences. Carrying out this recommendation would involve selecting, adapting, merging, and validating models for application to peatland areas, or alternatively, modifying currently used models to account for peatland influences on water quality. Clarke-Whistler et al. (1984) have developed a preliminary model for peatland waters which includes hydrology, biogeochemistry, and toxicity components. The study of Schwartz and Milne-Home (1982a, 1982b), who examined the factors controlling the chemistry of stream water in terrain dominated by muskeg, may also be an appropriate modelling approach.

Recommendation No. 9. **Develop and apply methodology for examining peatland use impacts and for applying models on a geographic, or spatial basis.**

The impact of a peatland development on downstream waters partly depends on the extent of that development. A small development within a drainage basin may have a minimal effect on water quality whereas several small or a few large developments can have a cumulatively large impact. It is necessary to account for this spatial or geographic factor in the models indicated in the previous recommendation. This can be accomplished by merging models with baseline information in a geographic information system (GIS). Natural resources management is carried out through synthesis and analysis of large amounts of geographically based information. Such information should be updated, analysed, edited, modelled, and in other ways modified to meet specific needs. A water resources GIS containing hydrometric, wetland, hydrologic, and hydrochemical data would enable wetlands and water resources managers or researchers to integrate the data in a computer environment, and would provide an effective tool for analysing various peatland development scenarios.

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14. GLOSSARY

This glossary provides definitions of terms used in this report as well as some additional commonly used terms. Sources for the definitions are as follows:

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absorption, specific - (1) quantity of water recharged into a recharge well per unit time and per unit rise of head; (2) ratio of the quantity of water that can be absorbed by soil which contains retained water only, to the total amount of water when fully saturated, or to the total soil volume.

accretion, rate of - amount of infiltrated water per unit area and time.

acidity, active - the activity of hydrogen ion in water or in the aqueous phase of a soil, measured and expressed as a pH value.

acidity of water - amount of acid, given as millimoles of a strong base per liter of water, necessary to titrate a sample to a certain pH value.

acidity, total - amount of both weak and strong acids expressed as millimoles of a strong base necessary to neutralize those acids using, for example, phenolphthalein as indicator.

acrotelm - uppermost layer of a peat deposit, with variable water content, high hydraulic conductivity, periodic aeration, and intense biological activity.

active layer - (1) the top layer of ground, in areas underlain by permafrost, which is subject to annual freezing and thawing; (2) sometimes used as a synonym for 'acrotelm'.

aerobic - (1) having molecular oxygen as a part of the environment; (2) growing only in the presence of molecular oxygen, such as aerobic organisms; (3) occurring only in the presence of molecular oxygen, as applied to certain chemical or biochemical processes such as aerobic decomposition.

afforestation - conversion of bare land into forest land by planting of forest trees.

aggradation - the process whereby a stream deposits its excess load to its channel.

aliphatic - of or pertaining to a broad category of carbon compounds distinguished only by a straight, or branched, open-chain arrangement of the constituent carbon atoms. The carbon-carbon bonds can be saturated or unsaturated.

alkalinity - the buffering system or titratable base in water; the milliequivalents of hydrogen ions neutralized by a liter of water; often expressed as $CaCO_3$ in $mg \cdot L^{-1}$.

allochthony - refers to something being formed elsewhere and transported to the site in question.

allochthonous peat - peat of sedimentary origin. This peat is formed from the remains of plants brought in (mainly by water) from outside the site of deposition.

amorphous peat - the structureless portion of an organic deposit in which the plant remains are decomposed beyond recognition.

amphoteric - capable of acting either as a base or as an acid, for example, amino acids which contain both a basic group (NH_2) and an acidic group (COOH).

anaerobic - (1) having no molecular oxygen in the environment; (2) growing in the absence of molecular oxygen, such as anaerobic bacteria; (3) occurring in the absence of molecular oxygen, as in a biochemical process.

anion - an ion carrying a negative charge of electricity. The common peat soil and water anions are bicarbonate, carbonate, sulphate, chloride, and hydroxyl.

anthropogenic - having its origin in the activities of man.

aromatic - of or pertaining to organic compounds that resemble benzene in chemical behaviour.

ash - the loss on ignition of a peat sample; the percent of original material remaining as residue after heating at 450°C for 16 hours in an electric muffle furnace.

authigenic, autochthonous - formed in situ, or on the spot.

autochthonous peat - peat which has been formed in situ.

available nutrient - that portion of any element or compound in the soil that can readily be absorbed and assimilated by growing plants.

baseflow - flow of water in a stream when all storm water or snow melt has been routed through the system.

base saturation percentage - the extent to which the adsorption complex of a soil is saturated with exchangeable cations other than hydrogen and aluminum.

basin, drainage basin - drainage area of a stream or lake.

benthic - deriving from or occurring at the bottom of a body of water.

biogeochemical - a biochemical that is no longer part of a living organism and has undergone, or is undergoing, alteration in waters, soils, or sediments.

bitumen - a general name for various solid and semi-solid hydrocarbons.

BOD (biochemical oxygen demand) - index of water pollution which represents the content of biochemically degradable substances in the water.

bog - a peat-covered area or peat-filled wetland, generally with a high water table. The water table is at or near the surface. The surface is often raised or level with the surrounding wetlands, and is virtually unaffected by the nutrient-rich groundwaters from the surrounding mineral soils. Hence, the groundwater of he bog is generally acid and low in nutrients. The dominant peat materials are *Sphagnum* and forest peat underlain, at times, by fen peat. The associated soils are Fibrisols, Mesisols and Organic Cryosols. The bogs may be treed or treeless and they are usually covered with *Sphagnum*, feathermosses, and ericaceous shrubs.

brown moss peat - peat composed of various proportions of mosses of Amblystegiaceae (*Scorpidium*, *Drepanocladus*, *Calliergon*, *Campylium*), *Hypnum*, and *Tomenthypnum*.

calcium carbonate equivalent - the total inorganic carbon content of soil material expressed in terms of percent calcium carbonate ($CaCO_3$).

capacity, sediment- or load-carrying - maximum sediment quantity which can be carried by a channel.

capacity, water-carrying - maximum discharge capable of being conveyed in any cross section of a watercourse in unit time.

capillarity - phenomenon associated with the surface tension of liquids, particularly in capillary tubes and porous media where gas, liquid and solid interfaces meet; the attractive force between two unlike molecules, illustrated by the rising of water in capillary tubes of hairlike diameters or the drawing-up of water in small interstices.

capillary action - action due to capillarity, e.g. holding water in the soil against gravity.

capillary fringe - belt of subsurface water held above the zone of saturation by capillarity.

capillary rise - rise of water above the phreatic surface through the action of capillarity.

capillary water - water held in the soils above the phreatic surface by capillarity; soil water above hygroscopic moisture and below the field capacity.

carr - (1) transitional peatland with peat usually rich in wood, bark, and cone remains of deciduous or coniferous trees; (2) shrub-covered wetlands.

cation - an ion carrying a positive charge of electricity. The common soil cations are calcium, magnesium, sodium, potassium, and hydrogen.

cation exchange - the interchange between a cation in solution and another on the surface of any surface-active material in the soil such as clay or organic matter.

cation exchange capacity (total exchange capacity) - the total amount of exchangeable cations that a soil can adsorb. In SI units, it is expressed in centimoles positive charge per kg of soil ($cmol(+)kg^{-1}$).

catotelm - the lower portion of a peat deposit, with constant water content, low hydraulic conductivity, absence of aeration, and slight biological activity.

cellulose - A degradation-resistant polysaccharide composed of glucose subunits that is the major structural component of the cell walls of plants, accounting for more than 50% of the organic carbon in the biosphere.

climax vegetation - stable, self-perpetuating plant communities that are the end-products of plant succession.

colloid, soil - organic or inorganic matter having very small particle size and a correspondingly large surface area per unit of mass. Most colloidal particles are too small to be seen with the ordinary compound microscope.

conductance, electrical; conductivity, electrical - ability of water to conduct an electric current per unit area divided by the voltage drop per unit length; specific conductance (conductivity) refers to the electron flow between two 1-cm^2 electrodes, set 1 cm apart.

copropel - the brown or grey pulpy material formed from the dead remains of microscopic plants and animals in the top muds of eutrophic lakes and marshes.

Cryosolic Order - an order of soils in the Canadian System of Soil Classification. Cryosolic soils are mineral or organic soils that have perennially frozen material within 1 m of the surface in some part of the soil body. The mean annual soil temperature is less than 0°C. They are the dominant soils of the zone of continuous permafrost and become less widespread to the south in the zone of discontinuous permafrost; their maximum development in organic and poorly drained, fine textured materials. The Order has three Great Groups: Turbic Cryosol, comprising mineral soils that display marked cryoturbation and generally occur on patterned ground; Static Cryosol, mineral soils without marked cryoturbation; and Organic Cryosol, organic soils. Organic Cryosols are further subdivided into Subgroups on the basis of degree of decomposition of the peat, depth to mineral material, and presence of ice in the soil (see reference (1) for further details).

cumulo layer - a layer of sandy, silty, or clayey material in an Organic soil.

degradation - erosion of a stream's channel; the inverse of aggradation.

degree-days - the difference between the mean daily temperature and a selected standard temperature, accumulated daily over a period of time, such as the growing season.

deforestation - removal of forest.

density, bulk - mass of an oven-dry soil sample per unit gross volume (including pore space) expressed as $Mg \cdot m^{-3}$.

deposition - the accumulation of material left in a new position by a natural transporting agent such as water, wind, ice, or gravity, or by the activity of man.

detritus - dead organic matter (OM) and its associated microbial elements, particulate (POM) or dissolved (DOM).

diplotelmic - adjective describing a mire in which both acrotelm and catotelm are present.

discharge - the volume of streamflow passing a point during some period of time; usually expressed as cubic feet/second (cfs) or as $L \cdot s^{-1}$.

DOC (dissolved organic carbon), DOM (dissolved organic matter) - dissolved organic material not retained by membrane filters with pore size 0.45 um.

drainage - the removal of excess surface water or groundwater from land by natural runoff and percolation, or by means of surface or subsurface drains.

drawdown - (1) lowering of the water table or piezometric surface caused by the extraction of groundwater by pumping, by artesian flow from a borehole, or by a spring emerging from an aquifer.

dy - the sediment of dystrophic lakes, rich in humus and partially decayed organic matter.

dysic - pH <4.5 in all parts of the control section of an organic soil.

dystrophy - the condition in water in which decay is hindered and recycling of nutrients is slowed; there is a high loading of allochthonous organic matter, but a low level of autochthonous input; dystrophic lakes are heavily stained (brown water) and have a high content of humic substances.

ecosystem - the total living and nonliving components of a community.

edaphic - referring to the ground or soil, especially with reference to materials derived from them or their influence.

eluviation - the transportation of soil material in suspension or in solution within the soil by the downward or lateral movement of water.

ericaceous - of or relating to the heath family.

erosion - the wearing away of the land surface by running water, wind, ice or other geological agents, including such processes as gravitational creep.

euic - pH >4.5 in all parts of the control section of an organic soil.

eutrophic - term referring to peatlands that are relatively nutrient-rich; also refers to soils and waters with high nutrient content and high biological activity.

eutrophication - process by which waters become more eutrophic either as a natural phase in the maturation of a body of water or artificially (as by fertilization or pollution).

evapotranspiration - the combined loss of water from a given area and during a specific period of time, by evaporation from the soil surface and by transpiration from plants.

exchangeable cation - a cation that is held by the adsorption complex of the soil and is easily exchanged with other cations of neutral salt solutions.

fen - a fen is a peat-covered or peat-filled wetland with a high water table that is usually at or above the surface. The waters are mainly nutrient-rich, minerotrophic waters from mineral soils. The dominant peat materials are shallow to deep, well to moderately decomposed fen peat. The associated soils are Mesisols, Humisols, and Organic Cryosols. The vegetation consists dominantly of sedges, grasses, reeds, and brown mosses, with some shrub cover and, at times, a scanty tree layer.

fertility, peat - the status of a soil in relation to the amount and availability to plants of elements necessary for plant growth.

fiber, rubbed or unrubbed - the organic material retained on a 100-mesh sieve (0.15 mm) either with or without rubbing, except for wood fragments that cannot be crushed in the hand and are larger than 2 mm in the smallest dimension. Rubbed fiber refers to materials rubbed between the fingers ten times or processed in a blender.

fibric - organic materials containing large amounts of weakly decomposed fibers whose botanical origins are readily identifiable; in reference to organic soils, the amount of rubbed fiber retained on a 100 mesh (0.15 mm) sieve is greater than 40%.

Fibrisol - see Organic soil.

flark - usually elongated, wet, and muddy depressions in string peatlands; may be several hundred metres in length. On slopes flarks are narrow, only a few metres wide. On horizontal peatlands they may be a hundred or more metres wide. The flark's long axis is always perpendicular to the direction of the contours. The term is applied to any water-filled or sedgy area between strings.

floodplain - the land bordering a stream, built up of sediments from overflow of the stream and subject to inundation when the stream is at flood stage.

flow, overland - mass movement of water over the land surface resulting from saturated conditions in the upper profile, due to snowmelt or intense precipitation.

flow, pipe - occurs within systems of tunnels, shafts, and sinkholes along margins of peatland and steep rock slopes, or along the central axes of raised bogs; possibly due to shrinkage cracks caused by drought.

forb - a herbaceous plant which is not a grass, sedge, or rush.

forest peat - peat materials derived mainly from trees such as black spruce, and from ericaceous shrubs and feathermosses.

frost-free period - the period or season of the year between the last spring frost and the first autumn frost.

fulvic acid - that fraction of humic substances that is soluble under all pH conditions.

fungi - the allophytic plants that lack chlorophyll and are filamentous in structure; molds.

gelbstoffe - a complex mixture of natural compounds dissolved in seawater, characterized by light absorbance that increases with decreasing wavelength, giving a yellow colour to the water and causing a blue fluorescence when irradiated with long-wave UV radiation.

gilvin - a name proposed for the yellow humic materials that stain natural waters.

Great Group - a category in the Canadian system of soil classification. It is a taxonomic grouping of soils having certain morphological features in common and a similar pedogenic environment.

groundwater - water that is passing through or standing in the soil and the underlying strata in the zone of saturation. It is free to move by gravity.

growing season - period with soil temperatures over 5°C at a depth of 50 cm.

gyttja - semireduced, fine-grained, organic, profundal sediments of eutrophic lakes; predominantly coprogenic, grey-brown to blackish sediment.

haplotelmic - adjective describing a mire in which only the catotelm remains.

hardness - the quality of water that prevents soap from dissolving; mostly caused by Ca and Mg in solution.

hemicellulose - a degradation-resistant polymer of the sugar D-xylose, with side chains of other sugars; related structurally to cellulose by its similar linkage between the sugar subunits, but is somewhat less resistant to degradation than cellulose.

herb - any flowering plant except those developing persistent woody bases and stems above ground.

horizon, soil - a layer of soil or soil material approximately parallel to the land surface; it differs from adjacent genetically related layers in properties such as colour, structure, texture, consistence, and chemical, biological, and mineralogical composition. More detailed descriptions of horizons and layers may be found in reference (1) above.

humic - organic materials that are highly decomposed and contain little fiber; peat soil material having ≤10% rubbed fibres.

humic acid - that fraction of humic substances that is not soluble in water under acid conditions (below pH 2), but becomes soluble at higher pH.

humic substances - a general category of naturally occurring, biogenic heterogeneous organic materials that can generally be characterized as being yellow to black in colour, of high molecular weight, and refractory.

humification - the processes by which organic matter decomposes to form humus.

humin - that fraction of humic substances that is not soluble in water at any pH value.

Humisol - see Organic soil.

hummocky moraine - an area of knob and kettle topography that may have been formed either along a live ice front or around masses of stagnant ice.

humus - (1) the high-molecular-weight, polymeric fraction of organic matter that remains after most of the plant and animal residues have decomposed (it is usually dark coloured); (2) humus is also used in a broader sense to designate the humus forms referred to as forest humus; (3) all the dead organic material on and in the soil that undergoes continuous breakdown, change, and synthesis.

hydraulic conductivity - the proportionality factor in Darcy's equation that relates the velocity of water in a soil or other porous medium to the hydraulic gradient; the quantity of water that will flow through a unit cross-sectional area of a porous medium per unit time under a hydraulic gradient of 1 (100%) at a specified temperature; expressed as cubic metres per day through a cross section of one square metre under a hydraulic gradient of 1 at a temperature of 15.6°C.

hydromorphic - developed under conditions of excess moisture; hydromorphic soils are found in areas of poor drainage.

hydrophilic acids - the dissolved organic substances in natural waters that are organic acids and that are not retained by XAD resin at pH 2; similar to humic and fulvic materials, with more carboxyl and hydroxyl character and lower molecular weights.

hydrophyte, hydrophytic - a plant that grows in water, or in wet or saturated soils; water-loving.

hyetograph - (1) map or chart displaying temporal or areal distribution of precipitation; (2) graph displaying the intensity of precipitation versus time.

hypha, mycelium - threadlike filaments, branched or composing a network, that constitute the vegetative structure of a fungus.

impeding horizon - a horizon which hinders the movement of water by gravity through soils.

infiltration - the downward entry of water into the soil.

interception - process by which precipitation is caught and held by vegetation, then lost by evaporation without reaching the ground.

interflow waters - runoff water that infiltrates the surface soil and moves laterally toward streams; such water is ephemeral and shallow (above the main groundwater level).

lacustrine - originating in, or derived from, a lake.

lagg - the zone where water collects at the margin of a peatland, near the mineral ground of the surrounding site. The water is relatively rich in bases and supports a eutrophic type of vegetation, with the communities resembling those of a fen.

laminations - layering or bedding less than 1 cm thick in a stratified sequence.

landforms - the various shapes of the land surface resulting from a variety of actions such as deposition or sedimentation (eskers, lacustrine basins), erosion (gullies, canyons), and earth crust movements (mountains).

leaching - the downward movement within the soil of materials in solution.

lentic - referring to standing water, as in ponds and lakes.

ligand - a functional group, ion, or molecule bound to a central atom (e.g., metal) in a complex between chemical species called a coordination complex.

lignin - the most abundant, natural, aromatic organic polymer that is a major structural component of wood. There is no general agreement about the structure of lignin, however, it is known to lack a regular sequence of monomers. It contains phenolic, hydroxyl, and methoxyl groups; phenols are formed when lignin decomposes.

lime (in soil) - a soil constituent consisting principally of calcium carbonate, and including magnesium carbonate, and perhaps the oxide and hydroxide of calcium and magnesium.

limestone - a sedimentary rock composed of calcium carbonate.

limnic - peat formation occurring on or in deep water by free-floating or deeply rooted plants.

limnogenous - water derived from lakes or rivers.

limnology - study of freshwater lakes or reservoirs.

lithic layer - bedrock under the control section of a soil; in organic soils, bedrock occurring within a depth of between 10 cm and 160 cm from the surface.

littoral - referring to the marginal region of a body of water; the shallow, near-shore region; often defined by the band from zero depth to the outer edge of the rooted plants.

marl - loose, earthy deposit of calcium or magnesium carbonate, believed to have accumulated in freshwater basins fed by mineral water springs.

marsh - a marsh is a mineral or a peat-filled wetland which is periodically inundated by standing or slowly moving water. Surface water levels may fluctuate seasonally, with declining levels exposing drawdown zones of matted vegetation or mud flats. The waters are nutrient-rich. The substratum usually consists dominantly of mineral material, although some marshes are associated with peat deposits. The associated soils are dominantly Gleysols with some Humisols and Mesisols. Marshes characteristically show a zonal or mosaic surface pattern of vegetation consisting of unconsolidated grass and sedge sods, frequently interspersed with channels or pools of open water. Marshes may be bordered by peripheral bands of trees and shrubs, but the predominant vegetation consists of a variety of emergent non-woody plants such as rushes, reeds, reed-grasses, and sedges. Where open water areas occur, a variety of submerged and floating aquatic plants flourish.

mercaptan - any of a group of organic compounds containing an -SH moiety (thiol group).

mesic - organic materials at a stage of decomposition between that of fibric and humic materials; peat soil material with >10% and <40% rubbed fibers.

Mesisol - see Organic soil

mesophyte - a plant that grows under intermediate moisture conditions.

mesotrophic - containing a moderate amount of plant nutrients.

meteoric - originating from the atmosphere.

methoxyl - the functional group CH_3O.

micelle - a colloidal particle composed of organic molecules aggregated on the basis of solvent affinity; in polar media, micelles form with hydrophobic moieties on the inside and hydrophilic moieties directed outward.

microclimate - (1) the climate of a small area resulting from the modification of the general climate by local differences in elevation or exposure; (2) the sequence of atmospheric changes within a very small region.

microrelief - small-scale, local differences in relief, including mounds, swales, or hollows.

millisiemen (ms) - one one-thousandth of a siemen; a unit of electrical conductance, the reciprocal ohm.

minerotrophic - a supply of water to vegetation originally derived from mineral soils or rocks but sometimes via lakes or rivers as intermediates; it may be eutrophic, mesotrophic, or oligotrophic.

mire - (1) an English word which is, in the general sense, a term embracing all kinds of peatlands and peatland vegetation (bog and fen); (2) a section of wet, swampy ground; bog; marsh; wet, slimy soil of some depth; deep mud, etc.

mor - a humus form of well drained to imperfectly drained sites consisting of organic horizons sharply delineated from the mineral soil.

moraine - a mound, ridge, or other distinct accumulation of unsorted, unstratified glacial drift, predominantly till, deposited chiefly by direct action of glacial ice in a variety of topographic landforms.

morphology, soil - the physical constitution, particularly the structural properties, of a soil profile as exhibited by the kinds, thickness and arrangement of the horizons in the profile, and by the texture, structure, consistence and porosity of each horizon.

mottling - spotting and blotching of different colour or shades of colour interspersed with the dominant colour.

mounded - a type of microtopography consisting of small basins and knolls generally of less than 1 m relief.

muck - fairly well decomposed organic soil material relatively high in mineral content, dark in colour, and accumulated under conditions of imperfect drainage.

muskeg - a North American term frequently employed for peatland. The word is of Algonquin Indian origin and is applied in ordinary speech to natural and undisturbed areas covered more or less with *Sphagnum* mosses, tussocky sedges, and an open growth of scrubby trees.

oligotrophic - (1) designation for peatlands that are poor to extremely poor in nutrients and with low biological activity; (2) containing a small amount of plant nutrients; refers to waters low in nutrient loading with low primary production of organic material by algae and/or macrophytes. Growth in an oligotrophic water is often limited by low levels of phosphorus and nitrogen.

ombrogenous - produced by rain; refers to peatland areas that receive nutrients from precipitation.

ombrotrophic - a supply of nutrients exclusively from rain water (including snow and atmospheric fallout), therefore making nutrition extremely oligotrophic often in an unbalanced way.

order, soil - a category in the Canadian System of Soil Classification. All the soils within an order have one or more characteristics in common.

organic carbon, soil - the percent by weight of carbon in organic forms in soil materials, determined by the difference between total carbon (determined by dry combustion) and inorganic carbon (determined by acid dissolution).

organic matter, soil - the organic fraction of the soil; includes plant and animal residues at various stages of decomposition, cells and tissues of soil organisms, and substances synthesized by the soil population. It is estimated by multiplying the soil organic carbon content by 1.724.

Organic soil - an order of soils in the Canadian System of Soil Classification, that have developed predominantly from organic deposits. The majority of organic soils are saturated for most of the year, unless artificially drained, but some of them are not usually saturated for more than a few days. They contain 17% or more organic carbon, and: (1) if the surface layer consists of fibric organic material and the bulk density is less than 0.1 $Mg \cdot m^{-3}$ (with or without a mesic or humic Op less than 15 cm thick), the organic material must extend to a depth of at least 60 cm; or (2) if the surface layer consists of organic material with a bulk density of 0.1 $Mg \cdot m^{-3}$ or more, the organic material must extend to a depth of at least 40 cm; or (3) if a lithic contact occurs at a depth shallower than stated in (1) or (2) above, the organic material must extend to a depth of at least 10 cm.

Peat soils that have permanently frozen layers are classed as soils of the Cryosolic order in the Canadian System of Soil Classification (q.v.).

Classification of Organic soils is based on a vertical section of the soil called a control section. The control section extends to a depth of 1.60 m or to a lithic or terric contact, and is further subdivided into tiers, or layers, as follows:

Surface tier - the upper 0.40 m of peat.
Middle tier - the tier below the surface tier, having a thickness of 0.80 m or extending to a lithic, terric or hydric contact.
Bottom tier - the tier below the middle tier, having a thickness of 0.40 m or extending to a lithic, terric or hydric contact.

Classification of Organic soils into Great Groups and Subgroups is based on the degree on decomposition of the material in the various tiers. The classes describing materials on the basis of decomposition degree are fibric, mesic and humic (q.v.). The Great Groups within the Organic soil order are described below. Descriptions of the numerous Subgroups can be found in 'The Canadian System of Soil Classification' (Agriculture Canada Expert Committee on Soil Survey 1987; citation in Chapter 2).

Fibrisols - Organic soils with dominantly fibric middle tiers, or middle and surface tiers if a terric, lithic, or hydric contact occurs in the middle tier.

Mesisols - Organic soils with dominantly mesic middle tiers, or middle and surface tiers if a terric, lithic, or hydric contact occurs in the middle tier.

Humisols - Organic soils with dominantly humic middle tiers, or middle and surface tiers if a terric, lithic, or hydric contact occurs in the middle tier.

Folisols - Organic soils with dominantly upland organic (folic) materials of forest origin

outwash - stratified detritus (chiefly sand and gravel) washed out from a glacier by meltwater streams and deposited in front of or beyond the terminal moraine of an active glacier.

palsa - mound of peat with a permanently frozen core; usually ombrotrophic; generally much less than 100 m across and from one to several metres high. Palsas may be treeless or may contain a few stunted tamaracks or black spruce (wooded palsa). Palsa growth is due to the buildup of segregated ice mainly in the mineral soil.

paludification - formation of mire systems over previously forested land, grassland or even bare rock, due to climatic or autogenic processes. The literal meaning is 'swamping'.

parent material - the unconsolidated and more or less chemically weathered mineral or organic matter from which the solum of a soil has developed by pedogenic processes.

particle size distribution - the amounts of the various soil separates (different size ranges of particles) in a soil or peat sample, usually expressed as weight percentages and determined by sedimentation, sieving, micrometry, or combinations of these methods.

peat - material constituting peatlands, exclusive of the live plant cover, consisting largely of organic residues accumulated as a result of incomplete decomposition of dead plant constituents under conditions of excessive moisture (submergence in water and/or waterlogging).

peatland - a generic term including all types of peat-covered terrain.

pedogenic - pertaining to the mode of origin of the soil, especially the processes or soil-forming factors responsible for the development of the solum.

pelagic - referring to open water regions not directly influenced by the shore and bottom; limnetic.

peptization - to bring into colloidal suspension.

perched water table - a water table due to the 'perching' of water on a relatively impermeable layer at some depth within the soil. The soil within or below the impermeable layer is not saturated with water.

percolation - the downward movement of water through saturated or nearly saturated soil.

permafrost - (1) perennially frozen material underlying the solum; (2) a perennially frozen soil horizon.

permafrost table - the upper boundary of permafrost, usually coincident with the lower limit of seasonal thaw.

permeability, soil - the ease with which gases and liquids penetrate or pass through a bulk mass of, or a layer of soil or peat. Because different soil horizons vary in permeability, the specific horizon should be designated.

pH - the negative logarithm of the hydrogen-ion activity of a solution or of soil or peat. The degree of acidity or alkalinity of a soil or peat sample is determined by means of a glass, quinhydrone, or other suitable electrode or indicator at a specified moisture content or soil-water ratio, and expressed in terms of the pH scale.

pH-dependent cation exchange capacity - the difference between the effective cation exchange capacity and the cation exchange capacity of a soil measured at a pH higher than that of its natural value.

phenolic - of, relating to, or containing a phenol group which is a hydroxyl group bonded directly to an aromatic ring structure.

phytoplankton - small, free-floating aquatic microorganisms dwelling in oceans, lakes, rivers, and large streams, which are capable of photosynthesis; e.g., algae and Cyanobacteria.

plankton - small, free-floating aquatic organisms, formed mainly of water, carbohydrates, and proteins; phytoplankton is generally more abundant than zooplankton.

polyelectrolyte - a macromolecule containing multiple ionic functional groups (either cationic or anionic).

polygon - patterned ground caused by permafrost with recognizable depressions along polygonal circumferences.

POC (particulate organic carbon), POM (particulate organic matter) - all particulate organic material retained by a membrane filter with pore size of about 0.45 μm.

porosity, soil - the volume percentage of the total bulk not occupied by solid particles.

profile, soil - a vertical section of the soil through all its horizons and extending into the parent material.

pyrophosphate index - an indicator of the degree of decomposition of peat materials measured by extracting samples with sodium pyrophosphate and determining the optical density of the extracts. The greater the optical density, the higher the index and the degree of decomposition or humification.

reaction, soil - the degree of acidity or alkalinity of a soil, usually expressed as a pH value. Descriptive terms commonly associated with certain ranges in pH are: extremely acid, less than 4.5; very strongly acid, 4.5 to 5.0; strongly acid, 5.1 to 5.5; moderately acid, 5.6 to 6.0; slightly acid, 6.1 to 6.5; neutral, 6.6 to 7.3; slightly alkaline, 7.4 to 7.8; moderately alkaline, 7.9 to 8.4; strongly alkaline, 8.5 to 9.0; and very strongly alkaline, greater than 9.0.

recharge - process, natural or artificial, by which water is added from outside to the zone of saturation of an aquifer, either directly into a formation or indirectly by way of another formation.

refractory - not easily degraded.

relief - the difference in elevation between the high and low points of a land surface.

riparian - referring to the stream-side; the land bordering streams.

runoff - that part of precipitation that flows towards the stream on the ground surface (surface runoff) or within the soil (subsurface runoff or interflow).

sapropel - a slowly decomposing, black, unconsolidated, jellylike sludge comprising plant remains (mostly algae), found in anaerobic conditions in the sediments of lakes and seas.

sedge peat - peat composed mostly of the stalks, leaves, rhizomes, and roots of sedges (*Carex* spp.).

sedimentary peat - a material composed of plant debris and faecal pellets less than a few tenths of a millimetre in diameter and having brown or gray-brown colours when dry. It has slightly viscous water suspensions, is slightly plastic but not sticky, and shrinks upon drying to form clods that are difficult to rewet. It has few or no plant fragments recognizable to the naked eye.

sedimentation - process of settling and depositing by gravity of suspended matter in water.

seepage, soil - the emergence of water from the soil along an extensive line of surface.

shallow water - semipermanent to permanent standing or flowing water with relatively large and stable expanses of open water, which are locally known as ponds, pools, sloughs, shallow lakes, bays, lagoons, oxbows, impoundments, reaches, or channels. Shallow waters are distinguished from deep waters by the upper 2 m limit, although depths may occasionally exceed this during periods of abnormal flooding. During droughts, low water, or intertidal periods, drawdown flats may be temporarily exposed. Included in this class are all basins in which summer open water zones exceed 8 ha in size, regardless of the extent of bordering wetlands. These shallow water units are delineated from wetland complexes by the outer border of floating vegetation mats or by midsummer surface water levels, usually expressed by peripheral deep marsh emergents or shrubs. All other wetland basins less than 8 ha in area, with summer open water zones occupying 75% or more of the basin diameter, are classed as shallow water. The margins may be unvegetated, or may have rooted emergent vegetation including trees if confined to a narrow margin occupying no more than 25% of the basin diameter. Vegetation, if present in the open water zone, consists only of submerged and floating aquatic plant forms.

shrub - a woody perennial plant differing from a tree by its low stature and by generally producing several basal shoots instead of a single trunk.

silvic - pertaining to organic soils developed in forest peat; used in describing organic soil families.

soligenous - referring to peatlands with water percolating through them and/or carrying minerals into the peatland from outside sources.

solum - upper horizons of a soil in which parent material has been modified and in which most plant roots are contained; usually consists of the A and B horizons.

solution, soil - the aqueous liquid phase of the soil and its solutes consisting of ions dissociated from the surfaces of the soil particles and of other soluble materials.

sphagnic - pertaining to Organic soils developed in peat derived mainly from *Sphagnum* spp.; used in describing organic soil families.

Sphagnum peat - peat consisting mainly of *Sphagnum* spp.; usually poorly decomposed and raw; may also contain *Eriophorum* spp., *Carex* spp., and ericaceous species.

spring - a place where groundwater flows naturally from a rock or the soil onto the land surface or into a body of surface water.

stratigraphy, peat - a vertical sequence of layers of different materials within a peat deposit. Differences may be due to floristic composition, state of decomposition, or incidence of extraneous materials.

stream order - the position a section of a stream occupies in relation to the tributaries contributing to it; the higher the order, the more tributaries it has.

string - in string peatlands, the elevated, better-drained portion supporting mosses, sedges, brush, or trees; narrow, usually with its long axis across the slope; may form into net patterns.

string peatland - gently sloping peatland complex consisting of strings up to 2 m high, alternating with flarks (usually wet or water-covered, trough-like depressions, oriented across the slope of the peatland and perpendicular to the water movement). These may be parallel or occur in a webbed, sinuous, or net-like pattern. In flat watersheds, with no slope, the flarks become enlarged to pools of irregular size and shape.

Subgroup, soil - a category in the Canadian System of Soil Classification. It is a subdivision of the Great Group, and therefore is defined more specifically.

subsidence - lowering in elevation of a considerable area of a peatland surface due to loss of organic matter by biological oxidation, compaction and, shrinkage due to water removal.

surface tension - surface energy per unit area at the interface of two phases (e.g. air and water) which produces capillarity.

swamp - a peat-filled area or a mineral wetland with standing or gently flowing waters occurring in pools and channels. The water table is usually at or near the surface. There is strong water movement from margin or other sources, hence the waters are nutrient-rich. If peat is present, it is mainly well decomposed forest peat underlain at times by fen peat. The associated soils are Mesisols, Humisols, and Gleysols. The vegetation is characterized by a dense tree cover of coniferous or deciduous species, tall shrubs, herbs, and some mosses.

telluric water - also known as minerotrophic; originating from the earth.

telmatic - peat formation at the water table due to plants growing under conditions of periodic flooding.

terrestrialization - formation of a mire system by filling of a water body with organic remains, usually by gradual extension of peat-forming communities outwards from the shoreline of a lake.

terrestric - peat formation above the general water table.

terric - unconsolidated mineral soil.

terric layer - an unconsolidated mineral substratum underlying organic soil material.

texture, soil - the relative proportions of the various soil separates (sand, silt, and clay) in a soil.

till - unstratified glacial drift deposited directly by the ice and consisting of clay, sand, gravel, and boulders intermingled in any proportion.

topogenous - produced by relief, as a peatland complex determined by topography; indicates that the source of water for a peatland is the regional water table in a depression that predated peat formation.

topography - the surface features of a district or region, especially the relief and contours of the land.

trace element - chemical element present in a minor amount in water or soil.

transpiration - process by which water from vegetation is transferred into the atmosphere in the form of vapour.

trophic level - the relative nutritional position of organisms or populations within a food web; for example, all organisms that feed on algae are at the same trophic level.

trophic status - nutrient status; availability of nutrients to plants. See oligotrophic, mesotrophic, and eutrophic.

turbidity - condition of a liquid due to fine, visible material in suspension, which impedes the passage of light through the liquid.

unsaturated zone - the zone above the water table in an aquifer; the vadose zone.

volume weight - the bulk density a peat sample assumes after settling in a waterlogged condition; mainly determined on horticultural peats, and sometimes used as a substitute or estimate of the bulk density of undisturbed peat.

von Post humification scale - scale describing peat moss in varying stages of decomposition ranging from H1, which is completely unconverted, to H10, which is completely converted.

water capacity - the percentage of water remaining in organic soil material (peat) after having been saturated and after drainage of free water has practically ceased.

water content - volume of water per unit volume of peat. Maximum content is reached when all pore spaces are filled with water and peat is saturated.

water table - the upper surface of groundwater or that level below which the soil is saturated with water.

waterlogged - saturated with water.

water yield - quantity of water discharged under the action of gravity from a peat deposit saturated up to or beyond its normal moisture capacity, and lying above the water table.

water yield, coefficient of - ratio of the volume of water discharged from a layer of peat above the water table to the bulk volume of this layer, when the water table is lowered some distance from the surface of the peatland.

wetland - land having the water table at, near, or above the land surface or which is saturated for long enough periods to promote wetland or aquatic processes as indicated by hydric soils, hydrophytic vegetation, and various kinds of biological activity that are adapted to the wet environment. Wetlands include peatlands and areas that are influenced by excess water but which, for climatic, edaphic or biotic reasons, produce little or no peat. Shallow open water, generally less than 2 m deep, is also included in wetlands.

zooplankton - the fraction of the plankton community composed of animals.