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**ENVIRONMENTAL IMPACT OF PEAT
HARVESTING AND COMBUSTION UPON
AQUATIC ECOSYSTEMS - A REVIEW**

by

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EXECUTIVE SUMMARY

This paper presents a review of the available literature on the impact of drainage waters from peatlands upon receiving waters. Emphasis has been placed upon water chemistry and topics covered include inorganic and organic water chemistry, acidity, water quality, primary productivity of receiving waters, and the potential impact of peat combustion on the atmospheric transport of contaminants to water bodies.

Some concerns on peatland drainage appear to be unfounded including acidification of receiving waters, long-term increase of suspended solids, and eutrophication. Other areas need more research such as the possibility of increased mercury loading of receiving waters, aluminum toxicity, decreased phosphorus and metal bioavailability, and the possible presence of toxic organic compounds in drainage. These latter concerns are not strictly limited to the possible use of peat as an energy source, but are also applicable to forest drainage, horticultural peat extraction, development of northern reservoirs, and to naturally-acidic aquatic ecosystems.

RESUME ADMINISTRATIF

Le présent rapport passe en revue la documentation existante traitant de l'incidence des eaux de drainage des tourbières sur les eaux réceptrices. Nous avons attaché une importance particulière à la chimie de l'eau. Les sujets relevés comprennent la chimie organique et inorganique de l'eau, l'acidité et la qualité de l'eau, la productivité primaire des eaux réceptrices et les répercussions que peuvent avoir sur les nappes d'eau les retombées atmosphériques de contaminants produits par la combustion de la tourbe.

Certaines inquiétudes liées au drainage des tourbières ne semblent pas fondées, notamment l'acidification des eaux réceptrices, l'augmentation à longue échéance de la quantité de solides en suspension et l'eutrophisation. D'autres aspects nécessitent plus de recherches; c'est le cas de l'augmentation de la teneur en mercure des eaux réceptrices, de la toxicité due à la présence d'aluminium, de la réduction en phosphore et de la biodisponibilité des métaux ainsi que la présence de composés organiques toxiques dans les eaux de drainage. Les préoccupations précitées tiennent non seulement à l'exploitation de la tourbe comme combustible, mais s'appliquent aussi au drainage des forêts, à l'extraction de la tourbe à des fins horticoles, à l'aménagement de réservoirs dans le nord et aux écosystèmes aquatiques naturellement acides.

ABSTRACT

Two areas of environmental concern in peat development for energy use are the impact of peatland drainage upon receiving waters and the atmospheric deposition of potential contaminants from peat combustion. This paper will discuss the literature available upon the potential environmental impacts from these activities with emphasis upon metals, dissolved organics, naturally-occurring acids and sulphur.

RESUME

L'exploitation de la tourbe comme combustible suscite deux préoccupations environnementales: l'incidence du drainage des tourbières sur les nappes d'eau réceptrices et les retombées atmosphériques de contaminants issus de la combustion de la tourbe. Le présent rapport examine les documents publiés à ce jour au sujet des répercussions écologiques des activités précitées et se penche en particulier sur les métaux, les produits organiques dissous, les acides naturels et le soufre.

INTRODUCTION

Canada is estimated to have the largest area of peatland in the world, some 111×10^6 ha (Tarnocai, 1985). Yet, compared to countries such as Finland, the USSR and Ireland, the peat resource is relatively unused except for horticultural purposes. The use of peat as a fuel source for energy production has received new interest in Canada (Monenco Ontario Ltd., 1982). Such use would have a potential environmental impact depending upon the size of power plant. For example, a large facility of 600 MW capacity as exists in Finland and the USSR would combust approximately 5×10^6 tonnes of peat per year. This would require some 25×10^6 ha of peatland to be drained which requires 12×10^6 km of ditches to be constructed. To get the peat to the 50-55% water content necessary for combustion, 4 tonnes of water would be released into the environment per tonne of peat dried to this level (Lindstrom, 1980; Monenco Ontario Ltd., 1982). This is assuming that other dewatering methods are not utilized.

The purpose of this review is to examine current knowledge of the potential impact of these drainage waters upon the quality of receiving waters such as lakes and streams. Emphasis will be placed upon questions of the acidity of such waters, metal behaviour, nutrients and dissolved organics. In terms of biology, primary production of aquatic ecosystems will be considered. Also, potential sources of atmospheric inputs of metals from peat combustion will be

discussed. Due to space constraints, hydrological aspects of peatland drainage will not be considered in this review, nor will potential toxicological problems related to aquatic invertebrates and fish. In the latter case, research is lacking as will be discussed later.

THE INORGANIC WATER CHEMISTRY OF PEATLANDS

There are three main classes of peatlands: bogs, fens and swamps. Each of these classes has a unique hydrogeochemical regime which is reflected in the vegetation ecology of the peatland. In this paper, we will emphasize two types, the ombrotrophic bog and minerotrophic fens. The term ombrotrophic literally means rain-fed (Greek: ombros = rain, trophe = nourishment). The surface of bogs receive all water and inorganic constituents entirely from atmospheric deposition including precipitation and dustfall. In contrast, minerotrophic peatlands are fed by soligenous waters, i.e. ground and surface waters.

As a direct result of the hydrogeochemical differences between fens and bogs, the chemical composition of their surface waters are very different. Bog waters are quite acidic with pHs of approximately 4 and contain less than 2 mg/l Ca^{++} ion. Fen waters have a pH in the range of 6-8 and contain 5-50 mg/l Ca^{++} .

Bog surface waters are depleted in such major elements as Na, K, Ca and Mg relative to the composition of "normal" freshwater (Table 1). Since these elements are the products of weathering of

TABLE 1. Water chemistry from bogs (Luther, Ontario) and average fresh water (Bowen, 1979). All concentrations in ug/l.

Ion or Parameter	Concentration in Bog Water	Concentration in Ave. Freshwater
Na	500	6000
K	500	2000
Mg	1000	4000
Ca	2000	15,000
Al	400	50
Fe	500	40
Mn	30	8
DOC (mg/l)	50	5
pH	4	7

rock-forming minerals, and since bogs do not receive groundwater or surface runoff, the concentrations of their cations would approximate that of rain. Bog water is enriched in Fe and Mn by a factor of 8-10 relative to typical freshwaters (Table 1). Due to the abundance of decomposing organic matter, bog waters are characterized by reducing conditions. For example, Fe is reduced from Fe(III) to the much more soluble Fe(II) while Mn is reduced from Mn(IV) to the more soluble Mn(II). Bog waters are also enriched in Al by a factor of approximately 10, due to the increased solubility of Al at lower pH.

The speciation of metals in highly organic waters is of major interest as it is the concentration of a particular species, not the total dissolved concentration, that determines bioavailability. Some metals acquire enhanced toxicity as a result of the formation of organic complexes, e.g. Hg as will be discussed later. The subject of metal-organic complex formation in natural water has been recently reviewed by Shotykh (1984) and a detailed discussion of this subject is beyond the scope of this paper. It should be pointed out that bog waters are ideal candidates for the organic complexation of metals for several reasons. For example, major ions which tend to reduce ligand availability are relatively depleted in bog water. Certain metals which tend to form stable organic complexes, e.g. Al, Fe and Mn, are relatively enriched. Needless to say, the high concentration of dissolved organic matter (DOM) promotes this complex formation.

Another important factor controlling the concentration of dissolved metals is the solubility of various solid phases. The subject of mineral stability has been reviewed by Nesbitt (1984). This tends to be a complicated subject in bogs as depending upon the metal in question, a number of solid phases can form: oxides - Cr_2O_3 , hydroxides - $\text{Fe}(\text{OH})_3$, carbonates - PbCO_3 , phosphates - $\text{Fe}_3(\text{PO}_4)_2 \cdot \text{H}_2\text{O}$, or sulphides - ZnS . Native metals may also form such as Cu. A detailed discussion of this concept is beyond the scope of this paper except to note that a number of different minerals are stable in a peat-forming environment. For example, the high pH and oxygen conditions found in fens favours the formation of carbonate minerals including MnCO_3 , FeCO_3 , ZnCO_3 and PbCO_3 . Under conditions of high dissolved phosphorus, vivianite, $\text{Fe}_3(\text{PO}_4)_2 \cdot 8\text{H}_2\text{O}$, has been found to form. Under slightly reducing conditions, some metals such as V and Cr form stable oxides, while others such as U require strongly reducing conditions. In zones of active sulphate reduction, sulphide minerals may form. Copper will precipitate at pH 4 as the native metal before sulphate is reduced to sulphide, i.e. at a much higher oxygen partial pressure. More research is needed on the role of mineral formation in influencing the chemistry of peat waters.

THE ACIDITY OF BOG WATERS

Bog waters are acidic with pH values of approximately 4. The cause of this acidity is fairly easy to understand: such waters

contain abundant acids from the decomposition of organic matter but very few bases which result from weathering such as HCO_3^- . The acids, found both in bogs and fens include:

1. CO_2 which hydrolyzes in water to form carbonic acid;
2. low molecular weight organic acids such as acetic and oxalic which are products of metabolism;
3. higher molecular weight acids such as humic acid (HA) and fulvic acid (FA).

Fens, however, receive surface and groundwaters rich in bicarbonate ion and other weathering products of silicate and carbonate rocks. These ions consume protons thus increasing solution pH. This buffering mechanism is absent from bogs which do not contain bicarbonate ions.

A number of other chemical processes which affect the pH and buffering capacity of bog waters are discussed by Gorham et al. (1984). They maintain that cation exchange reactions at the bog surface, redox reactions at or below the water table, and interactions with DOM are the most significant processes which control bog acidity. In addition, precipitation ("acid rain") may be a significant source of H^+ ion to bogs and contains organic acids as well as strong mineral acids, e.g. sulphuric and nitric acid (Keene et al., 1983; Keene and Galloway, 1984).

Streams and rivers draining poorly buffered watersheds exhibit a marked seasonality in pH values. Minimum values generally occur during the spring high flow period while maximum pH values occur

during the summer dry period. This had been observed for rivers in Finland (Haapala et al., 1975), Norway (Webb, 1982) and Nova Scotia (Watt et al., 1983). The range of such pH values is from below 5 in the spring to over 7 in the summer. Changes in pH of bog drainage waters can also vary from the pH of the bog itself. For example, water from surface soils and bogs in southeastern Alaska show pH values below 4.7 but nearby streams have pH values greater than 6 (Johnson, 1981). We have measured lower pH values in bog surface waters (average of pH 3.9 in summer) from Barrington bog in Shelburne County, Nova Scotia while drainage water in Judas Creek, which is the natural drainage course, had pH values averaging 4.9 at the same time. During the spring wet season, the bog and drainage waters were the same pH, 4.0. The pH also increases downstream from the bog. During summer, values ranged from pH 4.1 to 4.5 along a 2 km sampling zone. In the spring, values increased from 4.0 to 4.2.

Others have observed this increased pH of drainage waters. Sallantausta and Patila (1983) monitored runoff water from several Finnish peatlands that were ditched and compared them to undisturbed bogs. The pH of drained peatlands ranged from 3.9-6.5 with a majority of values from 5.0-6.5. The control site had a lower pH range of 3.7-4.3. Ramberg (1981) found a similar pH increase in runoff waters from peatland forest drainage. This result is probably due to neutralization of acidified drainage water by mineral soils exposed when the peatlands were ditched.

In conclusion, a stream which receives drainage water from bogs will experience a range of acidities. Immediately after ditching, the stream will receive a "slug" input of acidic water. This will be ameliorated with time as the flow subsides and as various neutralizing processes occur. These include degassing of the previously-trapped bog water resulting in a loss of CO_2 and the neutralization of drainage water by mineral soils exposed by ditching. In any case, the ultimate pH values in the drainage stream will be higher than they were originally in the bog, and likely higher than they were before the bog was drained.

DISSOLVED ORGANIC MATTER IN PEATLANDS

Waters draining from peatlands are usually quite high in DOM content. This is true both for natural drainage and after ditching. We have observed values of dissolved organic carbon (DOC) ranging from 17 to 58 mg/l in Barrington Bog and Judas Creek, varying considerably with both season and runoff. Sallantausta (1984) monitored the drainage of both natural and mined peatlands in Finland. Summer and autumn DOM concentrations were somewhat elevated in drained watersheds compared to reference watersheds. In the spring, drained watersheds often had lower DOM. The annual mean DOM concentration was not found to be different. The author stated that, although mean concentrations were not affected by peat extraction, the situation immediately after

draining may be different because of the increased runoff (Sallantauss and Patila, 1983). They also state that the quality of organic matter may be affected by mining.

DOM in peatland waters is thought to be composed mainly of weak organic acids. These could be in the form of HA and FA which are weak acid polyelectrolytes (Stevenson, 1982) or acids with a more distinguishable molecular structure. The latter are not necessarily weak acids. Shotyk (1985) found concentrations of free, dissolved oxalic acid in bog pore waters up to 0.4 mg/l. This acid is amongst the strongest organic acids ($pK_1 = 1.25$, $pK_2 = 3.81$) and is stable over a large range of oxygen partial pressures. It also forms stable complexes with metals. Its presence suggests that other low molecular weight organic acids may also be present in bog waters.

Strong and weak acid concentrations are frequently estimated in natural waters using the Gran titration method. Glover and Webb (1979), Henriksen and Seip (1980) and Webb (1982) used this method to study weak acid concentrations in surface waters from Norway and Scotland. By their estimations, 10-30% of weak acids in these waters were organic. Oliver et al. (1983) studied the contribution of humic substances to the acidity of coloured natural waters. They isolated aquatic HA and FA from a wide variety of environments including bogs and determined their carboxyl contents by titration. An empirical relation was found between carboxyl content of the humic matter isolated from these waters and the DOC of the original water with a precision of $\pm 10\%$ which was felt to be a good estimate:

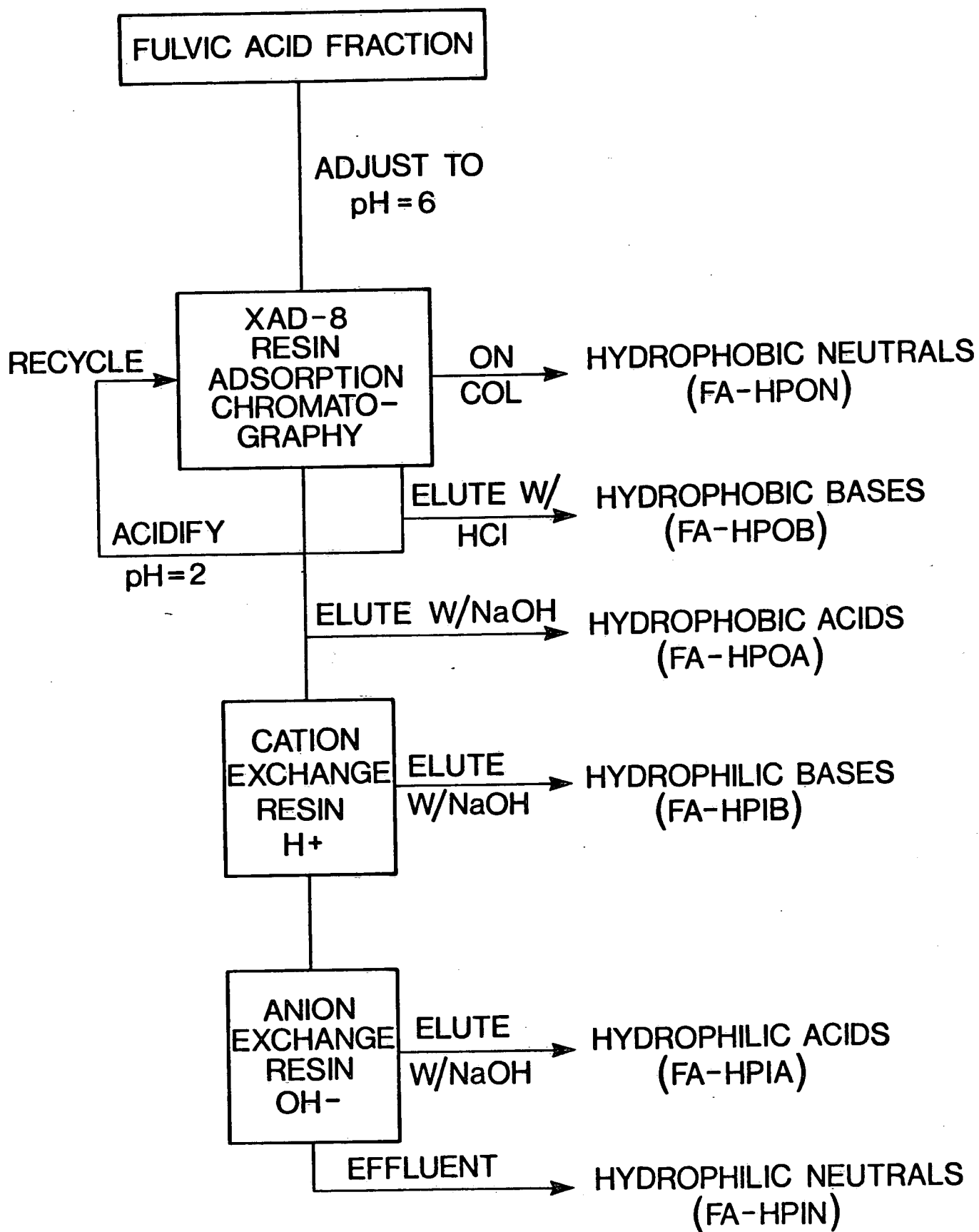
$$-\text{COOH} (\mu\text{eq/l}) = \text{DOC} (\text{mg/l}) \times 10$$

Another approach in studying the character of DOM in natural waters is to isolate and quantify the various fractions. We have used the method of Leenheer (1981). The HA fraction of the DOM is isolated from filtered water by precipitation at $\text{pH} < 2$ (Stevenson, 1982). The remaining DOM is, by definition, the FA fraction. This fraction is further fractionated into six sub-fractions as shown in Figure 1. The resulting seven sub-fractions are quantified by measuring their contribution to the original DOC of the filtered water. The sum of the acidic sub-fractions is represented in terms of %DOC by:

$$\text{HA}(\% \text{DOC}) + \text{FA-HPOA}(\% \text{DOC}) + \text{FA-HPIA} (\% \text{DOC}) = \text{Sum acid fractions} (\% \text{DOC}).$$

The upper layers of bog water from Barrington bog and from Judas Creek which drains it averaged approximately 65% of DOC as acidic fractions in the summer of 1984. Using the empirical relationship of Oliver et al. (1983), we estimate that this represents a carboxyl content of about 300 $\mu\text{eq/l}$. In the summer, bog and creek water differed in organic composition. One-third of the sum of acidic fractions was HA in creek water while the bog water was one-thirtieth. This discrepancy is likely caused by the greater proportion of deeper bog water in the drainage creek during the dry summer period. The sum of acidic fractions of bog water sampled from

FIGURE 1. DOM Fractionation Procedure After Leenheer (1981).



the lower, or humified layer also contained one-third HA and slightly more carboxyl groups (approximately 320 ueq/l).

We conclude that the character of bog-derived DOM which enters natural drains is affected by seasonal flows. We expect that this effect would be greater in the case of a drain which received water from a ditched peatland. This difference in character may affect the metal-binding capacity, colour and degradability of the DOM in drainage creeks.

EFFECT OF PEATLAND DRAINAGE ON WATER QUALITY

One of the main concerns about peatland drainage is the effect upon water quality in receiving bodies such as streams and lakes. This question is difficult to answer as many of the receiving waters are already highly organic due to natural drainage from undisturbed peatlands. Several studies have been made concerning this question. Verry (1975) investigated stream flow chemistry from undisturbed peatlands, both ombrotrophic and minerotrophic, in Minnesota, U.S.A. Concentrations of various water quality parameters including colour, specific conductivity, Fe, Ca, Cl, Na, Mg, Al, Zn, Si, total N and total P were inversely related to flow rate. In terms of yields (tonnes of nutrient exported per km² per year), peatlands were similar to forests growing on mineral soils. Bog waters were higher in total N and Al than fens, similar in total P, Fe, Cl and K,

but less in SO_4^{-2} , Na, Mg, Ca, Mn, Zn and Si than fens on an average annual flow-weighted concentration basis. Clausen (1980) studied the water chemistry of disturbed peatlands in Minnesota. He found elevated colour, suspended sediments, acidity, K, Fe, Al and Na during the first two years after drainage. Fens were higher in pH, alkalinity, specific conductivity, Ca and Mg than bogs.

The water quality of peatland drainage in New Brunswick was studied by Korpijaakko and Pheeney (1976). Suspended peat sediment was not increased in drainage water; in fact, the authors reported such waters were clearer than in adjacent streams. Of particular interest was the pH of drainage waters. In the bog, water pH ranged from 3.96 to 4.20. In the drainage water, pH of 5.45 was found. This decreased acidity has been previously discussed.

A review paper on the effects of peatland forest drainage in Finland was written by Heikurainen et al. (1978). They studied the water quality from undrained peatlands and those drained over the past 20-40 years. Waters from drained areas were higher in conductivity (=total ions) and pH, but equal in concentration in organic C, COD and total N. Suspended solids reached a peak approximately ten weeks after ditching, but became reduced afterwards. Sallantausta and Patila (1983) also investigated the effect of peatland drainage upon water quality of Finland. As found in previously-mentioned studies, suspended solid release was negligible except during excavation of ditches. Phosphorus was found to increase up to ten-fold in

concentration in some waters upon drainage, but not in all cases studied. The authors felt this could possibility lead to eutrophication. This latter effect could be enhanced if the drainage peatlands are subsequently fertilized to enhance forest production (Simola, 1983).

A potential environmental concern related to peatland drainage is the increased discharge of mercury. Simola and Lodenius (1982) studied the vertical distribution in a core taken from a lake in Finland. The core was dated by varve chronology which allowed a time resolution of one year. The increase of mercury began during the period 1942-1952. They found that mercury doubled in concentraion starting in the late 1960's with a flux rate that increased five times in the period 1967-1977 compared to the end of the 19th century. Since no agricultural or industrial sources were present, peatland drainage was considered the cause of the increased mercury loading. The proposed mechanism for this is increased dissolved organic carbon increasing the solubility of Hg. Increased solubility of Hg due to the formation of stable organic complexes was previously noted by Jackson et al. (1980) who concluded this could decrease the flux of Hg into sediments. Further studies were carried out in Finland by Lodenius (1983), who found increased Hg in fish, especially pike, in lakes receiving peatland drainage, especially in man-made lakes. He felt this increase was related to several environmental factors including low pH, high humic matter content, low temperature and low

primary production. Lodenius et al. (1983) further studied this phenomenon using peat lysimeters. The transport of Hg from peat soils to water was facilitated by humus. They also found that increased acidity promoted Hg binding to peat and bioaccumulation and speculated that acid rain would reduce leachability of Hg. This would agree with the findings of Jackson et al. (1980). Thus, more research is needed to explain some of the contradictory aspects of the behaviour of Hg in peatland drainage.

PRODUCTIVITY OF PEAT-DRAINAGE INFLUENCED WATERS

An important question is the effect of highly organic waters upon the primary productivity of aquatic ecosystems. Limnologists have used the term "dystrophic" to refer to waters that are high in DOM. Such waters have been considered to be low in phytoplankton productivity, thus poor in algae, and also receive high loadings of allochthonous organic matter (Wetzel, 1983). In contrast to earlier times, dystrophic lakes are no longer considered to be a special type, but to be part of the oligotrophic-eutrophic continuum (Berg and Petersen, 1956; Wetzel, 1983). Yet, highly humic waters are usually quite oligotrophic and several explanations have been advanced in order to explain this fact.

Few studies have been made on the actual productivity of such waters. Tolonen and Hosiaisloma (1978) used bioassays to compare the growth of algae in both minerotrophic and oligotrophic

waters from Finnish peatlands. Algal growth was ten times higher in minerotrophic waters than in bog waters. In terms of chemical composition, minerotrophic waters were much higher in major cations, anions and dissolved organic matter. No significant differences were found in major nutrients including N and P, SO_4^{-2} and trace metals. This data would suggest that the productivity of bog waters is either limited by major cations, inhibited by a factor related to their lower pH, or limited by another factor.

The observation that primary production is low in all highly humic lakes was backed up by a study done on Finnish polyhumic lakes by Arvola (1983). A possible reason for this is that the dark-brown colour limits light penetration and therefore photosynthesis, a possibility also advanced by Ilmavirta (1980).

Another factor that could reduce productivity in highly humic waters is nutrient limitation, especially for P. Jackson and Schindler (1975) found that humic complexes of Al and Fe could bind phosphate ions thus lowering the bioavailability of P. An important study in this area is that of Jackson and Hecky (1980). They noted that the primary productivity of lakes and reservoirs in northern Manitoba was inversely proportional to DOC and non-dialyzable Fe, organic carbon, nitrogen and H-bonded polymeric hydroxyl groups. This would suggest allochthonous humic-FeOOH complexes with strong, covalent metal-ligand bonds. They concluded that primary production was not depressed by reducing the bioavailability of P, light or

lowered pH. Apparently, colloidal humic-Fe complexes are formed which sequester Fe and essential micronutrients such as metals. They also postulated other DOM effects such as the accumulation of toxic phenolic compounds in such waters. Also, some binding of P could take place. A similar depression in the primary productivity of river water was observed in southeastern Alaska and attributed to humic substances (Sugai and Burnell, 1983). They felt that organic-metal complexes increased the transport of metals in fjords where such metals would normally precipitate.

Humic substances have been shown to have stimulatory effects upon algal growth. Prakash et al. (1972) stated that humus, especially low molecular weight compounds, can act as chelators and enhance algal growth. Other work reviewed by Sinclair (1981) has shown that substances found in peat can stimulate plant growth, both on a direct physiological basis and on an indirect basis. Such effects are dependent upon the concentration of the compound in question. For example, at high concentrations, algal inhibition can take place due to binding of nutrients by DOM and reduced bioavailability (Sakamoto, 1971; Prakash et al., 1972). Glooschenko and Moore (1973) stated that chelators could either stimulate or inhibit algal growth depending upon such factors as the nature of the chelator, its concentration, the particular metal and its concentration, antagonistic metal interactions, water chemistry, microbial influences, temperature, pH and light regime.

Another possible role of DOM would be as a source of nutrition. Particulates could be utilized by invertebrates or fish as a source of food depending upon the chemical composition of such particulates and their calorific value. DOM could be a potential source of heterotrophic nutrition for algae (Allen, 1969; Wetzel, 1968). For example, the algal flora of peatlands tends to be unique and certain algal groups such as cryptophytes can be common (Ilmavirta, 1980). This group is known to utilize DOM as a heterotrophic food source. It is beyond the scope of this paper to go into further detail on this subject.

IMPACT OF PEAT COMBUSTION ON WATER QUALITY

So far, this paper has been concerned with the potential impact of drainage waters from peatlands. Another potential input of contaminants to water bodies is via atmospheric deposition. Peat combustion would be a potential source of metals to the atmosphere and will be discussed briefly in this review.

Peat is a precursor to coal and is quite similar in concentration of potentially toxic metals. Its use as a fuel source for combustion should present similar environmental concerns (Glooschenko and Capobianco, 1982). Jantunen et al. (1984) discussed the chemical properties of peat fly ash. This data is presented for selected metals along with metal concentrations in coal fly ash

TABLE 2. Concentrations of selected metals in fly ash and coal. All concentrations in mg/kg.

Metal	Peat Fly Ash ¹	Coal Fly Ash ²	W. Canada Coal ³	Crust ⁴
As	2-32	440	0.5-800	1.8
Cd	1.5-3	31	<0.1-0.3	0.16
Cu	53-3900	140	60-180	68
Hg	0.3-1	0.05	approx. 1	0.08
Ni	100-2600	650	5-450	99
Pb	53-4000	650	<2-150	13
U	to 1500	30	<2-20	2
V	100-2000	1180	10-1250	136
Zn	28-4200	5900	3-400	76

¹Peat fly ash data from Jantunen et al., 1984.

²Coal fly ash data from Klein et al., 1975.

³Western Canadian coal data (as ash) from Kronberg et al., 1981.

⁴Earth crustal data from Fairbridge, 1972.

(Klein et al., 1975), western Canadian coals (Kronberg et al., 1981), and the earth's crust (Fairbridge, 1972). Several things are apparent. Peat fly ash is similar to coal fly ash in most metals except peat is somewhat lower in As and Cd, but higher in Hg and U. Also, these metals are all enriched in peat fly ash compared to their crustal abundance. Thus, concern with fly ash emission and disposal is important to prevent environmental contamination.

Another potential concern is that of SO₂ emission from peat combustion. Most of the peats considered for combustion are derived from freshwater plants and are fairly low in sulphur. Peats average 0.1 to 0.2% sulphur which is approximately one-tenth that found in coals (Monenco Ontario Ltd., 1981). Thus S emissions from peat would contribute less to acid rain than with the combustion of coal.

In conclusion, atmospheric emissions from peat combustion would be quite comparable to those from coal. Similar environmental controls would therefore be necessary for the two fuel sources.

CONCLUSIONS

We have reviewed the pertinent literature on the impacts of peatland drainage upon aquatic ecosystems. Our emphasis has been on the chemistry of drainage waters, both natural and artificial. Our discussion of biological aspects has been limited to primary production as research upon the impact of peatland drainage to aquatic

invertebrates and fish is lacking. However, the toxicological aspects of such waters are extremely important and merit further research.

Some concerns on peatland drainage appear to be unfounded. Acidification of receiving waters is limited as the pH of drainage water is higher than that found in the source bog. Any increase of suspended solids is limited in time. Nutrient concentrations found in drainage waters are similar to those in upland forests and should not create any eutrophication problems.

Some problems that might exist in receiving waters merit further research. These include the possibility of increased mercury loading of receiving waters, aluminum toxicity, decreased metal and phosphorus bioavailability, changes in the degradability and metal-binding capacity of DOM, and the possible presence of toxic organic compounds in drainage waters. These latter concerns are not strictly limited to the possible use of peat as an energy source, but are also applicable to forest drainage, horticultural peat extraction, development of northern reservoirs and to naturally-acidic ecosystems.

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