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**Proceedings of the
Fifth Annual Applied
Aquatic Studies Workshop,
Edmonton, Alberta,
February 21-22, 1989**

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Proceedings of the Fifth Annual Applied Aquatic Studies Workshop,
Edmonton, Alberta, February 21-22, 1989

by

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INTRODUCTION

The Annual Applied Aquatic Studies Workshop is a forum for the exchange of information and views on environmental issues related to fisheries and aquatic sciences. The idea of the workshop stemmed from initial discussions between the University of Alberta and the Alberta Environmental Centre, but many other organizations have actively participated in the workshop over its five year life. These include Alberta Environment, the Boreal Institute for Northern Studies, Canadian Society of Environmental Biologists, Alberta Recreation and Parks, Alberta Energy, and Alberta Forestry, Lands and Wildlife. The majority of papers presented at the workshop deal with specific issues in Alberta and western Canada, with smaller numbers of papers centering around generic topics or issues from the rest of Canada and the USA.

The 1989 Workshop had five sessions: Limnology of Brownwater Systems, Lake Rehabilitation, Drinking Water and Urban Water Management, Toxicology and Treatment of Pulp Mill Effluents, and Contributed Papers. These proceedings are organized according to the five sessions.

The proceedings of the 1988 workshop have not been published. We are, however, fortunate to include two papers from that workshop in the current proceedings. These papers are found under the Contributed Papers session.

**LIMNOLOGY OF
BROWNWATER SYSTEMS**

Geochemistry of Dissolved Organic Matter in Brownwaters from Atlantic Canada

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INTRODUCTION

For some time the acidity of natural waters has been the subject of much controversy, especially the delineation of the role that natural acidity from dissolved organic matter (DOM) plays in terms of the total acidity. It is possible that both organic and inorganic acidity are important in some areas (Patrick *et al.*, 1981), and this has been recently shown for Nova Scotia waters (Gorham *et al.*, 1986; Kerekes *et al.*, 1986) and in Finland (Kortelainen *et al.*, 1986; Kortelainen and Mannio, 1987). The question of the extent of acidity from these sources remains open for most areas.

To address this question properly, it is essential to characterize DOM in aquatic systems particularly in natural waters containing high concentrations of DOM; a situation common in eastern Canada. Hence this work was undertaken to determine the general character of DOM found in natural waters collected from three study areas in eastern Canada.

The DOM in the aquatic systems studied (bogs, streams, lakes, rivers, etc.) contains varying degrees of components from two sources; organic material "extracted" from the surrounding soils by natural leaching processes, and material of aquatic origin. This DOM pool is treated as a starting point for characterization in the same way as the base extract commonly used to define soil humic matter. Subsequent fractionation and characterization of the DOM utilizes techniques developed by both soil and aquatic chemists.

METHODS

Study Areas and Sampling Sites

Filtered water samples were obtained from three study areas in eastern Canada during the period of July, 1984 - March, 1986. At Sept-Iles (PQ) samples were taken from six sites representing different degrees of development for horticultural peat. In Kejimikujik (NB) and nearby areas, samples representing a broad range of DOM content were collected from lakes,

ivers, headwater streams, river mouths, and a bog. Samples were taken along the course of Judas Creek, which drains Barrington Bog (Shelburne County, NS), pore waters from wells installed in the bog, and from the Clyde River, into which Judas Creek flows.

Fractionation of DOM

DOM is by definition composed of particles less than 0.45 μm in size and can be separated into humic and fulvic acid fractions as usually defined in soil science (Stevenson, 1982). Here the organic aggregate which is insoluble at $\text{pH} = 2$ is called the humic acid (HA) fraction. DOM which remains soluble at $\text{pH} = 2$ contains traditional fulvic acid and low molecular weight components and is treated as a "fulvic fraction". These two fractions contain all the DOM. The method of Leenheer and co-workers (Leenheer, 1981, 1985; Leenheer *et al.*, 1982; Leenheer and Noyes, 1983), with modifications (Richmond and Bourbonniere, 1987), was used to further separate the fulvic fraction based on physico-chemical characteristics. Thus the entire DOM pool is fractionated into seven operationally-defined fractions; HA, fulvic hydrophobic- acid (HPOA), -base (HPOB), and -neutral (HPON), and fulvic hydrophilic-acid (HPIA), -base (HPIB), and -neutral (HPIN).

RESULTS AND DISCUSSION

DOM fractionations were completed on 160 of the samples collected for this study. Statistical data (in mg C L^{-1}) for this set are presented in Table 1 as well as the calculated average distribution of the seven organic fractions normalized to DOC. The composition of DOM from the study area is quite variable and on average the seven fractions isolated are ranked by concentration: $\text{HPOA} > \text{HPON} > \text{HA} > \text{HPIN} > \text{HPIA} > \text{HPIB} > \text{HPOB}$.

Relative Importance of Seasonal vs. Sample Type Variability

All of the samples used in this study can be classified into nine sample types based on their sampling environment. These are bog pore waters - upper layers (BUP) and - lower layers (BLP), bog natural drainage (BND), bog amended drainage (BAD), bog drainage with saline influence (BSL), headwater streams (HDW), river mouths (RVM), Pebblelogitch Lake (PEB), and Beaverskin Lake (BVS). These samples can also be sorted by seasons: SPRING (March

- May), SUMMER (June - August), AUTUMN (September - November), and WINTER (December - February).

Table 1. Statistical data for all samples with complete DOC fractionation (n = 160).^a

	DOC	HA	HPOA	HPIA	HPOB	HPIB	HPON	HPIN
Minimum	4.99	0.0	0.26	0.0	0.0	0.0	0.0	0.0
Maximum	71.9	25.6	31.2	14.4	2.02	11.4	18.2	15.8
Mean	25.3	4.48	7.90	2.72	0.15	0.86	5.06	45.12
SD	13.7	4.86	5.78	3.11	0.28	1.56	3.46	3.26
RSD	54.2%	109%	73.2%	115%	194%	181%	68.4%	79.2%
Frac-C (% of DOC)	17.7	31.2	10.8	0.59	3.40	20.0	16.3	
Cum (% of DOC)	17.7	48.9	59.7	60.3	63.7	83.7	100	

^aAll concentrations given in mg C L⁻¹

SD = $\pm$ Sample standard deviation (n-1 degrees of freedom)

RSD = (SD / Mean) x 100%

Frac-C = Mean fraction (mg C L⁻¹) / Mean DOC (mg C L⁻¹) x 100%

Cum-C = Cumulation of Frac-C

To compare the relative influence of the two factors, analyses of variance were conducted according to the unbalanced one-way classification model of Graybill (1961). Results for six parameters grouped by types and seasons are shown in Table 2 square values may be compared across columns for each parameter because they represent proportions of the same total variance. Thus the ratio of mean squares indicates the relative importance of these factors for each parameter. For DOC, HPOA, HPIA, and HPIN, the ratio is slightly higher than unity, suggesting that the sample type is a slightly more (for HPOA about 5 times more) important contributor to the total variance than is season. For HPON, sample type is much more important than season, and for HA, season is more important than sample type.

The values for the F statistic presented in Table 2 show, with a high degree of certainty, that the classification of samples by type explains most of the total variance for DOC, HPOA, HPIN, and HPIA (certainty decreasing in the order given). For HA it is less certain, and for HPON much less certain that the total variance can be explained by such classification. Similarly the variance for DOC, HPOA, HPIN, and HPIA may be explained to a large degree

by seasonal classification, with a high certainty. It is less certain that HA variance can be explained by seasonal classification, and that variance for HPON cannot be explained at all by such sorting.

Table 2. Analysis of variance for one-way unbalanced classification for six parameters grouped by sample type or by season.^a

PARAMETER	Sorting by Sample Type			Sorting by Seasons			
	Mean Square Types	Mean Square Replic.	F	Mean Square Seasons	Mean Square Replic.	F	Ratio ^b
DOC	1823	101	18.0 ^c	1669	159	10.5 ^c	1.09
HA	55.1	22.1	2.4 ^d	83.6	22.5	3.7 ^d	0.66
HPOA	285	20.1	14.2 ^c	242	29.4	8.2 ^c	1.18
HPIA	52.9	7.4	7.1 ^c	48.4	9.0	5.4 ^c	1.09
HPON	21.2	11.5	1.8 ^e	4.1	12.2	-	5.17
HPIN	73.1	7.3	10.0 ^c	70.7	9.5	7.4 ^c	1.03

^aN = 160; Degrees Freedom: Types (8); Replic. (151); Seasons (3); Replic. (156)

^bRatio = Mean Square Types / Mean Square Seasons

Conf. Levels: ^c99%; ^d97%; ^e<95%

Acidic DOM Fractions

The total of fractions defined as acidic (HA + HPOA + HPIA) comprises on average 59.7% of the DOC. Note that acidic fractions are not necessarily the sole repositories of the acidic functional groups in DOM, but differences in the physico-chemical properties used to isolate them suggest that acidic fractions may differ from each other in the classes of compounds which they contain (Leenheer *et al.*, 1982). In Table 3 the acidic fractions are sorted by type and season. The mean proportions of the DOC in each fraction, and the sum of acidic fractions are given in Table 3.

For sample types which can be considered "integrated" (RVM, PEB, BVS) and for deeper pore waters (BLP) the HA content is greater than for the remaining sample types. HA is lower in the spring compared to other seasons. HPOA is much less variable by sample type or season except for BVS, a clearwater lake which receives little contribution from the thin soils in its watershed (Kerekes, 1984). A higher proportion of HPIA occurs in bog related sites but

somewhat lower amounts are in the integrated sample types, and also lower proportions occur in spring and winter.

Table 3. Acidic fractions of DOM sorted by type and season.^a

Type/Season	n	HAH	POA	HPIA	SUM
BUP	12	10.5	34.0	15.4	59.9
BLP	4	23.6	24.8	15.4	63.8
BND	36	16.5	35.9	11.5	63.9
BAD	11	13.5	29.1	9.76	52.4
BSL	7	18.7	33.0	10.2	61.9
HWS	29	16.6	29.9	9.38	55.9
RVM	32	22.5	28.8	9.26	60.6
PEB	15	22.4	28.1	7.11	57.6
BVS	14	24.5	7.66	3.48	35.6
SPRING	27	12.1	29.6	7.83	49.5
SUMMER	59	17.8	32.3	12.1	62.2
AUTUMN	36	19.2	32.6	10.5	62.3
WINTER	38	19.2	28.0	9.60	56.8
OVERALL	160	17.7	31.2	10.8	59.7

^aValues are mean proportions of the DOC in each fraction.

Acidic fractions can occur in all types of natural waters, whether they are clear or coloured. In this study two lakes (PEB and BVS) are different in DOC, pH, and DOM character (Fig. 1). The coloured lake (PEB) is only about 1 km away from the clear lake (BVS); both receive the same atmospheric input, and both have much the same inorganic content (Kerekes *et al.*, 1982).

CLEAR vs. COLOURED LAKE

Figure 1. Representations of the general character of a clear and coloured water lake. Pies are scaled to mean DOC (mg C L⁻¹).

Titrateable Acidity of Fractions

A selection of fractions defined as acidic were titrated with a base to estimate the total COOH content. Fractions were concentrated (HA was first dissolved off glass fibre filters), and the titration curve was determined. There was considerable blank from resin bleed and interference from high salt contents. A series of standards and blanks were also titrated and appropriate corrections were applied. Total COOH for the HA and HPOA fractions is estimated by titrating in the pH range of 2.75 - 7, and for the HPIA fraction the appropriate range is pH 3 - 7. The fractions from 18 samples have been titrated, and the results are summarized in Table 4.

Table 4. Estimates of COOH density for acidic DOM fractions.^a

Site	(Type, n)	HAH	POA	HPIA
Beaverskin Lake	(BVS, 4)	16	12	2
Pebbleloggitch Lake	(PEB, 5)	13	8	11
Mersey River, Site 1	(HWS, 3)	16	9	14
Mersey River, Site 3	(RVM, 4)	14	9	17
Pebbleloggitch Bog	(BLP, 1)	11	15	14
Clyde River	(RVM, 1)	12	6	1
All Samples (n = 18)		14	10	10
SD ^b		3	3	6
RSD (%) ^b		21	30	60

^aValues are in $\mu\text{eq COOH mg}^{-1} \text{ C}$

^bSee Table 1 for definitions of statistical measures

There are some differences in the carboxyl density with sample type and the fraction. The greatest variability occurs in the HPIA fraction. The average values for the HPOA fraction compare favourably with those for fulvic acid, 10 - 13 $\mu\text{eq mg}^{-1} \text{ C}$ (Oliver *et al.*, 1983) and for HPOA in Thoreau's Bog, 10 $\mu\text{eq mg}^{-1} \text{ C}$ (McKnight *et al.*, 1985). The HA values in this study are higher than those of Oliver *et al.* (1983) (5 - 10 $\mu\text{eq mg}^{-1} \text{ C}$), and also the HPIA value is lower than the value of 13 $\mu\text{eq mg}^{-1} \text{ C}$ determined by McKnight *et al.* (1985). The precipitation of HA in this study precedes fractionation; possibly HPIA components are co-precipitated with HA. Some samples of HPIN also exhibited acidic character.

Effect of Amended Bog Drainage

Ditching of a bog to increase drainage requires about two metres of excavation. The impact of amended drainage is best illustrated by the Sept-Iles data shown in Table 5. On average about 30% more DOC is released by amended drainage than in undisturbed systems. The absolute average concentrations of all DOM fractions also increases for amended drainage but the relative proportions show mixed results. HA and HPIA show slightly increased proportions, not enough to change the acidic nature of the DOM, and are probably compensated by the decrease in HPOA relative proportion. Since the acidic fractions contain the majority of

the functional groups which can complex metals, then the relative capacity for metal complexation is likely not changed by the amended drainage. However, the absolute capacity of drainage water to complex metals and to provide organic acidity is concentration dependent, and will increase by roughly 30%. The impact of this will depend upon the properties of the receiving waters. Increased complexation may sequester metals which are micronutrients, but may have a beneficial impact upon environments where toxic metals are a problem. Further information on this work can be found in Bourbonniere (1987).

Table 5. Comparison of amended and undisturbed drainage water from Sept-Iles horticultural peat bogs.^a

		DOC	HA	HPOA	HPIA	HPON	HPIN
Sept-Iles (n=6)	Mean	21.98	2.71	7.69	1.65	5.34	3.21
Undisturbed	Frac-C (%)		12.34	35.00	7.50	24.31	14.63
Sept-Iles (n=11)	Mean	29.04	3.92	8.43	2.83	6.49	6.62
Amended	Frac-C (%)		13.49	29.02	9.74	22.36	22.80

^aMean values are in mg C L⁻¹ and Frac-C is a % of the sample DOC.

Impact of DOM Character on Metal-Organic Interactions and Aluminum Toxicity

Investigations have demonstrated that HA can precipitate at pH significantly higher than the value at which it is traditionally defined (pH 2). Of the DOC which can be precipitated at pH 2 for a high DOC streamwater sample (MPIT), half of it is precipitable at pH 4.6 (Table 6). This value is within the normal springtime range for this site according to long term monitoring data. For the sample used in this study the 50% HAT level corresponds to a loss of 10% (2 mg L⁻¹) of the DOC simply by acidification. Similarly the TMIP sample lost about 15% (1.2 mg L⁻¹) of its DOC to HA precipitation at pH 4.2, PBCR lost about 10% (0.8 mg L⁻¹) at pH 4.0, and PBBG lost about 3% (0.8 mg L⁻¹) at pH 3.4.

Table 6. pH values at which 50% of the traditionally-defined HA (% HAT) had precipitated according to three determination methods.

Sample	Carbon	Abs 330	Density
PBCR ^a	4.0	3.6	3.0
PBBG ^b	3.4	2.8	3.1
TMIP ^c	4.2	3.8	3.8
MPIT ^d	4.6	2.2	2.3

^aPebbleloggitch Creek

^bPebbleloggitch Bog lower pore water

^cThomas Meadow Bog pore water

^dMoose Pit Brook

The significance of the DOM lost to early HA precipitation will vary with the characteristics of each site. If the loss coincides with the onset of conditions which put stress on sensitive biota, it could be more important. For instance during spring melt, pH often decreases to annual low values, DOM is diluted by snowmelt, and inorganic monomeric aluminum is increased initially (Hendershot *et al.*, 1986). This combination of conditions places newly hatched fish at risk of aluminum toxicity. Further reductions of DOM by precipitation of HA, which would otherwise complex aluminum rendering it non-toxic, may place the fish at increased risk.

In a general way, water of differing character can protect alevins in an environment that would otherwise be toxic (see Table 7). Bioassay tests which were made using water from Moose Pit Brook amended with Al (Treatments #2, 3, and 5) afforded much greater protection than Treatment #7 made with Medway River water, amended with Al, and comparable in DOC. Treatment #4, also made with Moose Pit water and amended, was equally toxic to #7, but it contained less DOC. So Moose Pit water exhibits a limit to its protecting ability.

Table 7. Summary of bioassay results for various concentrations of inorganic monomeric Al and dilutions of natural waters (Medway (Med.) and Moose Pit (M.P.)).

Treatment	DOC ^a	IMAL ^b	LT ₅₀ ^c	% Mort ^d
1. Medway	6.8	1.3	>30	0
2. M.P. dil Med.	DOC 6.8	9.3	>30	10
3. M.P. dil Med.	HA 7.1	9.4	>30	5
4. M.P. dil Med.	HPOA 4.4	11.2	10	100
5. M.P. dil Med.	HPIA 7.4	10.4	>30	5
6. Moose Pit	24.5	3.4	>30	5
7. Med. + M.P.	A 16.8	10.2	19	100
8. Simulated Med.	0	6.0	>30	30
9. Simulated M.P.	0	16.1	10	100
10. Simulated M.P. no Al	0	0.2	26	70

^aValues in mg C L⁻¹

^bInorganic monomeric Al in µeq L⁻¹

^cTime to 50% mortality in days

^dPercent mortality for the 30-d experiment

Although Moose Pit and Medway waters differ in all DOM fractions, the most striking differences are the higher proportion of HPOA and lower proportion of HPON in the Moose Pit sample. The acidic fractions contain carboxyl and hydroxyl functions (Leenheer, 1981) which can form strong complexes with Al in natural waters (Kramer, 1986). The dilutions of treatments #2, 3, and 5 were made relative to DOC and the other acidic fractions and resulted in a greater amount of Moose Pit sourced HPOA in the treatments. In contrast, treatment #4, diluted relative to HPOA, resulted in equal Moose Pit sourced HPOA and lesser amounts of DOC and other acidic fractions. The latter combination proved more toxic (Table 7).

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REFERENCES

- Bourbonniere, R.A. 1987. Organic geochemistry of bog drainage waters. In: Proc. Symposium '87 - Wetlands/Peatlands (Rubec, C.D.A. and Overend, R.P., Eds.). Edmonton, AB. pp. 139-145.
- Gorham, E., J.K. Underwood, F.B. Martin, and J.G. Ogden III. 1986. Natural and anthropogenic causes of lake acidification in Nova Scotia. *Nature* 324:451-453.
- Graybill, F.A. 1961. An Introduction to Linear Statistical Models, Vol. 1. McGraw-Hill, New York, NY. 463 pp.
- Hendershot, W.H., A. Dufresne, H. Lalande, and F. Courchesne. 1986. Temporal variation in aluminum speciation and concentration during snowmelt. *Wat. Air Soil Poll.* 31:231.
- Kerekes, J.J. 1984. Review of the regional aquatic program. In: Report of the Fall 1984 Workshop, Atlantic Region LRTAP Monitoring and Effects Working Group (Taylor, B.L., Ed.), pp. 83-85. Environment Canada.
- Kerekes, J., G. Howell, S. Beauchamp, and T. Pollock. 1982. Characterization of three lake basins sensitive to acid precipitation in central Nova Scotia (June 1979 to May 1980). *Int. Rev. Ges. Hydrobiol.* 67:679-694.
- Kerekes, J., S. Beauchamp, R. Tordon, C. Tremblay, and T. Pollock. 1986. Organic versus anthropogenic acidity in tributaries of the Kejimikujik watersheds in western Nova Scotia. *Wat. Air Soil Poll.* 31:165-173.
- Kortelainen, P., and J. Mannio. 1987. The contribution of acidic organic anions to the ion balance of lake waters. In: Acidification and Water Pathways, Vol. II. pp. 229-238. Norwegian National Committee for Hydrology, Oslo.
- Kortelainen, P., J. Mannio, and I. Makinen. 1986. Strong and weak acids in lake waters - a methodological study. *Aqua Fennica* 16:221-229.
- Kramer, J.R. 1986. Aluminum geochemistry. In: Aluminum in the Canadian Environment (Havas, M. and J.F. Jaworski, Eds.). pp. 25-49. National Research Council of Canada, Ottawa. NRCC No. 24759.
- Leenheer, J.A. 1981. Comprehensive approach to preparative isolation and fractionation of dissolved organic carbon from natural waters and wastewaters. *Environ. Sci. Technol.* 15:578-587.
- Leenheer, J.A. 1985. Fractionation techniques for aquatic humic substances. In: Humic Substances in Soil, Sediment, and Water. (Aiken, G.R., D.M. McKnight, R.L. Wershaw, and P. MacCarthy (Eds.)). pp. 409-429. John Wiley & Sons, New York.

- Leenheer, J.A., and T.I. Noyes. 1983. A filtration and column-adsorption system for onsite concentration and fractionation of organic substances from large volumes of water. USGS Water Supply, Washington, DC. Paper No. 2230. 40 pp.
- Leenheer, J.A., T.I. Noyes, and H.A. Stuber. 1982. Determination of polar organic solutes in oil-shale retort water. *Environ. Sci. Technol.* 16:714-723.
- McKnight, D., E.M. Thurman, R.L. Wershaw, and H. Hemond. 1985. Biogeochemistry of aquatic humic substances in Thoreau's Bog, Concord, Massachusetts. *Ecology* 66:1339-1352.
- Oliver, B.G., E.M. Thurman, and R.L. Malcolm. 1983. The contribution of humic substances to the acidity of colored natural waters. *Geochim. Cosmochim. Acta* 47:2031-2035.
- Patrick, R., V.P. Binetti, and S.G. Halterman. 1981. Acid lakes from natural and anthropogenic causes. *Science* 211:446-448.
- Richmond, M., and R.A. Bourbonniere. 1987. Manual for the fractionation of dissolved organic matter in natural waters. Environment Canada. NWRI Report 87-145. 29 pp.
- Stevenson, F.J. 1982. *Humus Chemistry*. John Wiley & Sons, New York, NY. 443 pp.

Interaction of Humic Substances with Aquatic Pollutants

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INTRODUCTION

The majority of dissolved organic carbon (DOC) (by operational definition smaller than 0.45 μm in size) is present in natural waters as polymerized organic acids called humic substances. Humic substances are breakdown products of plant-derived lignin materials. Their elemental composition is 50% carbon, 4 - 5% H, 35 - 40% oxygen, 1 - 2% N and less than 1% sulfur and phosphorus. The major functional groups include carboxyl, phenolic hydroxyl, carbonyl and hydroxyl groups. Humic substances are polyelectrolyte in nature, with varying molecular weights (up to several hundred thousand) and with different functional groups. They comprise 50 - 75% of DOC and thus are a major class of organic compounds in natural waters. Humic and fulvic acids are the two environmentally significant fractions of the humic substances.

Dissolved humic and fulvic acids have a polyanionic backbone; at pH 6 - 8 they provide many metal binding sites and buffering capacity in natural waters. Aquatic humic substances are a class of organic acids which are yellow in colour, polyelectrolytic, and non-volatile in nature. Humic acids are acid insoluble. Fulvic acid is soluble in both acid and alkali and functionally active in the pH range 2 - 11. The dissolved fulvic acid is a low molecular compound of 2 nanometres in diameter. Thus most of the DOC behaves as individual dissolved ions of humic and fulvic acids.

In natural waters, large aggregates of humic acids are in the colloidal form, and are commonly found associated with clay minerals or oxides of iron and aluminum. The molecular weight of this colloidal humic acid fraction ranges from 2000 to 100,000; this fraction represents 10% of the total DOC. Also the colloidal humic acid has fewer carboxylic and hydroxyl groups than the fulvic acid fraction.

ISOLATION

Anion exchange resins were used to isolate humic substances from water; however, poor recovery was achieved with resins due to the strong binding of humic substances onto the resins. With the advent of non-ionic XAD resins in the late 1960s quantitative isolation of humic substances was made possible on a routine basis (Thurman *et al.*, 1978). The absorption of humic substances on XAD resin is through weak physical forces. The sample is first acidified to pH 2.0 with concentrated hydrochloric acid and pumped onto the XAD resin where the organic acids are absorbed in the protonated form. The sample is then eluted with dilute sodium hydroxide to desorb the humic substances. This change in ionic character is sufficient to adsorb and desorb the sample efficiently from the resin.

Generally, the XAD resins isolate about 50% of the DOC from an average surface water sample. For bog and marsh waters, the recovery of DOC increases by 75 to 90%. The substances that are separated are yellow organic acids, which account for 85% of the organic colour of the water. For a review of isolation methods of humic substances refer to Thurman (1979, 1985) and Thurman and Malcolm (1981, 1983).

HUMIC PROPERTIES OF ENVIRONMENTAL SIGNIFICANCE

Binding of Humic Substances to Inorganic Surfaces

Organic matter in natural waters adsorbs to alumina surfaces which provides a net negative charge over the entire pH range. Thus suspended particulate matter and bottom sediments contain a coating of natural fulvic and humic acids. The extent of surface coverage by adsorbed organic matter depends on pH of the water and the iso-electric point of the mineral surface. Although the concentration of DOC may be small, the coverage of mineral surfaces by organic matter is large. Tipping (1981) estimated that 30 to 40% of the surface of iron hydroxide will be covered by organic matter present at ambient concentrations in most surface waters.

Surfactant Properties of Humic Substances

Due to their hydrophobic nature, humic substances exhibit surfactant properties, especially during the spring high water period. As a result of this property humic substances appear as

foam or micelles on water surfaces. Humic substances are also known to dissolve oils and hydrocarbons and they have been used to disperse small oil pools in natural waters.

Interaction with Metal Ions

Metal ions present in natural waters can form complexes with dissolved and suspended organic matter and with organic matter in bottom sediments. Humic and fulvic acids are the most important complexing agents of the dissolved organic fraction. The metal-binding constants of this fraction range between 10^3 to 10^{11} for metals such Cd^{++} , Pb^{++} and Hg^{++} in river water. Due to the variation in the type and number of functional groups among the different sources of humic and fulvic acids used, there could be variation in the binding constants reported for the same metal ion. In addition to simple complexes or chelates, mixed ligand complexes have been reported for several metal ions. The properties of these mixed complexes differ from those of the simple parent complexes (Manning and Ramamoorthy, 1973; Ramamoorthy and Manning, 1974).

Metal ions exist in natural waters as mixed ligand complexes. This is due to the presence of a variety of anions which originate from both natural and anthropogenic sources. The increased uptake of metal and phosphate nutrients by plants on the addition of nitrilotriacetic acid (NTA) to soil was suggested to be due to the formation of phosphate-M-NTA (where M = metal ion) complexes (Abdulla and Smith, 1963). Schnitzer (1969) has also proposed that metal-fulvate complexes in soil react with phosphate to form phosphate M-fulvate complexes. Many metal phosphates are insoluble and consequently immobile in neutral and alkaline soils, whereas carboxylate M-phosphate complexes are often soluble over a wide pH range (Flaig, 1971).

Binding of Organic Compounds

Several studies have shown that humic substances bind to pesticides and other organic compounds (Carter and Suffet, 1982; Gjessing and Berglind, 1982; Perdue, 1983; Thurman, 1985). These studies showed that humic substances at concentrations found in natural waters increased the solubilities of insoluble or sparingly soluble organic compounds by 2 to 3 times. DDT was shown to bind effectively with aquatic humic substances (Carter and Suffet, 1982) and its binding strength was found to vary with pH, ionic strength and other competing inorganic ions present. The reported range of $\log K$ (K = binding constant) of DDT binding to humic

substances varied from 4.83 to 5.74. The binding constant was higher for the more hydrophobic commercial humic acid than for the hydrophilic DOC from a natural reservoir.

Trihalomethane (THM) Production

Humic substances in water are an important source of organic matter implicated in the production of THMs in the chlorination of water and sewage. This was first identified by Rook (1977) and since then many studies have followed the role of humic substances in THM production. The four books on water chlorination and THM production by Jolley (1978) and Jolley *et al.* (1978, 1980, 1983) are recommended as major references for this topic.

In order to lower the formation of THMs during water treatment, studies were undertaken to look at the removal of humic substances through coagulation by alum. The optimum pH for the removal of humic substances by alum was reported to be between pH 5 and 6, and the quantity of alum required was correlated with the humic content of the water (Hall and Packham, 1965). A later study by O'Melia and Dempsey (1981) showed that Hall's results were applicable only to humic concentrations of $>25 \text{ mg L}^{-1}$. The probable cause of coagulation could be the charge neutralization of the adsorbed humic substances by aluminum polymers, resulting in the formation of flocculates if the overall charge of the Al-humic polymer is near zero.

O'Melia and Dempsey suggested two processes for the removal of humic substances at low ($<5 \text{ mg L}^{-1}$) concentrations of humic substances: (i) aluminum polymers can interact with humic molecules in the pH range of 5 to 6, resulting in under-sized colloids which may not settle but could be filtered out, and/or (ii) the precipitated aluminum hydroxide may absorb humic substances from dilute solution and settle them due to larger size.

Other adsorbents, such as ferric hydroxide, calcium carbonate, and manganese oxides, were also tested for removal of humic substances (O'Melia and Dempsey, 1981). The presence of calcium was observed to enhance the adsorption of humic substances possibly by complex formation (O'Melia and Dempsey, 1981). It is also possible that the organically coated mineral particles will stay in suspension due to electrical repulsion of their negative charges.

Photodegradation of Pesticides

Recent studies have shown that the photodegradation of some pesticides (belonging to groups such as carbamates, phosphates, ureas, triazole derivatives and pyrethroid) which are

non-degradable or nearly non-degradable in standard laboratory tests, was significant in the presence of humic substances. This indirect or sensitized photolysis involved the strong adsorption of sunlight in the ultraviolet-region by humic substances, followed by initiation of degradation processes by energy transfer to the non-absorbing pesticide molecules (Uta Jensen-Korte *et al.*, 1987).

Adsorption of Mutagens

A recent study by Sato *et al.* (1987) on the adsorption of mutagens by humic acids from river water concluded that:

"Humic acid inhibited the mutagenicity of various mutagens. The inhibitory effect was desmutagenic, heat-resistant and increased with an increase of the humic acid molecular weight. Typical monomeric components of humic acids had no desmutagenic effect. The desmutagenic effect of humic acid was caused by adsorption of mutagen, not by decomposition of mutagen. The adsorption activity was largest at its critical micelle concentration and the adsorbed mutagen was released by ultrasonication.

Humic acids exist in the natural environment in large amounts and may play an important role for natural purification by adsorption of mutagens."

Effects of Bioaccumulation of Pollutants

Humic and fulvic acids play a significant role in the bioavailability of aquatic pollutants. The determinants are the molecular weight and size of the humic and fulvic acids which bind to the pollutant. Metal complexes of low molecular weight and charge tend to bioaccumulate more than did complexes of high molecular weight and high charge. Transport across cell membranes may be slower for complexes of humic substances with high molecular weight, large size and high electrical charge. Studies on the effect of dissolved organic matter on the uptake and accumulation of heavy metals have been reported in the literature (Heavy Metals in the Environment, 1975-1988; Ramamoorthy and Blumhagen, 1984; Campbell and Evans, 1987).

Recent work by Kukkonen and Oikari (1987) showed significantly lower bioconcentration of hydroabietic acid and benzo(a)pyrene by *Daphnia magna* in a natural humic water when compared with a standardized soft freshwater. For pentachlorophenol, no such effect could be

observed. Therefore, depending on the chemical involved, the natural humic water might affect the acute toxicity of aquatic contaminants.

REFERENCES

- Abdullah, I., and M.S. Smith. 1963. Influence of chelating agents on the concentration of some nutrients for plants growing in soil under acid and alkaline conditions. *J. Sci. Fd. Agric.* 14:98-109.
- Campbell, J.H., and R.D. Evans. 1987. Inorganic and organic ligand binding of lead and cadmium and resultant implications for bioavailability. *Sci. Total Environ.* 62:219-227.
- Carter, C.W., and I.H. Suffet. 1982. Binding of DDT to dissolved humic materials. *Env. Sci. Technol.* 16:735-740.
- Flaig, W. 1971. Organic compounds in soil. *Soil Sci.* 111:19-33.
- Gjessing, E.T., and K.L. Berglind. 1982. Analytical availability of hexachlorobenzene (HCB) in water containing humus. *Vatten* 38:402-405.
- Hall, E.S., and R.F. Packham. 1965. Coagulation of organic colour with hydrolyzing coagulants. *J. Amer. Wat. Works Assoc.* 57:1149-1166.
- Heavy Metals in the Environment. 1975-1988. Proceedings of the International Conferences on Heavy Metals in the Environment. CEP Consultants Ltd., Edinburgh, U.K.
- Jolley, R.L. 1978. Water chlorination: environmental impact and health effects. Vol. 1. Ann Arbor Science, Ann Arbor, MI.
- Jolley, R.L., W.A. Brungs, J.A. Cotruvo, R.B. Cumming, J.S. Mattice, and V.A. Jacobs. 1983. Water chlorination: environmental impact and health effects. Vol. 4, Book 1. Chemistry and Water Treatment. Ann Arbor Science, Ann Arbor, MI.
- Jolley, R.L., W.A. Brungs, and R.B. Cumming. 1980. Water chlorination: environmental impact and health effects. Vol. 3. Ann Arbor Science, Ann Arbor, MI.
- Jolley, R.L., H. Gorchev, and D.H. Hamilton, Jr. 1978. Water chlorination: environmental impact and health effects. Vol. 2. Ann Arbor Science, Ann Arbor, MI.
- Kukkonen, J., and A. Oikari. 1987. Effects of aquatic humus on accumulation and acute toxicity of some organic micropollutants. *Sci. Total Environ.* 62:399-402.
- Manning, P.G., and S. Ramamoorthy. 1973. Equilibrium studies of metal-ion complexes of interest to natural waters - VII; Mixed-ligand complexes of Cu(II) involving fulvic acid as primary ligand. *J. Inorg. Nucl. Chem.* 35:1577-1581.

- O'Melia, C.R., and B.A. Dempsey. 1981. Coagulation of natural organic substances in water treatment. Progress Report at the Research Seminar in Environmental Engineering, U.S. Environmental Protection Agency. Cincinnati, OH. 24 pp.
- Perdue, E.M. 1983. Association of organic pollutants with humic substances: Partitioning equilibria and hydrolysis kinetics. In: Aquatic and Terrestrial Humic Materials. (Christman, R.F., and E.T. Gjessing, eds.). pp. 441-460. Ann Arbor Science, Ann Arbor, MI.
- Ramamoorthy, S., and K. Blumhagen. 1984. Uptake of Zn, Cd, and Hg by fish in the presence of competing compartments. Can. J. Fish. Aq. Sci. 41:750-756.
- Ramamoorthy, S., and P.G. Manning. 1974. Equilibrium studies of metal-ion complexes of interest to natural waters - VIII; Fulvate-phosphate, fulvate-NTA and NTA-phosphate complexes of Pb^{2+} , Cd^{2+} and Zn^{2+} J. Inorg. Nucl. Chem. 36:695-698.
- Rook, J.J. 1977. Chlorination reactions of fulvic acids in natural waters. Env. Sci. Technol. 11:478-482.
- Sato, T., Y. Ose, H. Nagase, and K. Hayase. 1987. Adsorption of mutagens by humic acid. Sci. Total Environ. 62:305-310.
- Schnitzer, M. 1969. Relationship between fulvic acid, a soil humic compound and inorganic soil constituents. Proc. Soil. Sci. Soc. Amer. 30:75-81.
- Thurman, E.M. 1985. Aquatic humic substances. In: Organic Geochemistry of Natural Waters. (Martinus Nijhoff/Dr. W. Junk, publ). Dordrecht, The Netherlands.
- Thurman, E.M., and R.L. Malcolm. 1983. Structural study of humic substances: New approaches and methods. In: Aquatic and Terrestrial Humic Materials. (Christman, R.F., and E.T. Gjessing, eds.). pp. 1-23. Ann Arbor Science, Ann Arbor, MI.
- Thurman, E.M., and R.L. Malcolm. 1981. Preparative isolation of aquatic humic substances: Environ. Sci. Tech. 15:463-466.
- Thurman, E.M. 1979. Isolation, characterization, and geochemical significance of humic substances from ground water. Ph.D. Thesis, University of Colorado, Boulder, CO.
- Thurman, E.M., R.L. Malcolm, and G.R. Aiken. 1978. Prediction of capacity factors for aqueous organic solutes adsorbed on a porous acrylic resin. Anal. Chem. 50:775-779.
- Tipping, E. 1981. Adsorption to goethite (α -FeOOH) of humic substances from different lakes. Chem. Geol. 33:81-89.
- Uta Jensen-Korte, C. Anderson, and M. Spiteller. 1987. Photodegradation of pesticides in the presence of humic substances. Sci. Total Environ. 62: 335-340.

Influences of Anthropogenic Activities on Waters Associated with Peatlands

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ABSTRACT

This presentation is derived from a report prepared for Alberta Environment in which the nature of peatlands in Alberta and the influences of anthropogenic activities, as reported in the world literature, on waters associated with peatlands are reviewed (Turchenek *et al.*, 1987). For further information readers may refer to the final version of the report which was completed during the latter part of 1989.

The area of peatlands in Alberta is estimated at 12.7 million ha, accounting for about 19% of the total area of the province (Tarnocai, 1984). Peatlands are most abundant in the boreal regions of the province, and in the far north they are characterized by permafrost conditions in the subsurface. There has been relatively little utilization of peatlands in Alberta, but it is expected that some types of development will soon increase. The exploitation of peatlands in other parts of the world has been shown to have various effects on the environment, particularly in terms of affecting the quality and quantity of water in lakes and streams draining the peatlands.

The human activities that can affect the nature of peatlands and their receiving waters fall into different categories. The first of these is direct exploitation involving drainage prior to development. These include developments such as agriculture, forestry, and horticultural peat moss extraction. In parts of Canada, extraction of peat for use as fuel also falls into this category. A second category of activities consists of those that do not actually exploit the peatland but nevertheless affect it through direct physical disturbances. Linear features such as roads, pipelines, and cutlines through peatlands are examples of such disturbances. This category also includes situations in which peatlands have been drained as part of water table regulation measures required for developments remote from the peatland itself. A third category of activities with potential impacts on peatland waters involves contamination by pollutants from near or distant sources. Deposition of acidic substances, heavy metals, and dust are included in this category.

Peatlands are generally characterized by "brown waters", or waters with high contents of pigmented organic substances. Organic materials in water are commonly expressed in terms of total organic carbon (TOC), which can be subdivided into dissolved organic carbon (DOC) and particulate organic carbon (POC) fractions. The DOC fraction comprises 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 autochthonous and allochthonous detritus of plant and animal origin (Cole, 1983). In different classes of wetlands, concentration ranges of DOC are as follows: 5 to 15 mg L⁻¹ in marshes; 10 to 30 mg L⁻¹ in swamps; 3 to 10 mg L⁻¹ in both poor and rich fens, and 30 to 400 mg L⁻¹ in ombrotrophic bogs (Thurman, 1985). Streams and lakes in drainage basins with a prevalence of bogs are therefore generally more strongly coloured than those in which fens, swamps or marshes are predominant.

Much attention has been given to the dissolved organic substances which impart a yellow to brown colour to water. The reasons for this are that: (1) the organic materials are acidic in nature and, therefore, influence the acid status and buffering capability of water, 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. Humic materials along with other dissolved and particulate organic materials have a direct influence on water quality parameters such as dissolved oxygen levels, chemical oxygen demand, biochemical oxygen demand, colour, turbidity and suspended solids.

Disturbances of wetland surfaces such as ditching for drainage and removal of surface vegetation can result in increased levels of DOC as well as of particulate organic materials in downstream waters. Studies comparing the quality of runoff waters from natural and developed peatlands have been carried out in Finland, the Soviet Union, the United States, and in Canada. Elevation of DOC and TOC levels has been demonstrated in some of these studies but not in others. Increases in chemical oxygen demand, biochemical oxygen demand, colour, turbidity, and dissolved solids, combined with decreases in dissolved oxygen levels, were demonstrated in some of the reports. The studies have indicated that noncompliance with water quality guidelines can occur. However, the levels of these parameters were often found to exceed the water quality

guidelines in undeveloped as well as in developed peatlands. Thus, thorough evaluation of baseline, or pre-development, conditions is required for valid evaluation of peatland development impacts on water quality.

The major peatland use in Alberta is farming. Total area of peatlands in agricultural Counties, Municipal Districts and Improvement Districts in the province has been estimated to be about 4.4 million ha (Wehrhahn and Marciak, 1986). In some counties, about 20% of the peatlands are suitable for agriculture. If this percentage is applied to all peatlands in agricultural regions, then it can be estimated that about 1 million ha of peatland can eventually be converted to agricultural use. It is evident that impacts on lakes and streams will be greatest in those regions with the highest concentrations of peatlands.

Ditching peatlands to control water tables and thereby improve forest growth is commonly practised in Scandinavian countries and in the former USSR. There is much interest in draining peatlands in Alberta to increase timber production. However, only pilot projects are being carried out to determine the feasibility of drainage and success of tree growth. Should this technique prove to be successful, up to 4 million ha could eventually be drained (Hillman, 1988).

Horticultural peat moss extraction is the only other exploitative activity in peatlands in Alberta. Although there are 4 or 5 operations in the province, a small proportion of the total peatland area is involved. The impacts on water quality of other non-exploitative activities in peatlands have not been investigated to a great extent. Effects of acid and heavy metal deposition on wetlands have perhaps received the greatest attention, and are reviewed in the document prepared for Alberta Environment (Turchenek *et al.*, 1987).

REFERENCES

- Cole, G.A. 1983. Textbook of Limnology. 3rd Ed. The C.V. Mosby Co., Toronto, ON. 401 pp.
- Hillman, G.R. 1988. Improving wetlands for forestry in Alberta. Canadian Forestry Service and Alberta Forest Service, Edmonton, AB. 22 pp.
- Stumm, W., and J.J. Morgan. 1981. Aquatic Chemistry - An Introduction Emphasizing Chemical Equilibria in Natural Waters. John Wiley and Sons, New York, NY. 780 pp.
- Tarnocai, C. 1984. Peat resources of Canada. National Research Council of Canada, Ottawa, ON. NRCC No. 24140. 17 pp.

- Thurman, E.M. 1985. Organic Geochemistry of Natural Waters. Martinus Nijhoff/Dr. W. Junk Publishers, Dordrecht, The Netherlands. 497 pp.
- Turchenek, L.W., M.E. Pigot, L.D. Andriashek, L. Rochefort, and B.J. Nicholson. 1987. Present and potential effects of anthropogenic activities on waters associated with peatlands in Alberta. Draft report prepared for Alberta Environment by Alberta Research Council and University of Alberta. 375 pp.
- Wehrhahn, R., and L. Marciak. 1986. Alberta Agriculture, Edmonton. Personal communication.

A Review of the Important Gradients in Northern Mires and Their Use in Classification in Continental Western Canada

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INTRODUCTION

Mires are characterized by wet, organic soils called peats, which are partially decomposed plant materials that accumulate as a result of slow decomposition in wet, anaerobic conditions. Under the Canadian System of Soil Classification, organic layers must be at least 40-60 cm deep to be recognized as peatlands (Anonymous, 1978). This effectively excludes areas with predominantly shallow peat deposits, where the vegetation is often rooted in the underlying mineral soil, as is the case with marshes and swamps. Peatlands occupying sites with excess water are abundant in boreal and subarctic ecosystems (Moore and Bellamy, 1974). Within the boreal-subarctic area, there are three main peatland regions: a cold, northern, subarctic region; a wet, maritime, oceanic region; and a dry, inland, continental region (Zoltai and Pollett, 1983).

Mires are unique in that they accumulate their own substrate, and thus create their own environment, regardless of the climatic zone in which they are found. The accumulation of peat raises the mire surface above the influence of the original mineral-rich substrate, so that the system is fed mainly by seeping ground water and flowing surface water. Further accumulation of peat can raise the entire system above these minerotrophic influences, so that the system is fed only by mineral-poor rain water. Weber (1908) was one of the first to describe this hydrotopographical development in peatlands from *Niedermoore* (low bogs) to *Hochmoore* (raised bogs).

The variation in peatland vegetation has long been described by Swedish researchers in terms of four major gradients (Tuomikoski, 1942; DuRietz, 1949; Sjors, 1952). The hummock to hollow gradient reflects microtopographic variation within short distances, and can be related to water level. The mire expanse to mire margin gradient is related to physiognomic or shade gradients from an open, non-forested centre to a forested margin. The poor-rich (also called ombrotrophic to minerotrophic) gradient represents a chemical or ionic gradient related to the

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type of water supply within or between peatlands. The oceanic to continental gradient is related to climatic differences between moist, maritime conditions and dry, continental conditions.

Together, these four gradients are reflected in vegetation and represent the variability that can be found within and between peatlands. These four gradients also suggest that peatlands can be classified on the basis of several independent factors. Considering these factors, peatlands have been distinguished by three main criteria: vegetation, hydrotopography and chemistry.

VEGETATION

Vegetation studies assume that vegetation is the best integrator and thus indicator of the environment. The vegetation is intimately related to both the physical and chemical environment from which it derives its resources. Some species have strict requirements and a narrow range of environmental tolerances which limit their distribution to specific sites. Those species that are exclusive to a certain site type are good indicator species of that environment. Those that show a marked preference for a certain site type, although also present to a smaller extent in other environments are labelled characteristic species. Many peatland plant species have broad limits of tolerances to physical gradients, possibly as a consequence of having developed physiological races (Longton, 1974; Pakarinen, 1979), and therefore, they have limited value as indicator or characteristic species. The physiognomy of the vegetation can be used to distinguish between peatland types. Stratification of vegetation has a strong affect on the micro-climate within the peatland, and can be considered an important feature in distinguishing peatland types. Many early classifications of peatlands distinguished between forest, shrub, sedge and moss peatlands. In addition, one of the most basic divisions in peatland vegetation is between *Sphagnum*, (Sphagnopida) dominated peatlands which are relatively acidic, and brown moss (Bryopsida) dominated peatlands which are more alkaline.

HYDROTOPOGRAPHY

One of the fundamental divisions in peatlands is between ombrotrophic or rain-fed peatlands and minerotrophic or ground water fed peatlands. These two terms can be related to the ombrogenous and geogenous peatlands of von Post and Granlund (1926) and also Weber's (1908) morphological division of peatlands into *Hochmoore*, *Ubergangsmoore* (transitional), and *Niedermoores*. Kulczynski (1949) used ombrophilous, transitional, and rheophilous to indicate

vegetation tolerating ombrotrophic and minerotrophic conditions. In addition, topogenous peatlands, with stagnant or slow moving waters showing mainly vertical oscillations in the water table, are distinguished from soligenous ones with flowing or horizontal water movements. Vertical water movements can sometimes have similar affects on vegetation as does flowing water at a constant level (Malmer, 1986). Classifications based on hydrotopography (Radforth, 1952; Bellamy, 1968; Goode, 1973) do not necessarily correspond with those based on vegetation. Hydrotopography affects the vegetation in two different ways that are often difficult to distinguish, that is directly through water level and fluctuation and indirectly through nutrient availability in the water.

CHEMISTRY

Peatlands, as a whole, are considered nutrient-poor in comparison to upland sites where the vegetation is rooted in a more nutrient-rich mineral soil. The supply of nutrients in a peatland depends on the quality and flow rate of the water entering the peatland (Sjors, 1952). The quality of nutrients in the water depends on the chemical nature of its origin, and whether it is through atmospheric deposition or contact with the superficial and/or bedrock geology of the area. The fundamental division of peatlands into ombrotrophic wherein the water supply is exclusively from precipitation and minerotrophic wherein the water supply is originally derived from surface or ground water has often been studied with reference to pH, conductivity, calcium, magnesium, potassium and sodium content of the water. Calcium, magnesium and sodium are by far the most abundant cations present in the ground water. Nutrients present in much smaller and often limited amounts, such as nitrogen and phosphorus, are actively taken up by plants and are key elements in the fertility status of any ecosystem. In this paper, the term nutrient will refer to nitrogen and phosphorus content, while mineral or ionic content will refer to such major ions as calcium, magnesium and sodium.

CORRELATIONS BETWEEN VEGETATION AND ENVIRONMENTAL FACTORS

The correlation between peatland vegetation and element content (often stated as ion content) in surface waters has been extensively studied (Kivinen, 1935; Witting, 1949; DuRietz, 1949; Sjors, 1952, 1959, 1961a, 1961b, 1963; Gorham, 1956; Ritchie, 1957; Henoch, 1960; Jeglum, 1971, 1972; Vitt *et al.*, 1975; Horton *et al.*, 1979; Slack *et al.*, 1980; Sims *et al.*, 1982;

Karlin and Bliss, 1983), but the terminology is not always consistent. The poor-rich vegetation gradient (DuRietz, 1949) in peatlands refers to the low number of indicator species present in *Sphagnum* dominated peatlands as opposed to the relatively high number of indicator species found in brown moss dominated peatlands. This poor-rich vegetation gradient reflects an underlying hydrotopographical and chemical gradient often referred to as the ombrotrophic-minerotrophic gradient (Sjors, 1952), with ombrogenous vegetation formed under ombrotrophic conditions, and geogenous vegetation formed under minerotrophic conditions. The bog-fen concept represents a composite of vegetational, chemical and hydrotopographical gradients. Bogs are poor in the number of indicator species, *Sphagnum* dominated, and ombrotrophic, while fens have a higher number of indicator species, often brown moss dominated, and minerotrophic.

Although the water chemistry *per se* is a major factor in determining the type of vegetation, it is often difficult to separate its effects from hydrologic differences. Sites with similar water chemistry may support different plant communities as a result of different hydrotopographic developments that control the flow rate of water, and indirectly, the amount of minerals and nutrients available. The use of water chemistry as a basis for classification is hampered by seasonal variation in water supply, and it is therefore best to combine such data with flow rates and annual water budgets to produce a complete picture (Moore, 1984). However, in many cases this is not feasible.

Peat chemistry, on the other hand, is less subject to seasonal variation. Seasonal variations in the water supply do not greatly affect the macronutrient content of the peat which is several orders of magnitude larger than that of the water. The chemical analysis of peat (Malmer and Sjors, 1955; Olenin, 1951) has been used in supplement descriptions of peatlands. More recently, attempts have been made to use peat chemistry to characterize peatland types (Zoltai and Tarnocai, 1971; Pollett, 1972; Stanek and Jeglum, 1977). These studies have generally showed ombrotrophic peats to have lower mineral contents than minerotrophic peats, a trend that is found in the water chemistry as well.

CLASSIFICATION

As can be seen, it is not the lack of criteria that makes peatland classification difficult. Rather, it is a question of which criteria are to be considered most important, and whether an agglomerative or divisive approach to classification is taken. The Zurich-Montpelier

(Braun-Blanquet, 1932) approach that stresses vegetation has a strong tradition in central Europe. The basic unit is the plant association defined by its own indicator and characteristic species. This is an agglomerative approach whereby similar associations are grouped together into larger units, regardless of any spatial relations. For example, treed string communities in a patterned peatland may be grouped with forested peatlands, even though the strings are an integral part of open, non-treed patterned peatlands.

The Finnish approach (Cajander, 1913; Ruuhijarvi, 1960; Eurola, 1962) is based on the site-type concept, where habitats that are ecologically similar are considered to also support a similar vegetation. Using a divisive approach, seven main units of peatland vegetation are distinguished; each defined by its indicator species and four physical parameters: the trophic status, the water level associated with the hummock to hollow gradient, the source and flow of water, and the physiognomy of the vegetation (Eurola *et al.*, 1984). One problem is the lack of perfect correspondence between vegetational, hydrotopographical and chemical criteria, such that the defining physical parameters may each cover more than one category. This approach, nevertheless, serves to compartmentalize peatlands into types based on distinct criteria, and can be practical for broad-based studies and non-specialists (Allington, 1961; Jeglum *et al.*, 1974; Wells, 1980).

Other approaches have been based on one or two criteria considered to be important. In most of these studies, three basic peatland types have been recognized:

Peatland Types	Reference
Hochmoore; Übergangsmoore; Nidermoore	Weber, 1908
Oligotrophic; Mesotrophic; Eutrophic	Tsinzerling, 1938
Ombrophilous; Transitional; Rheophilous	Kulczynski, 1949
Bog; Poor Fen; Rich Fen	DuRietz, 1949

At this level of differentiation, the above terms all describe the same three types of peatlands. A more detailed description of these three basic types is given below, adapted from DuRietz (1949).

THE BOG POOR FEN-RICH FEN CONCEPT

DuRietz's (1949) original division of peatlands into bogs, poor fens and rich fens was based on vegetation criteria, and more precisely, the number of indicator species. The poor-rich gradient in peatland vegetation refers to the variation from *Sphagnum* dominated communities with a poor or low number of indicator species, to brown moss dominated communities with a rich or large number of indicator species. A summary of indicator and characteristic species that have been used to define these peatland types has been given in Chee and Vitt (1989), and is presented here based on studies in Alberta (Tables 1 and 2).

Sjors (1952) arranged the vegetation types of DuRietz along a water chemistry gradient using pH and conductivity gradients. He found the poor-rich vegetation gradient corresponded well to a pH gradient. Because the poor-rich vegetation gradient parallels the ombrotrophic-minerotrophic hydrotopographic gradient, the two sets of terms have often been used synonymously. A synthesis of the characteristics of the bog-fen gradient is as follows.

Bogs

Bogs are *Sphagnum* dominated peatlands having a small number of indicator species that include *Sphagnum cuspidatum*, *S. balticum*, *Mylia anomala* and ground lichens (DuRietz, 1949; Sjors, 1950, 1961a, 1982; Vitt and Slack, 1975; Malmer, 1986). The vegetation is ombrophilous, receiving water only from precipitation. Bogs are usually raised or otherwise isolated from the influence of ground water that is relatively mineral rich. Consequently, the mineral content of bogs is extremely low (Sjors, 1952; Malmer, 1965; Vitt and Slack, 1975; Horton *et al.*, 1979). As such, geogenous fen plants (those requiring a higher mineral content), such as *Menyanthes trifoliata*, *Carex rostrata*, *Carex lasiocarpa* and *Sphagnum fallax* are absent. In Alberta, ombrogenous vegetation in the Caribou Mountains with *Picea mariana* and *Sphagnum* dominated vegetation, especially *S. fuscum*, *S. magellanicum* and ground lichens of the genera *Cladina*, *Cetraria* and *Cladonia* has been described (Horton *et al.*, 1979), while black spruce bog islands dominated by *Sphagnum angustifolium*, *Sphagnum fuscum* and *Sphagnum magellanicum* has also been described at Mariana Lakes (Nicholson, 1987). Bog Islands, usually associated with stagnant water caused by divergence around protruding upland islands or peninsulas, are landscape features of northern Alberta.

Table 1. Distribution of common vascular plant species in fens of Alberta.

FEN TYPE	POOR FENS	MODERATE-RICH FENS	EXTREME-RICH FENS
<i>Carex limosa</i>			
<i>Menyanthes trifoliata</i>			
<i>Eriophorum chamissonis</i>			
<i>Carex aquatilis</i>			
<i>Scheuchzeria palustris</i>			
<i>Carex pauciflora</i>			
<i>Kalmia polifolia</i>			
<i>Ledum groenlandicum</i>			
<i>Picea mariana</i>			
<i>Betula grandulosa</i>			
<i>Andromeda polifolia</i>			
<i>Salix pedicellaris</i>			
<i>Larix laricina</i>			
<i>Rubus arcticus</i>			
<i>Carex rostrata</i>			
<i>Carex tenuis</i>			
<i>Carex Diandra</i>			
<i>Eriophorum polystachion</i>			
<i>Caltha palustris</i>			
<i>Carex praegracilis</i>			
<i>Carex disperma</i>			
<i>Rumex occidentalis</i>			
<i>Stellaris lonchophylla</i>			
<i>Epilobium palustre</i>			
<i>Galium labradoricum</i>			
<i>Potentilla palustris</i>			
<i>Carex lasiocarpa</i>			
<i>Betula pumila</i>			
<i>Triglochin maritima</i>			
<i>Petasites sagittata</i>			
<i>Myrica gale</i>			
<i>Tofieldia glutinosa</i>			
<i>Muhlenbergia glomerata</i>			
<i>Scirpus cespitosus</i>			
<i>Scirpus hudsonianus</i>			

Table 2. Distribution of common bryophyte species in fens of Alberta.

FEN TYPE	POOR FENS	MODERATE-RICH FENS	EXTREME-RICH FENS
<i>Drepanocladus exannulatus</i>			
<i>Sphagnum jensenii</i>			
<i>Calliergon stramineum</i>			
<i>Sphagnum fallax</i>			
<i>Sphagnum angustifolium</i>			
<i>Sphagnum magellanicum</i>			
<i>Sphagnum fuscum</i>			
<i>Tomenthypnum falcifolium</i>			
<i>Sphagnum subsecundum</i>			
<i>Drepanocladus lapponicus</i>			
<i>Sphagnum warnstorffii</i>			
<i>Aulacomnium palustre</i>			
<i>Tomenthypnum nitens</i>			
<i>Drepanocladus vernicosus</i>			
<i>Sphagnum squarrosum</i>			
<i>Drepanocladus sendtneri</i>			
<i>Brachythecium turgidum</i>			
<i>Plagiomnium ellipticum</i>			
<i>Amblystegium serpens</i>			
<i>Calliergonella cuspidata</i>			
<i>Helodium blandowii</i>			
<i>Hypnum pratense</i>			
<i>Drepanocladus aduncus</i>			
<i>Drepanocladus polycarpus</i>			
<i>Brachythecium mildeanum</i>			
<i>Bryum pseudotriquetrum</i>			
<i>Meesia triquetra</i>			
<i>Campylium stellatum</i>			
<i>Calliergon giganteum</i>			
<i>Calliergon trifarium</i>			
<i>Drepanocladus revolvens</i>			
<i>Scorpidium scorpioides</i>			

Fens

Fens have a higher number of indicator species, and generally a greater species richness. Ubiquitous species include *Menyanthes trifoliata*, *Carex rostrata*, *C. lasiocarpa*, *C. chordorrhiza* and *Potentilla palustris*. Additionally, several fen types can be distinguished, each having its own indicator and characteristic species. Fens are minerotrophic (influenced by ground water) and have higher mineral content than does precipitation. The pH is usually greater than 4. The different fen types cannot be distinguished by hydrotopography. Their waters may be topogenous (stagnant) or soligenous (flowing) depending on the slope. Fens may be patterned with a network of drier "strings", with wet "flarks" or pools in between, or they may be non-patterned. Fens can be further distinguished by their vegetation and water chemistry.

Poor Fens

Poor fens are *Sphagnum* dominated, minerotrophic peatlands. Indicator species present in western Canada include *S. lindbergii*, *S. maju*, *S. jensenii*, *S. riparium*, *S. fallax* and *S. pulchrum* (DuRietz, 1949; Sjors, 1950, 1982; Ruuhijrvi, 1960; Persson, 1961; Malmer, 1986). Other species such as *Drepanocladus exannulatus* and *S. subsecundum* may also be present (Sjors, 1963; Vitt *et al.*, 1975). The ground waters of these peatlands have a pH of 4.0 - 5.5 (Sjors, 1952, 1963; Persson, 1961; Malmer, 1965; Gorham, 1967; Vitt *et al.*, 1975; Karlin and Bliss, 1984). In the Swan Hills of Alberta, non-forested, minerotrophic peatlands dominated by *S. jensenii*, *S. maju* and *Drepanocladus exannulatus*, and with pH of 4.2 - 5.8 were considered as poor fens (Vitt *et al.*, 1975). Nicholson (1987) also reported *Sphagnum fallax* and *Sphagnum jensenii* dominated poor fens at Mariana Lakes in northeastern Alberta.

Rich Fens

Rich fens are brown moss dominated, minerotrophic peatlands. Critical indicator species include *Drepanocladus revolvens*, *Campylium stellatum*, *Bryum pseudotriquetrum*, *Tomenthypnum nitens*, along with *Tofieldia glutinosa*, *Triglochin maritima* and *T. palustre* (Sjors, 1961a, 1963; Ruuhijrvi, 1960; Persson, 1961; Malmer, 1986). Surface water pH is 5.7 - 8.4. Two types of rich fens have been recognized: moderate-rich fens and extreme-rich fens.

Extreme-rich fens have additional indicator species of highly calcareous conditions. These include *Scorpidium scorpioides*, *Calliergon trifarium*, *Meesia uliginosa*, *Catoscopium nigratum*,

Calliergon turgescens and vascular plants as *Muhlenbergia glomerata*, *Carex microglochin*, *Habenaria hyperborea*, *Schoenus ferrugineus*, *Epipactis palustris* and *Liparis loeselii* (Sjors, 1950, 1961a, 1963, 1982; Slack *et al.*, 1980; Malmer, 1986). The highly calcareous ground water, often with marl (precipitated calcium carbonate), has a pH in the 7.0 - 8.4 or higher range (Sjors, 1952, 1961a; Malmer, 1965; Gorham, 1967).

Moderate-rich fens, also called transitional rich fens, are less well-defined than extreme-rich fens. They have the basic rich fen indicator species, but not the indicator species of the highly calcareous conditions of extreme-rich fens. Other species of importance that have been noted include *Sphagnum warnstorffii*, *S. teres*, *S. contortum*, *S. subnitens* and *Drepanocladus exannulatus* (Sjors, 1961, 1982; Malmer, 1965, 1986). These species are not particularly exclusive nor most abundant in moderate-rich fens. Reported pH of this peatland types ranges from 5.5 - 7.3.

Rich fen studies have not always made a distinction between extreme-rich fens and moderate-rich fens. In the Rocky Mountain Foothills of Alberta, non-forested minerotrophic fens with an abundance of *Scorpidium scorpioides*, *Campylium stellatum*, *Calliergon trifarium*, *Catoscopium nigratum*, *Meesia triquetra* and *Triglochin palustre*, along with pH of 6.8 - 7.9, were identified as rich fens (Slack *et al.*, 1980). Based on both vegetation and surface water chemistry, these fens would be classified as extreme rich fens, whereas fens described in the Athabasca area are clearly moderate-rich fens.

Sjors (1952) pointed out that the three main peatland types are not compartments with strict boundaries, but form nodes along a continuum. This gradient approach has also been favoured in British studies (Bellamy, 1968; Daniels, 1978). Sjors recognized transitions between the three basic types, especially between moderate-rich fens as described above, and intermediate fens.

Intermediate Fens

DuRietz's (1949) original division of peatlands into bogs, poor fens and rich fens was modified by Sjors (1952) and Persson (1961, 1962) to include a fourth class, that of intermediate fens. Sjors originally described these fens as having the less exclusive species of rich fens, mixed with species of poor fens, but Persson believed them to have characteristic species as well. At present, intermediate fens can be described as distinct entities with *Scapania paludicola*,

Calliergon sarmentosum, *Odontoschisma elongatum*, *Cinclidium subrotundum*, and *Sphagnum subfulvum* as characteristic species (Sjors, 1952, 1982; Persson, 1961; Sonneson, 1970a, b; Malmer, 1986). Also present are such poor fen species as *Sphagnum papillosum*, *S. subsecundum*, *Drepanocladus exannulatus*, and such rich fen species as *Paludella squarrosa*, *Drepanocladus badius*, *Sphagnum warnstorffii*, *S. teres*, *Bryum pseudotriquetrum* and *Campylium stellatum*. The water chemistry of these fens have a pH range of 5.2 - 7.0 (Sjors, 1952; Persson, 1961, 1962), which is approximately the same range as that described for moderate rich fens. Fens such as these have not been described in Alberta, nor in North America, although the term, intermediate fen, has been used to designate a variety of peatland types that do not correspond to the original Scandinavian definitions.

SUMMARY

Relationships between hydrology, topography, vegetation, water chemistry, and peat chemistry can be used to distinguish peatland types in western Canada (Fig. 1). A corresponding classification can be devised to represent these relationships and we suggest the use of the bog-fen concepts as proposed by Scandinavian scientists. Chemically, bogs and fens are best distinguished along an acidity-alkalinity gradient that also includes differences in calcium, magnesium, sodium, and electrical conductivity; surface waters in these peatlands have little differences in nutrient contents. Wetland systems (swamps and marshes) differ in less peat formation coupled with higher amounts of nutrients in the surface waters. An understanding of the relationships between vegetation and the chemistry of the system will lead to better management and use of these important northern ecosystems.

Figure 1. Summary of the relationships between chemical characteristics and a wetland classification into bog, fen, swamp, and marsh. Modified from Turchenek *et al.* (1987).

REFERENCES

- Anonymous. 1978. The Canadian system of soil classification. Canada Soil Survey Committee, Subcommittee on Soil Classification, Canada Department of Agriculture. Publication No. 1646. 164 pp.
- Allington, K.R. 1961. The bogs of central Labrador - Ungava; an examination of their physical characteristics. *Geografisk Annaler* (Stockholm) 43:401-417.
- Bellamy, D.J. 1968. An ecological approach to the classification of European mires. In: *Proceedings of the 3rd International Peat Congress*. Quebec, Canada. August 18-23, 1968.
- Braun-Blanquet, J. 1932. *Plant Sociology: The Study of Plant Communities*. Translated by G.D. Fuller, and Conrad, H.S. McGraw-Hill, New York, NY.
- Cajander, A.K. 1913. Studien ber die Moore Finnlands. *Acta Forestalia Fennica*, 2:1-208.
- Daniels, R.E. 1978. Floristic analyses of British mires and mire communities. *J. Ecol.* 66:773-802.
- DuRietz, G.E. 1949. Huvudenheter och huvugranser i svensk myrvegetation. *Svenska Botaniska Tidskrift* 43:274-309.
- Eurola, S. 1962. Uber die Regionale Einteilung der Sdfinnischen Moore. *Annales Botanici Societatis Zoologicae Botanicae Fennicae 'Vanamo'* 33:1-243.
- Eurola, S., S. Hicks, and E. Kaakinen. 1984. Key to Finnish mire types. In: *European Mires* (Moore, P.D., Ed.). Academic Press, London, UK.
- Goode, D.A. 1973. The significance of hydrology in the morphological classification of mires. In: *Proceedings of the International Peat Society Symposium*, Glasgow, UK.
- Gorham, E. 1956. The ionic composition of some bog and fen waters in the English Lake District. *J. Ecol.* 44:142-152.
- Gorham, E. 1967. Some chemical aspects of wetland ecology. Tech. Mem. Assoc. Committee on Geotechnical Research, National Research Council of Canada. No. 90.
- Henoch, W.E.S. 1960. String bogs in the Arctic 400 miles north of the treeline. *Geograph. J.* CXXVI:335-339.
- Horton, D.G., D.H. Vitt, and N.G. Slack. 1979. Habitats of circumboreal-subarctic *Sphagna*. I. A quantitative analysis and review of species in the Caribou Mountains, northern Alberta. *Can. J. Bot.* 57:2283-2317.

- Jeglum, J.K. 1971. Plant indicators of pH and water levels in peatlands at Candle Lake, Saskatchewan. *Can. J. Bot.* 49:1661-1676.
- Jeglum, J.K. 1972. Boreal forest wetlands near Candle Lake, central Saskatchewan. *Musk-Ox*, 11:41-58.
- Jeglum, J.K., A.N. Boissoneau, and V.F. Haavisto. 1974. Toward a wetland classification for Ontario. Department of Environment, Canadian Forestry Service Information. Report 0-X-215.
- Karlin, E.F., and L.C. Bliss. 1984. Variation in substrate chemistry along microtopographical and water chemistry gradients in peatlands. *Can. J. Bot.* 62:142-152.
- Kivinen, E. 1935. Über Electrolytgehalt und Reaktion der Moorwasser. *Maatouskoelaitoksen Maatutkimusosasto Agreogeol. Julkaisu* 38. Helsingfors.
- Kulczynski, M.S. 1949. Peat bogs of Polesie. *Memoires de l'Academie Polonaise des Sciences et des Lettres, Classe des Sciences Mathematiques et Naturelles, Serie B: Sciences Naturelles.* No.15. 356 pp.
- Longton, R.E. 1974. Genecological differentiation in bryophytes. *J. Hattori Bot. Lab.* 38:49-65.
- Malmer, N. 1965. The southern mires. In: *The Plant Cover of Sweden.* *Acta Phytogeographica Suecica* 50:149-158.
- Malmer, N. 1986. Vegetational gradients in relation to environmental conditions in northwestern European mires. *Can. J. Bot.* 64:375-383.
- Malmer, N., and H. Sjors. 1955. Some determinants of elementary constituents in mire plants and peat. *Botaniska Notiser* 108:46-80.
- Moore, P.D., and D.J. Bellamy. 1974. *Peatlands.* Springer-Verlag, New York, NY.
- Moore, P.D. 1984. Classification of mires. In: *European Mires* (Moore, P.D., Ed.). Academic Press, London, UK.
- Nicholson, B.N. 1987. Peat paleoecology and peat chemistry at Mariana Lakes, Alberta. M.Sc. Thesis, Department of Botany, University of Alberta, Edmonton, AB.
- Olenin, A.S. (Ed.). 1951. *Klassifikatsiya vidov torfa i torfyanykh zalezhei.* *Glamoe Upravlenie Torfyanogo Fonda,* Moscow, USSR.
- Pakarinen, P. 1979. Ecological indicators and species groups of bryophytes in boreal peatlands. In: *Proceedings of the International Symposium on Classification of Peat and Peatlands, September 17-21, 1979.* Hyytil, Finland.

- Persson, A. 1961. Mire and spring vegetation in an area north of Lake Tornetrask, Torne Lappmark, Sweden. I. Description of vegetation. *Opera Botanica* 6:1-187.
- Persson, A. 1962. Mire and spring vegetation in an area north of Lake Tornetrask, Torne Lappmark, Sweden. II. Habitat conditions. *Opera Botanica* 6:1-100.
- Pollett, F.C. 1972. Classification of peatlands, Newfoundland. In: Proceedings of the 4th International Peat Congress, Helsinki, Finland.
- Radforth, N.W. 1952. Suggested classification of muskeg for the engineer. *Eng. J.* 35:1199-1210.
- Ritchie, J.C. 1957. The vegetation of northern Manitoba. II. A prisere of the Hudson Bay Lowlands. *Ecol.* 38:429-435.
- Ruuhijrvi, R. 1960. Über die Regionale Einteilung der Nordfinnischen Moore. *Annales Botanici Societatis Zoologicae Botanicae Fennicae 'Vanamo'* 31:1-360.
- Sims, R.A., D.W. Cowell, and G.M. Wickware. 1982. Classification of fens near southern James Bay, using vegetation physiognomy. *Can. J. Bot.* 60:2608-2623.
- Sjors, H. 1950. Regional studies in north Swedish mire vegetation. *Botaniska Notiser* 1950:174-221.
- Sjors, H. 1952. On the relation between vegetation and electrolytes in north Swedish mire waters. *Oikos* 2:242-258.
- Sjors, H. 1959. Bogs and fens in the Hudson Bay Lowlands. *Arctic* 12:1-19.
- Sjors, H. 1961a. Forest and peatland at Hawley Lake, northern Ontario. *National Museum of Canada. Bulletin* 171:1-31.
- Sjors, H. 1961b. Surface patterns in boreal peatlands. *Endeavor* 20:217-224.
- Sjors, H. 1963. Bogs and fens on the Attawapiskat River, northern Ontario. *National Museum of Canada. Bulletin* 186, Contributions to Botany, 1960-1961.
- Sjors, H. 1982. Mires of Sweden. In: *Ecosystems of the World. Mires: Swamp, Bog, Fen and Moor*. 4B. Regional Studies. (Gore, A.J.P., Ed.). Elsevier, The Netherlands.
- Slack, N.G., D.H. Vitt, and D.G. Horton. 1980. Vegetation gradients of minerotrophically rich fens in western Alberta. *Can. J. Bot.* 58:330-350.
- Stanek, W., and J.G. Jeglum. 1977. Comparisons of peatland types using macro-nutrient contents of peat. *Vegetation* 33:163-173.

- Tuomikoski, R. 1942. Untersuchungen ber die Untervegetation der Bruchmoore in Ostfinnland. I. Zur Metodik der Pflanzensoziologischen Systematik. *Annales Botanici Societatis Zoologicae Botanicae Fennicae 'Vanamo'* 17:1-203.
- Turchenek, L.W., M.E. Pigot, W.D. Andriashek, L. Rochefort, and B. Nicholson. 1987. Present and potential effects of anthropogenic activities on waters associated with peatlands in Alberta. Draft Report Prepared for Research Management Division, Alberta Environment, Edmonton, AB.
- Tsinzerling, Y.D. 1938. Rastitel'nost'bolot. pp. 355-428. In: Rastitel'n. S.S.R. 1. Akad. Nauk S.S.S.R., Moscow, Leningrad.
- Vitt, D.H., P. Achuff, and R.E. Andrus. 1975. The vegetation and chemical properties of patterned fens in the Swan Hills, north central Alberta. *Can. J. Bot.* 53:2776-2795.
- Vitt, D.H., and N.G. Slack. 1975. An analysis of the vegetation of Sphagnum-dominated kettle-hole bogs in relation to environmental gradients. *Can. J. Bot.* 53:332-359.
- von Post, L., and E. Granlund. 1926. Södra Sveriges torvtillgångar. *Sveriges Geol. Unders. Ser. C.* 335:1-127.
- Weber, C.A. 1908. Aufbau und Vegetation der Moore Norddeutschlands. *Botanische Jahrbücher für systematik (Leipzig)* 90:19-34.
- Wells, E.D. 1980. Peatlands of eastern Newfoundland: Distribution, morphology, vegetation and nutrient status. *Can. J. Bot.* 59:1978-1997.
- Witting, M. 1949. Kalciumhalten i några nordsvenska myrvegetation. *Svenska Botaniska Tidskrift* 43:2-3.
- Zoltai, S.C., and F.C. Pollett. 1983. Wetlands in Canada: Their classification, distribution and use. In: *Ecosystems of the World. Mires: Swamp, Bog, Fen and Moor. 4B Regional Studies.* (Gore, A.J.P., Ed.). Elsevier, The Netherlands.
- Zoltai, S.C., and C. Tarnocai. 1971. Properties of a wooded peatland in northern Manitoba. *Arct. Alp. Res.* 3:115-129.

General Characteristics of Brownwater Lakes in Northeastern Alberta

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ABSTRACT

Two synoptic surveys of 47 remote lakes in northeastern Alberta were conducted by aircraft in winter and summer during 1988. The lakes are located in the Birch Mountains Upland, the Muskeg Mountain Upland, and in the Athabasca Plain. The purpose of this work was to refine the sensitivity mapping originally published by the Western Canada-LRTAP Program.

A wide range of morphometric and edaphic characteristics typify the lakes of the region. Shallow headwater lakes on peatlands in these upland areas were heavily stained, low in dissolved solids, and naturally acidic. Larger lakes in the uplands were also stained but contained sufficient minerals to produce measurable alkalinity. Summer thermal stratification was weak in the deeper, upland lakes and hypolimnia were well oxygenated. By comparison, the lakes on the Athabasca Plain were extremely transparent, and high in dissolved solids. The chemical data will be discussed with reference to acidic and trophic status models.

The Prediction of Water Quality in Hutch Lake Reservoir - A Brown Water System

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ABSTRACT

Hutch Lake is located in northwestern Alberta, approximately 32 km north of the town of High Level. It is a shallow, productive brown water lake and it undergoes severe oxygen depletion in winter. Fish do not presently overwinter in the lake.

A proposal to raise the level of Hutch Lake by 6.5 m was approved in March 1987. A dam has been constructed downstream of the lake outlet, on the Meander River (Fig. 1). The dam will create a new reservoir with a surface area of 5.6 km² and a mean depth of 3.8 m (Table 1). The objective of this paper is to predict whether the newly created reservoir will have the capability to overwinter a sportfish population.

Surface water quality parameters for the existing lake were measured from samples collected from January 1987 to March 1988. The results indicate the poor water quality of the existing lake (Table 2). The dissolved oxygen level measurements demonstrate the oxygen depletion which the lake undergoes, especially in winter months (Fig. 2).

Although the new reservoir will have a much larger surface area (5.6 km²) than the existing lake (1.2 km²), mean depth will only increase from 1.0 to 3.8 m due to extensive flooding of low lying muskeg at the upstream end of the lake. The flooded muskeg will represent close to 50% of the total surface area of the new reservoir and will be less than 2 m deep.

Flooding of the upstream muskeg area will place an additional nutrient load and dissolved oxygen (DO) demand on the lake. It is expected that any gain in DO carrying capacity achieved by the increase in lake volume will be largely negated as a result of the organic-rich flooded muskeg.

The winter oxygen depletion rate (WODR) model developed by Babin and Prepas (1985) was used to predict the WODR in the existing Hutch Lake and the future reservoir. This model relates the WODR to lake morphometry (mean depth) and productivity (open water total phosphorus or chlorophyll a).

Figure 1. Hutch Lake Reservoir Location Plan.

Table 1. Hutch Lake project: morphometry of existing lake and future reservoir.

	Existing Lake	Future Reservoir
Lake elevation (m)	319.6	326.1 (full supply level)
Area (km ²)	1.2	5.6
Volume (10 ⁶ m ³)	1.2	21.4
Depth - mean (m)	1.0	3.8
Length (km)	2.8	12.5
Width - mean (km)	0.5	0.5
Drainage area (km ²)	470	470

Table 2. Surface water quality characteristics for Hutch Lake 1987-88.

Parameter	Open Water n = 6 Mean (mg L ⁻¹)	Under Ice n = 3 Mean (mg L ⁻¹)
Total Phosphorous	0.100	0.75
Total Nitrogen	2.26	3.62
Chlorophyll <u>a</u>	0.107	0.001
Total Ammonia-N	0.23	0.89
Apparent Colour*	200	250
Dissolved Organic Carbon	42.5	75.4
Total Phenols	0.008	0.013
Iron	0.14	0.75
Manganese	0.032	0.762

*Apparent Colour - PT-Co units

Mean summer total phosphorous in the euphotic zone (TP sum) was set at the open water mean for the existing lake. The euphotic zone depth equals approximately 1 m and total phosphorous was estimated to be 100 mg m⁻³ based on results of water quality samples. Therefore the TP sum was set at 100 mg m⁻². It is possible that TP sum values for the reservoir

Figure 2. Dissolved oxygen levels for open water and under-ice conditions in Hutch Lake (samples taken from January 1987 to March 1988; n = 9).

would be greater than mg m^{-2} for several years after reservoir creation due to trophic upsurge and greater availability of phosphorous from the newly flooded sediments.

An initial estimate of 8 mg L^{-1} of DO was used to estimate the length of time required after ice-on for DO to fall below the Alberta Surface Water Quality Objective (ASWQO) of 5 mg L^{-1} . The results of the predicted WODR and time to deplete DO are summarized in Table 3.

Average DO levels are predicted to fall below the ASWQO for aquatic life of 5 mg L^{-1} within three months after freeze-up. Note the vertical distribution of oxygen under ice is seldom uniform and levels immediately below the ice may be higher than those near the sediments.

These results must be interpreted with caution because of the following limitations of the WODR model. First, the model was not intended to predict changes in WODR over relatively

small changes in mean depth. Second, the WODR model was not developed for brown water lakes. The fraction of TP available for algal and macrophyte uptake is reduced in brown water lakes due to humic-iron-phosphate complex formation (Jackson and Hecky, 1980). These complexes also suppress primary productivity through the fixation of trace metals, therefore the TP sum value used may over estimate lake productivity. Despite these limitations, the WODR model is the best available predictive tool, and it was concluded that a sportfish population will not overwinter in the new Hutch Lake reservoir.

Table 3. Hutch Lake project: predicted WODR¹ and time for dissolved oxygen to deplete from 8 to 5 mg L⁻¹ for existing lake and future reservoir.

Parameter	Existing Lake	Reservoir
Area - km ²	1.2	5.86
Volume - m ³	1.2 x 10 ⁶	21.4 x 10 ⁶
Mean Depth - m	1	3.8
Assume:		
TP sum - mg m ⁻²	100	100
[O ₂] - mg L ⁻¹	8	8
Predictions:		
1. WODR - g m ² d ⁻¹	0.16	0.20
2. Time for O ₂ to deplete from 8 to 5 mg L ⁻¹ days	19	55

¹Winter Oxygen Depletion Rate

$$\text{WODR} = -0.101 + 0.00247 (\text{TP sum}) + 0.0134 (z)$$

TP sum = mean summer total phosphorous in the euphotic zone (mg m⁻²)

z = mean depth (m)

WODR expressed as g O₂ m² d⁻¹ (Babin and Prepas, 1985)

REFERENCES

- Babin, J., and E.E. Prepas. 1985. Modelling winter oxygen depletion rates in ice-covered temperature zone lakes in Canada. *Can. J. Fish. Aquat. Sci.* 42:239-249.
- Jackson, T.A., and R.E. Hecky. 1980. Depression of primary productivity by humic matter in lake and reservoir waters of the boreal forestry zone. *Can. J. Fish. Aquat. Sci.* 37:2300-2317.

**Biogeochemistry of Mires, Muskegs,
and Marshes**

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ABSTRACT

Mires are peat-forming ecosystems that are either minerotrophic fens or ombrotrophic bogs. Rich fens are dominated by brown mosses and have water high in alkalinity and base cations. Poor fens are dominated by *Sphagnum* moss and have water with less alkalinity, less base cations, but more hydrogen ion. Bogs are highly acidic, without any alkalinity base. *Sphagnum* moss dominates in ombrotrophic systems. Bogs and fens are both characterized by oligotrophic waters, while other wetlands, namely marshes and swamps, are eutrophic. Distinct chemical differences exist between mires in continental and maritime western Canada, particularly noticeable are differences in sodium, chloride, calcium, and bicarbonate components. Changes in hydrology (e.g., flooding, drainage), geochemistry (e.g. acidification, eutrophication) and climate (e.g., Holocene climate changes, projected temperature rises) all have dramatic effects on mire geochemistry and vegetation pattern.

LAKE REHABILITATION

**Impact of Lime on Water Quality in Fast Flushing Hypereutrophic Lakes:
Year One - Edmonton Stormwater Retention Lakes**

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ABSTRACT

Lime (CaCO_3) and/or Ca(OH)_2 was applied eight times to three stormwater retention lakes in Edmonton between June and August 1988. The purpose was to evaluate the short-term effects of lime on algal biomass and phosphorus concentration in rapidly flushed systems. Dosages of lime applied to the stormwater retention lakes ranged from 50 to 200 mg L^{-1} .

The first tests of the potential for lime to control nutrient and algal biomass levels in rapidly flushed systems are presented. When at least 50 mg L^{-1} Ca(OH)_2 was applied to the lakes, total phosphorus concentration and algal biomass decreased significantly ($P < 0.05$) over pretreatment conditions. When CaCO_3 was applied alone, neither total phosphorus nor algal biomass decreased significantly ($P < 0.05$). Based on these results the ability of lime to decrease algal biomass and nutrient levels in fast flushing lakes is promising.

Evaluation of Calcium Carbonate (CaCO_3) and Calcium Hydroxide (Ca(OH)_2) in Algicides in Prairie Drinking Water Dugouts

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ABSTRACT

Excessive algal biomass is a problem in prairies dugouts that are used for drinking water supplies. Algal biomass detracts from the aesthetic, agricultural, domestic and recreational value of these water bodies. Six drinking water dugouts in northwestern Alberta were studied for two summers. These hardwater dugouts have long water retention periods (5 years) and accumulate nutrients from agricultural runoff. In 1987, four dugouts were treated with high doses of lime ($250 \text{ mg L}^{-1} \text{ CaCO}_3$ or $250 \text{ mg L}^{-1} \text{ Ca(OH)}_2$) to determine the efficiency of each chemical for decreased algal biomass and phosphorus (a limiting nutrient). Algal biomass decreased to 20% of pretreatment levels after CaCO_3 treatments but the algae recovered during the next summer. After Ca(OH)_2 treatments, algal biomass decreased to 1% of pretreatment levels and remained depressed the next year. Phosphorus concentrations were greatly suppressed in Ca(OH)_2 -treated dugouts but little change was noted in CaCO_3 -treated dugouts. Colour decreased in both Ca(OH)_2 and CaCO_3 -treated dugouts. High doses of Ca(OH)_2 have tremendous potential for long-term improvements in water quality.

**Lake Rehabilitation in Alberta
With Emphasis on Algal Control in Eutrophic Waters**

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ABSTRACT

Green scum of excessive algal biomass is a common feature in lakes, ponds and drinking water dugouts in western Canada. The cause of this problem is a high input of the nutrient, phosphorus. Often this input is from natural sources and cannot be diverted before it enters surface waters. Excessive algal biomass can cause taste and odour problems, fish kills and reduced recreational potential. Traditional approaches to control algal biomass such as application of alum or copper sulphate (bluestone), have raised other concerns including long-term toxicity to non-target organisms. Over the past three years, we have worked on evaluating whether lime (CaCO_3 , $\text{Ca}[\text{OH}]_2$) could serve as a low cost, non-toxic alternative for treatment of excessive algal growth. These treatments have included two recreational lakes, over one dozen drinking water dugouts, and three stormwater retention lakes. The results are promising and include short-term and long-term reductions in total phosphorus, chlorophyll *a* and iron concentrations, as well as improvements in water transparency and under-ice dissolved oxygen concentrations. Progress made will be summarized as will areas which require further consideration.

An Evaluation of the Impact of Pure Oxygen on Water Quality and Fisheries of a Deep Eutrophic Lake; Amisk Lake, Year One

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ABSTRACT

Most deep Alberta lakes have high summer algal biomass, high phosphorus levels and incomplete turnover in spring and fall. This results in very low hypolimnetic dissolved oxygen (DO) levels during mid to late summer and mid to late winter. In May 1988, researchers from the University of Alberta, the National Water Research Institute in Burlington, Ontario, and Linde designed and installed a system to inject liquid oxygen into the deep waters of the north basin of Amisk Lake. The main objective of this project was to increase dissolved oxygen levels in the hypolimnion of the north basin, leading to improved water quality (lower algal biomass) due to decreased phosphorus release from sediments) and increased year-round habitat and food resources for sportfish. Preliminary results indicate that oxygenation may be an effective and economical way of increasing hypolimnetic dissolved oxygen levels. In the summer of 1988, hypolimnetic oxygen depletion rates in the north basin were reduced to about 33% of historic rates, and hypolimnetic phosphorus accumulation rates were about 58% of historic rates. Sportfish show a strong preference for dissolved oxygen levels above 2 mg L⁻¹.

**DRINKING WATER AND
URBAN WATER MANAGEMENT**

Water Quality Impacts on the Glenmore Reservoir

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ABSTRACT

Glenmore Reservoir, an Elbow River impoundment, provides source water to the $6.6 \times 10^5 \text{ m}^3 \text{ d}^{-1}$ capacity Glenmore Water Treatment Plant. This treatment plant produces drinking water for approximately half of Calgary's more than 625,000 people. Water quality in the Glenmore Reservoir is affected by drainage from the Elbow River watershed, agricultural activities and, to a lesser extent, domestic sewage inputs from a few small upstream communities. The Glenmore Reservoir itself is surrounded almost entirely by urban development. Water quality is also affected by local runoff, recreational use and inputs from more than 20 storm sewers. A series of water quality studies have been conducted in Glenmore Reservoir and a program has also been established to continuously monitor storm sewer effluents. This paper describes some findings of these programs in three main categories:

- (1) Trace chemical (heavy metals and organic compounds),
- (2) Physical and simple chemical measurements (turbidity, nutrients and ions) and
- (3) Biological aspects (chlorophyll *a* and fecal indicator bacteria).

Although Glenmore Reservoir reflects its urban environment in both chemical and microbial water quality, the retention time provided by the reservoir effectively attenuates sudden loadings from storm events or spring runoff. Microbial and chemical quality gradually improves as water moves through the impoundment to the water treatment plant intakes. Self-purification processes in Glenmore Reservoir reduce the impacts of chronic urban and storm sewer loadings.

INTRODUCTION

The City of Calgary relies on two surface water impoundments for the production of potable water. Glenmore Water Treatment Plant draws water from Glenmore Reservoir (Fig. 1), an impoundment of the Elbow River. The Bears paw Water Treatment Plant obtains water from the Bow River and its impoundment, the Bears paw Reservoir. Glenmore Reservoir is more vulnerable to contamination and water quality deterioration than Bears paw Reservoir for

• 1984 GLENMORE RESERVOIR SAMPLING SITES

△ 1985-86 GLENMORE RESERVOIR SAMPLING SITES

Figure 1. Glenmore Reservoir, Elbow River and storm sewer sampling sites: (1) WH: Elbow River at Weaselhead; (2) Sites 1-6 at surface, middle and bottom depths; (3) GWTP: intakes; (4) G20A, G19, G21, G18, G13: Major storm sewer outfalls.

several reasons. The Elbow River (Table 1) provides only one tenth as much water to Glenmore Reservoir as the Bow River provides to Bearspaw Reservoir. Since Glenmore Reservoir is ten times larger than Bearspaw Reservoir, 80% of the water entering Glenmore Reservoir is eventually drawn into Glenmore Water Treatment Plant for potable water production. Glenmore Reservoir has the limnological characteristics of a lake, with a water retention time of 26 days. In contrast, Bearspaw Reservoir, with a very short retention time of less than half a day, is really only a slower part of the Bow River. Glenmore Reservoir undergoes considerably less flushing than the Bearspaw Reservoir. If a contamination were to occur, it would not quickly pass through the reservoir. Unless Glenmore Water Treatment Plant were shut down, contaminated water would eventually enter the treatment plant. The Glenmore Reservoir retention time does, however, provide time for natural purification processes to occur. Source water quality improves considerably as water moves through the reservoir.

Table 1. Comparison of Calgary's two surface water supplies.

Glenmore Reservoir	Elbow Watershed and Bearspaw Reservoir	Bow River Watershed and Bearspaw Reservoir
Watershed Size	1,220 km ²	7,860 km ²
Annual Yield	2.9 x 10 ⁵ dam ³	2.9 x 10 ⁶ dam ³
Reservoir Capacity	2.8 x 10 ⁴ dam ³	3.0 x 10 ³ dam ³
Reservoir Retention	26 days	0.4 days

The major potential sources of contamination to Glenmore Reservoir include the Elbow River watershed itself, urban runoff through more than twenty storm sewers which directly enter the reservoir, recreational use and accidental spills (Marshall *et al.*, 1983). This paper compares and contrasts the impacts of the two major loading sources, the Elbow River watershed and urban runoff, on water quality in Glenmore Reservoir.

In general, inputs from the Elbow River have the greatest impact on reservoir water quality, but early indications of urban impacts have also been detected in monitoring programs underway in Glenmore Reservoir since 1982. Some results of these programs (Hargesheimer and Lewis, 1985, 1988) will be described, including:

- (1) Trace chemicals (heavy metals and organic compounds),

- (2) Physical and simple chemical measurements (turbidity, nutrients and ions) and
- (3) Biological aspects (chlorophyll *a* and fecal indicator bacteria).

RESERVOIR DESCRIPTION AND MONITORING PROGRAMS

Glenmore Reservoir is composed of several limnologically distinct areas, and water quality within these regions is quite different. The areas are created due to geographical, physical (flow and water level) as well as biological factors unique to those portions of the reservoir. In order to properly evaluate the condition of the reservoir as a whole, it is important to sample at sites which are representative of all of these limnological regions.

For monitoring purposes Glenmore Reservoir has been divided into four quadrants marked A, B, C and D in Fig. 1. (The locations of major storm sewer outfalls included in monitoring programs are designated by arrows.) Water quality data have been collected at sites within each of the four quadrants for the past seven years. The network of sampling sites includes six reservoir locations, one at the Elbow River entry point to the reservoir, and the remaining five at Glenmore Water Treatment Plant source water intakes and at storm sewer outfalls.

The Elbow River enters the reservoir in its north west corner and exits at Glenmore Dam near Glenmore Water Treatment Plant. The sampling point on the Elbow River at Weaselhead (WH) is indicative of water quality entering the reservoir. In the upper portions of the reservoir, recreational activity is light. In the remainder of the reservoir, recreation is restricted to non-contact activities such as sailing or canoeing. No motor boats are allowed. The headpond area between Glenmore Dam and Glenmore Trail causeway is the deepest part of Glenmore Reservoir with water depths of nearly 20 m and recreational activity is forbidden. Site 1, located in quadrant A, is representative of the main body of the reservoir. Sites 2, 3 and 4, in quadrant B, are located in the shallow embayment (often called Heritage Cove) isolated from the main flow channel. This area is heavily used for non-contact recreational activities such as sailing and canoeing. There are also three storm sewer outlets discharging into this area. In quadrant C, Site 5 is located in the main flow channel of the reservoir near a storm sewer outfall which drains the Earl Grey Golf Course and the Lakeview community. Site 6 in quadrant D is located in the narrowest part of the reservoir. Pollutants approaching the treatment plant intakes would pass through this channel.

GENERAL PHYSICAL CHARACTERISTICS OF GLENMORE RESERVOIR

Glenmore Reservoir is dimictic; there are two periods of complete mixing (spring and fall turnover). Both April and November lake turnovers (Fig. 2) cause sudden changes in treatment plant source water turbidity and bacterial counts. Between spring and fall turnover, thermal stratification does not occur in Glenmore Reservoir. Throughout the summer the reservoir is well mixed and this mixing is primarily wind-induced.

A total of 251 Secchi depth measurements were made in Glenmore Reservoir from May 14, 1985 to October 31, 1986. Secchi depth at Sites 1, 2, 5 and 6 (Fig. 1) ranged from 0.1 m to 7.5 m which corresponded to 0.7 to 100% of site depth respectively. These results indicate that light can penetrate through the entire water column to the sediments throughout the reservoir most of the time. Therefore, nearly the entire body of water is available for photosynthetic activity.

ELBOW RIVER WATERSHED IMPACTS ON GLENMORE RESERVOIR WATER QUALITY

The Elbow River itself provides the greatest load of nutrients, sediments and turbidity to Glenmore Reservoir. Turbidity values for the Elbow River at Weaselhead and the Glenmore Water Treatment Plant intakes from January 1982 to August 1986 are summarized in Table 2. Overall turbidity at Weaselhead was significantly greater (one tailed paired t-test, $p = 0.05$) than that observed at Glenmore Water Treatment Plant intakes. For each year of the five year period summarized in Table 2, maximum turbidity values and the greatest range of turbidity values were observed at Weaselhead.

Input into Glenmore Reservoir from the Elbow River has a more pronounced impact on turbidity than input from storm sewers. Increases in turbidity at Weaselhead are typically followed several days later by an attenuated increase in turbidity at the Glenmore Water Treatment Plant intakes. On September 12, 1985 more than 92 mm of rain fell in 24 h. Following the rain storm, turbidity in the Elbow River at Weaselhead peaked at 98.0 NTU on September 14 as shown in Fig. 3. As the storm flow proceeded through the reservoir, increases in turbidity were both delayed and attenuated. Turbidity values reached a maximum of 34 NTU at Glenmore Water Treatment Plant intakes three days later on September 17.

Figure 2. Indications of Glenmore Reservoir turnover. Typical changes in GWTP intakes water turbidity (NTU), Standard Plate Count (CFU mL⁻¹) and temperature (°C): (1) Spring (March to May, 1985); (2) Fall (October to November, 1985)

Table 2. Turbidity (NTU) in water samples from the Elbow River at Weaselhead and GWTP intakes (January, 1982 to August, 1986).

Sample			Turbidity (NTU)			
Year	Site	Number	Mean	Std. Dev. ¹	Maximum	Minimum
1982	Weaselhead	25	13.9	20.3	81.0	0.5
	GWTP Intakes	61	2.0	2.2	9.3	0.3
1983	Weaselhead	28	5.3	9.9	52.0	0.4
	GWTP Intakes	44	1.7	2.4	13.0	0.4
1984	Weaselhead	29	2.5	10.9	18.0	0.5
	GWTP Intakes	48	1.1	0.8	4.0	0.3
1985	Weaselhead	24	5.1	6.5	27.0	0.7
	GWTP Intakes	45	1.0	0.8	3.3	0.3
1986	Weaselhead	22	18.1	34.4	150.0	1.5
	GWTP Intakes	31	2.0	3.8	21.0	0.4
Overall	Weaselhead	128	8.5	18.4	150.0	0.4
	GWTP Intakes	229	1.5	2.2	21.0	0.3

¹Std. Dev. = Standard Deviation.

Figure 3. Increases in turbidity (NTU) observed at Weaselhead on the Elbow River (WH) in response to a rain storm reached the GWTP intakes (GR) several days later.

Natural "self-purification" processes are operating in Glenmore Reservoir, resulting in improved bacteriological water quality at the Glenmore Water Treatment Plant intakes. As water travels through the reservoir, turbidity is reduced through settling, and bacterial counts decrease through settling and die-off. Fecal coliform bacteria densities (Fig. 4) decrease as water travels from Weaselhead to Glenmore Water Treatment Plant intakes. Bacterial densities are considerably higher entering the reservoir in Elbow River water than those detected at the treatment plant intakes or in the body of the reservoir. Fecal coliform densities in the Elbow River at Weaselhead, like turbidity, fluctuate in response to Elbow River flow. Periods of high Elbow River flow correspond to high turbidity as well as high fecal coliform densities (Fig. 5).

The acceptable pH range for drinking water is 6.5 to 8.5 (CCREM, 1987). Table 3 summarizes annual pH values in samples from the Elbow River at Weaselhead and Glenmore Water Treatment Plant intakes from 1982 to 1986. Seasonal variations in pH are low and water at both sites is well buffered. During the five year period 1982 to 1986, there was no evidence of long term trend in pH levels. No significant alteration in annual pH toward a more alkaline or acidic state was apparent from one year to the next.

Alkalinity (Table 4), specific conductance (Table 5), and hardness (Table 6) in Glenmore Reservoir are also controlled largely by Elbow River flow events. Temporal cycles in specific conductance (Fig. 6) observed in the Elbow River at Weaselhead and Glenmore Water Treatment Plant intakes are offset. Seasonal changes are observed first in the Elbow River; similar values in specific conductance are then observed approximately one month later at Glenmore Water Treatment Plant intakes in response to the changing specific conductance of water entering the reservoir. Minimum values of all three of these parameters occur in the Glenmore Reservoir and Elbow River during June and July as a result of spring runoff dilution.

As in many lakes, a natural process of eutrophication is underway in Glenmore Reservoir. Presently, chlorophyll *a* concentrations are generally less than $2 \mu\text{g L}^{-1}$ in Glenmore Reservoir (Fig. 7). These levels are very low compared to other Alberta lakes (Prepas, 1983) which contain average summer chlorophyll *a* values of 2 to $89 \mu\text{g L}^{-1}$. It is interesting to note that chlorophyll *a* concentrations increase as water moves through the reservoir to Glenmore Water Treatment Plant intakes (Fig. 7). Highest concentrations of chlorophyll *a* are detected at the treatment plant intakes. This indicates that algae are multiplying in the reservoir. Even though these algae are not harmful, they are often responsible for taste and odour problems in drinking water. The

Figure 4. Least significance difference plot of fecal coliforms (log CFU 100 mL⁻¹) in the Glenmore reservoir sampling network May 1985 to May 1986.

Figure 5. Effect of Elbow River flow on turbidity (NTU) and fecal coliforms ($\text{CFU } 100 \text{ mL}^{-1}$) at Weaselhead on the Elbow River January 1982 to August 1986.

Table 3. pH in water samples from the Elbow River at Weaselhead and GWTP intakes (January 1982 to August 1986).

Sample			pH			
Year	Site	Number	Mean	Std. Dev. ¹	Maximum	Minimum
1982	Weaselhead	28	8.25	0.09	8.47	8.03
	GWTP Intakes	47	8.22	0.22	8.53	7.80
1983	Weaselhead	36	8.28	0.05	8.39	8.14
	GWTP Intakes	52	8.26	0.14	8.48	7.98
1984	Weaselhead	38	8.27	0.08	8.41	8.10
	GWTP Intakes	49	8.28	0.15	8.51	7.91
1985	Weaselhead	34	8.15	0.17	8.30	7.92
	GWTP Intakes	51	8.15	0.18	8.51	7.86
1986	Weaselhead	27	8.20	0.06	8.31	8.80
	GWTP Intakes	33	8.22	0.15	8.41	7.96
Overall	Weaselhead	163	8.23	0.11	8.51	7.92
	GWTP Intakes	232	8.22	0.17	8.53	7.80

¹Std. Dev. = Standard Deviation

Table 4. Alkalinity (mg L⁻¹) in water samples from the Elbow River at Weaselhead and GWTP intakes (January 1982 to August 1986).

Sample			pH			
Year	Site	Number	Mean	Std. Dev. ¹	Maximum	Minimum
1982	Weaselhead	28	165.6	15.2	182.4	128.5
	GWTP Intakes	47	158.5	15.5	191.4	132.6
1983	Weaselhead	36	155.1	10.7	67.2	126.3
	GWTP Intakes	52	150.4	23.5	191.1	120.8
1984	Weaselhead	38	152.9	9.0	172.6	132.7
	GWTP Intakes	49	145.3	19.0	177.9	116.4
1985	Weaselhead	34	146.8	8.8	173.6	132.4
	GWTP Intakes	51	147.5	22.2	181.5	114.2
1986	Weaselhead	27	149.8	8.6	164.0	132.7
	GWTP Intakes	33	149.1	16.0	176.4	128.8
Overall	Weaselhead	163	153.8	12.1	182.4	126.3
	GWTP Intakes	232	150.1	20.2	191.4	114.2

¹Std. Dev. = Standard Deviation

Table 5. Specific conductance ($\mu\text{mhos cm}^{-1}$) in water samples from the Elbow River at Weaselhead and GWTP intakes (January 1982 to August 1986).

Sample			Specific Conductance			
Year	Site	Number	Mean	Std. Dev. ¹	Maximum	Minimum
1982	Weaselhead	28	367.5	36.7	417.0	276.0
	GWTP Intakes	47	371.4	49.7	459.0	284.6
1983	Weaselhead	36	361.3	36.0	430.0	299.0
	GWTP Intakes	52	359.3	53.6	453.0	366.0
1984	Weaselhead	38	395.4	26.2	433.0	339.0
	GWTP Intakes	49	381.0	26.8	442.0	305.0
1985	Weaselhead	34	382.1	26.0	445.0	328.0
	GWTP Intakes	51	387.9	49.6	475.0	326.0
1986	Weaselhead	27	382.2	38.9	446.0	304.0
	GWTP Intakes	33	384.3	51.1	451.0	306.0
Overall	Weaselhead	163	378.1	34.6	446.0	276.0
	GWTP Intakes	232	376.3	49.8	475.0	266.0

¹Std. Dev. = Standard Deviation

Table 6. Hardness (mg L^{-1}) in water samples from the Elbow River at Weaselhead and GWTP intakes (January 1982 to August 1986).

Sample			Hardness			
Year	Site	Number	Mean	Std. Dev. ¹	Maximum	Minimum
1982	Weaselhead	27	211.9	21.5	240.0	166.6
	GWTP Intakes	46	212.1	27.2	258.0	170.2
1983	Weaselhead	36	208.2	17.7	231.6	166.2
	GWTP Intakes	52	203.1	28.5	254.1	164.8
1984	Weaselhead	38	219.4	14.3	238.0	182.6
	GWTP Intakes	49	209.9	24.8	246.6	171.0
1985	Weaselhead	34	203.9	12.9	232.6	175.0
	GWTP Intakes	51	206.0	30.6	258.6	161.0
1986	Weaselhead	27	202.1	18.5	229.6	159.6
	GWTP Intakes	33	199.8	25.1	233.4	160.8
Overall	Weaselhead	162	209.5	18.0	240.0	159.6
	GWTP Intakes	231	206.5	27.6	258.6	160.8

¹Std. Dev. = Standard Deviation.

Figure 6. Temporal cycles in specific conductance in water from the Glenmore Water Treatment Plant intakes (GR) and the Elbow River at Weaselhead (WH) January 1982 to August 1986.

Figure 7. Least significant difference plot of chlorophyll $\langle \text{MU} \rangle_a \langle \text{D} \rangle$ in the Glenmore Reservoir sampling network May 1985 to May 1986.

increasing frequency of blooms and associated taste and odour problems are symptomatic of reservoir eutrophication.

STORM WATER IMPACTS ON GLENMORE RESERVOIR

The storm sewers carry dissolved and suspended matter washed off urban areas into Glenmore Reservoir, mainly during large rain storms or periods of snow melt. These materials include heavy metals such as lead, microorganisms, synthetic organic chemicals (gasoline residues, pesticides, herbicides), dust, dirt, oil, grease, nutrients and many other substances (Marshall, Macklin and Monaghan Ltd., 1983). During rainstorms or periods of snow melt, storm sewers create sudden loads of these contaminants which enter the reservoir as plumes of contaminated water. These plumes mix with the comparatively clean water in the reservoir and are gradually diluted. Presently, concentrations of most storm water constituents have no measurable effect on water quality by the time water reaches Glenmore Water Treatment Plant intakes.

Table 7 summarizes mean chloride concentrations from Glenmore Water Treatment Plant intakes and the Elbow River site at Weaselhead from 1982 to 1986. As in many northern metropolitan areas, sodium chloride is applied to roads in Calgary in large amounts during winter months for ice removal. These ions may be important indicators for tracking urban pollutants. Storm events, chinook thaws and spring snow melts wash salts into Glenmore Reservoir through storm sewer outfalls. It would be expected that the Weaselhead location on the Elbow River would not be affected by urban runoff as much as Glenmore Reservoir. Chloride concentrations at both sites vary seasonally and minimum chloride concentrations occur in June (Fig. 8). The minima are the result of dilution by snow melt from the Rocky Mountains. In February and March, urban snow melt and road runoff peak. At this time, the concentrations of chloride at Glenmore Water Treatment Plant intakes reach an annual maximum almost twice that observed in the Elbow River at Weaselhead. Chloride levels at the plant intakes continue to be significantly greater than at Weaselhead until June. During the summer months, following dilution by Elbow River runoff, treatment plant intakes and Weaselhead chloride values are no

Table 7. Chloride (mg L^{-1}) in water samples from the Elbow River at Weaselhead and GWTP intakes (January 1982 to August 1986).

Sample			Chloride			
Year	Site	Number	Mean	Std. Dev. ¹	Maximum	Minimum
1982	Weaselhead	30	2.11	1.77	9.29	0.76
	GWTP Intakes	53	2.30	1.70	8.29	0.97
1983	Weaselhead	32	1.52	0.81	4.68	0.76
	GWTP Intakes	43	2.01	1.26	5.59	0.91
1984	Weaselhead	36	1.27	0.28	1.95	0.86
	GWTP Intakes	48	2.04	0.78	3.63	0.77
1985	Weaselhead	29	1.42	0.48	2.77	0.84
	GWTP Intakes	45	2.07	0.89	5.20	1.17
1986	Weaselhead	25	1.85	0.67	3.03	0.88
	GWTP Intakes	31	2.68	1.38	5.74	1.05
Overall	Weaselhead	152	1.61	0.98	9.29	0.76
	GWTP Intakes	220	2.19	1.26	8.29	0.77

¹Std. Dev. = Standard Deviation

longer significantly different. Not until November, when early winter road salt applications often begin, do Glenmore Water Treatment Plant intake chloride levels once again exceed those at Weaselhead. Similar results (Hargesheimer and Lewis, 1988) have been observed for sodium as well.

The synthetic organic compounds listed in Table 8 are contaminants of concern because of their widespread occurrence and significance even at trace levels (CCREM, 1987). Regular monitoring for these organic compounds, and many others, has been ongoing at Calgary's Glenmore Waterworks Laboratory since 1982. Water samples from treatment plant intakes generally contain below detection limits of the phenols, base-neutral extractable compounds and volatile organic compounds (VOCs) listed in Table 8.

¹ Data from January 1982 to August 1986.

Figure 8. Comparison of mean monthly chloride (mg L^{-1}) at Weaselhead (WH), Glenmore Water Treatment Plant intakes (GR) and Bearspaw Water Treatment Plant intakes (BR).

Table 8. Organic compounds quantitated in Glenmore Reservoir, Elbow River and storm sewer samples (May 1985 to May 1986).

GROUP I - PHENOLS		
Phenol	2,4-Dichlorophenol	
2-Chlorophenol	2-Nitrophenol	
2,4-Dimethylphenol	2,4,5-Trichlorophenol	
4-Chloro-3-Methylphenol	Pentachlorophenol	
4-Nitrophenol		
GROUP 2- BASE NEUTRAL COMPOUNDS		
(A) SYNTHETIC RESIDUES (POLYAROMATIC HYDROCARBONS, CHLORINATED ORGANICS, PLASTICIZERS, INDUSTRIAL RESIDUES)		
1,4-Dichlorobenzene	Anthracene	
Bis(2-chloroisopropyl)ether	Fluoranthene	
Hexachloroethane	Butyl Benzyl Phthalate	
Nitrobenzene	Chrysene	
Dimethyl Phthalate	Ethyl Hexyl Phthalate	
Acenaphthalene	Benzo (b) Fluoranthene	
Fluorene	Benzo (a) Pyrene	
4-Chlorophenyl phenyl ether	Dibenzo (a,h) Anthracene	
4-Bromophenyl phenyl ether	Benzo (g,h,i) Perylene	
(B) ALKANE HYDROCARBONS		
Dodecane	Heptadecane	Tricosane
Tetradecane	Eicosane	Hexacosane
GROUP 3 - NONCHLORINATED VOLATILE ORGANIC COMPOUNDS (VOCs)		
Benzene	Ethylbenzene	<MU>m<D>-Xylene
Toluene	<MU>o<D>-Xylene	<MU>p<D>-Xylene

VOCs such as benzene, toluene, ethylbenzene, and *o*-, *m*- and *p*-xylene are all associated with gasoline, oil and other fuels (US EPA, 1980; Verschueren, 1983). VOCs, because of their volatility, would not be expected to persist in surface water, particularly during the warmer spring and summer months. Concentrations of these compounds in turbulent, well-mixed, shallow waters are generally low (US EPA, 1979a, 1979b; Environment Canada, 1984). Gasoline residues, primarily benzene and toluene, were detected in storm sewer effluents. The gas chromatograph trace "A" (Fig. 9) compares the concentrations of benzene, toluene, ethylbenzene and xylenes recovered from Glenmore Water Treatment Plant intakes, Bearspaw Water Treatment Plant intakes, Elbow River at Weaselhead and storm water. Concentrations in storm water are low at concentrations ranging from 0.5 to 10 $\mu\text{g L}^{-1}$. The two treatment plant intakes samples contain traces of benzene, toluene and the xylenes, while the Elbow River at Weaselhead contains no detectable volatile organic compounds.

Table 9 compares concentrations of organic compounds in storm water and water in Glenmore and Bearspaw Reservoirs. Phenols are present in storm and reservoir water samples at comparable concentrations.

Table 9. Comparison of concentrations of organic compounds ($\mu\text{g L}^{-1}$) detected in storm water and Glenmore Reservoir.

Compound type	Urban Runoff	Glenmore Reservoir
Phenols	<0.1 - 15	<0.1 - 22
Polyaromatic Hydrocarbons	0.1 - 15	<0.1
Alkanes	25 - 50	6 - 16
Volatile Organic Compounds	0.5 - 10	<0.05 - 0.2

Concentrations of the other four groups of organic compounds, however, are considerably greater in storm water than in the reservoirs. Polyaromatic hydrocarbons detected in storm water effluents include pyrene, fluoranthene, anthracene, phenanthrene and naphthalene (Fig. 10).

Figure 9. Gas chromatograms of VOCs detected on the same sampling day (June 19, 1985) in water from: (A) Storm sewer G13; (B) GWTP intakes; (C) Weaselhead; and (D) BWTP intakes. Concentrations ($\mu\text{g L}^{-1}$) above the detection limit of $0.05 \mu\text{g L}^{-1}$ indicated in parentheses.

Figure 10. Polycyclic aromatic hydrocarbons quantitatively identified in storm water (September 12, 1985) by selected ion monitoring mass spectrometry.

For all chemical, physical and microbiological parameters, with the exception of sodium, chloride and a number of synthetic chemicals, storm water loadings are reduced to concentrations below detection as soon as the storm water enters Glenmore Reservoir. Presently, the reservoir's capacity for purification is still able to assimilate and attenuate loadings from urban runoff. Of the many metals, ions, nutrients and organic compounds monitored in storm water, a detectable impact of urban runoff could be identified only for concentrations of chloride, sodium, gasoline residues, polyaromatic hydrocarbons and aliphatic hydrocarbon residues.

CONCLUSIONS

The Glenmore Reservoir is not homogeneous and water quality at the Glenmore Water Treatment Plant intakes is not indicative of the reservoir's overall condition. Often, water quality at the Glenmore Water Treatment Plant intakes is much better than that in the main body of the reservoir. The reservoir has a great assimilative capacity. Heavy rain storms, for example, cause high turbidity and increase bacterial densities in the river and in the body of the reservoir. At the source water intakes, these effects are considerably attenuated. Source water quality at Glenmore Water Treatment Plant is still excellent and shows few measurable signs of urban impacts. Most of the pollutant loadings are present in the reservoir at $\mu\text{g L}^{-1}$ levels and can only be detected by monitoring programs close to the point sources.

In both chemical and microbial water quality, Glenmore Reservoir is beginning to reflect its urban environment. Specific conductance, alkalinity, and hardness in Glenmore Reservoir are controlled largely by Elbow River flow events. Turbidity and fecal indicator bacteria populations in the Elbow River and in the Glenmore Reservoir also fluctuate primarily in response to precipitation and Elbow River flow. As water travels through Glenmore Reservoir from Weaselhead to Glenmore Water Treatment Plant intakes, turbidity is reduced through settling and die-off. These natural "self-purification" processes reduce turbidity and bacterial densities at Glenmore Water Treatment Plant to lower levels than those originally present in the Elbow River or the body of the Glenmore Reservoir. The capacity for purification presently still exceeds loading. Input into Glenmore Reservoir from the Elbow River has a more pronounced impact on most chemical and microbiological water quality parameters than input from storm sewers.

REFERENCES

- Canadian Council of Resource and Environment Ministers. 1987. Canadian Water Quality Guidelines. Prepared by the Task Force on Water Quality Guidelines.
- Environment Canada. 1984. Benzene. Environmental and technical information for problem spills (EnviroTIPS). Technical Services Branch, Environmental Protection Programs Directorate, Environmental Protection Service, Ottawa, ON. 107 pp.
- Hargesheimer, E.E., and C.M. Lewis. 1985. 1984 Glenmore Reservoir Summer Water Quality Study. Internal Report prepared for the City of Calgary, Engineering Department, Waterworks Division, Calgary, AB. 113 pp.
- Hargesheimer, E.E., and C.M. Lewis. 1988. Water Quality in the Glenmore and Bearspaw Reservoirs. Internal Report prepared for The City of Calgary, Engineering Department, Waterworks Division, Calgary, AB. 221 pp.
- Marshall, Macklin and Monaghan Ltd. 1983. Elbow River Watershed Project Report on Water Quality Program, Dye Dispersion Study and Statistical Analysis. Prepared by Envirocon Ltd., Calgary, AB.
- Prepas, E.E. 1983. The Influence of Phosphorus and Zooplankton on Chlorophyll Levels in Alberta Lakes. Prepared for Alberta Environment, Research Management Division by Department of Zoology, University of Alberta, Edmonton, AB. RMD Report 83/23. 111 pp.
- US EPA. 1979a. Benzene. In: Water-related Environmental Fate of 129 Priority Pollutants. Vol.II. Halogenated aliphatic hydrocarbons, halogenated ethers, monocyclic aromatics, phthalate esters, polycyclic aromatic hydrocarbons, nitrosamines, miscellaneous compounds. pp. 71-1 to 71-9. Office of Water Planning and Standards, United States Environmental Protection Agency, Washington, DC. EPA-440/4-79-029b.
- US EPA. 1979b. Toluene. In: Water-related Environmental Fate of 129 Priority Pollutants. Vol.II. Halogenated aliphatic hydrocarbons, halogenated ethers, monocyclic aromatics, phthalate esters, polycyclic aromatic hydrocarbons, nitrosamines, miscellaneous compounds. pp. 71-1 to 71-9. Office of Water Planning and Standards, United States Environmental Protection Agency, Washington, DC. EPA-440/4-79-029b.
- US EPA. 1980. Ambient Water Quality Criteria for Benzene. Office of Water Regulations and Standards, Criteria and Standards Division, United States Environmental Protection Agency, Washington, DC. EPA 440/5-80-018.
- Verschueren, K. 1983. Handbook of Environmental Data on Organic Chemicals. 2nd Ed. Van Nostrand Reinhold Co., New York, NY. 1310 pp.

Drinking Water Quality Guidelines in Alberta (Development and Application)

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ABSTRACT

Alberta Environment sets standards and issues approvals for municipal water supply systems in the Province of Alberta. Health-related standards, as established by Health and Welfare Canada in the Guidelines for Canadian Drinking Water Quality, have been adopted and applied as the drinking water quality requirements for municipal water supplies in Alberta. This paper briefly outlines the administrative and technical approach followed by Health and Welfare Canada in developing the Guidelines for Canadian Drinking Water Quality. The application of the Guidelines in Alberta is also outlined. For general interest an Appendix is included which compares previous and current drinking water quality limits as established by Health and Welfare Canada.

INTRODUCTION

The availability and quality of drinking water can have a significant impact on both the public health and the overall quality of life within a community. A major objective of Alberta Environment is to ensure that drinking water supplies and treatment systems provide a high level of public health protection while being able to meet the water supply needs of the community.

To achieve this objective the following factors must be considered or controlled when planning, designing and operating a drinking water supply system:

- (1) the quality of drinking water that must be produced;
- (2) the quantity and quality variations of the raw water supply;
- (3) the ability of the raw water supply to meet the immediate and long term quantity needs of the community;

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- (4) the level and types of water treatment required to ensure good quality drinking water regardless of variations in raw water quality;
- (5) the level of operator skill required to operate the drinking water treatment facility; and
- (6) the type and amount of operational and performance monitoring required.

Alberta Environment endeavours to ensure that each of these factors is addressed when municipal or communal water supplies are developed and operated in Alberta. For example, the Alberta Water Resources Act requires that an approval be obtained to divert or use water for municipal purposes; the adequacy of the proposed water supply in terms of quantity is assessed as part of this approval. The Alberta Clean Water Act and associated Regulations establish a Permit to Construct and Licence to Operate approval system as well as quality, monitoring and operating requirements for municipal and communal water supplies. In addition, Alberta Environment also publishes standards and guidelines that apply to these systems.

The requirements applied to drinking water supplies in Alberta are generally developed by Alberta Environment and are based on accepted good practice in the water supply field in North America. The overall intent is to establish requirements that reflect the current state-of-knowledge and to apply these requirements to provincial conditions or needs. The only exception to this general approach relates to drinking water quality requirements which are entirely based on the Health and Welfare Canada's Guidelines for Canadian Drinking Water Quality.

The purpose of this paper is to outline the approach followed in developing the Guidelines for Canadian Drinking Water Quality and to explain how these Guidelines are applied in Alberta.

HISTORY

The first drinking water standards in North America were established in 1914 in the United States. These standards were developed under the Public Health Service Act and covered only the microbiological quality of water supplies on interstate carriers (Oleckno, 1982; EPA, 1986; Kawata, 1986). In Canada, drinking water quality standards were first promulgated by the Federal Government in 1923, establishing bacteriological standards for drinking waters on ships on the Great Lakes and Inland Waters (Health and Welfare Canada, 1968).

The United States standards were expanded a number of times during the period 1914 to 1974. They technically remained legally applicable only to interstate carriers, although they were widely adopted and used at the state level (EPA, 1986; Kawata, 1986). The 1923 Canadian guidelines were extended during the 1930s to include all common carriers and coastal shipping, and the Department of Health and Welfare subsequently adopted the 1943 United States Public Health Service Drinking Water Standards pending development of Canadian standards (Health and Welfare Canada, 1968).

In the mid-1960s, a Joint Committee on Drinking Water Standards was formed. The objective of the Committee, which consisted of federal, provincial and university health and environmental professionals, was to "develop and promote national criteria, standards and objectives for drinking water". Its work resulted in the document entitled: "Canadian Drinking Water Standards and Objectives 1968" (Health and Welfare Canada, 1968).

In the early 1970s the Joint Committee was replaced by a Federal-Provincial Working Group, consisting of Federal and Provincial representatives, which developed the document entitled: "Guidelines for Canadian Drinking Water Quality 1978" (Health and Welfare Canada, 1978).

In the early 1980s, a new Federal-Provincial Working Group was formed to evaluate the need to update the 1978 guidelines. Based on the recommendation of this Working Group, a standing Federal-Provincial Sub-Committee on Drinking Water was formed in 1986 to review and revise the guidelines on an on-going basis. The Federal Departments of Health and Welfare Canada and Environment Canada along with each of the Territories and Provinces have representatives on the Sub-Committee which recently completed a major review and updating of the 1978 guidelines. These revisions were published in the document entitled: "Guidelines for Canadian Drinking Water Quality" (Health and Welfare Canada, 1987).

It is the intent of the Federal-Provincial Sub-Committee on Drinking Water to update and republish these guidelines every one or two years.

PURPOSE AND TYPES OF DRINKING WATER QUALITY STANDARDS

The Canadian Drinking Water Standards and Objectives 1968 stated that (Health and Welfare Canada, 1968):

"Water for drinking, culinary, or other domestic uses should be safe, palatable, and aesthetically appealing. It should be free from pathogenic organisms, deleterious chemical and radioactive substances and objectionable colour, odour and taste. Other considerations, such as corrosiveness, tendency to form encrustations, and excessive soap consumption due to hardness should also be regarded as important in evaluating the quality of drinking water".

These drinking water characteristics and quality goals are restated in the 1978 and 1987 Guidelines for Canadian Drinking Water Quality (Health and Welfare Canada, 1978; Health and Welfare Canada, 1987) and can be considered as the objective of the guidelines.

To achieve this objective, drinking water quality limits for the following general categories of substances found in drinking water are established in the Guidelines:

- physical
- chemical (inorganic, organic and pesticides)
- microbiological
- radiological

The types of drinking water quality limits used in the 1987 Guidelines are (Health and Welfare Canada, 1987):

Maximum Acceptable Concentration (MAC)

Maximum Acceptable Concentrations are established for substances that are known or suspected to cause adverse effects on health. They are intended to safeguard health on the basis of lifelong consumption and the use of the water for all usual domestic purposes, including personal hygiene. Higher quality water may be required for some special purposes, including renal dialysis.

Interim Maximum Acceptable Concentrations (MAC)

In those instances where insufficient toxicological data exist to derive a maximum acceptable concentration with reasonable certainty, interim values are recommended taking into account the available health-related data, but employing additional safety factors to compensate for the uncertainties involved. Because of the nature of interim maximum acceptable concentrations, they must continually be reviewed.

Aesthetic Objectives (AO)

Aesthetic Objectives apply to certain substances or characteristics of drinking water which can affect its acceptance by consumers or interfere with good water supply practices. For certain parameters for which aesthetic objectives are necessary, health-related guidelines (maximum acceptable concentrations) are also derived.

Where only aesthetic objectives are specified, the values are below those considered to constitute a health hazard. The 1987 Guidelines specifically state that limits should not be regarded as implying the quality of drinking water may be degraded to these objectives, and every effort should be made to ensure drinking water is of the best possible quality.

A comparison of the parameters covered and drinking water limits in the 1968 Standards and Objectives, 1978 Guidelines and 1987 Guidelines is presented in Table I in the Appendix. A detailed discussion on the rationale for the different types of limits specified in the 1968 and 1978 documents when compared to the 1987 document is beyond the scope of this paper. The general rationale for the current approach is that it focuses on health and aesthetic water quality issues which can be addressed scientifically while avoiding the subjective issue of what is desirable or ideal drinking water quality.

DEVELOPMENT OF SPECIFIC DRINKING WATER QUALITY STANDARDS

The approach followed in developing maximum acceptable concentration and aesthetic objectives in the Canadian Drinking Water Quality Guidelines has been summarized by Toft (1985) and will be outlined in detail in a Supporting Document to the 1987 Guidelines for Canadian Drinking Water Quality which is scheduled for publication in the fall of 1989.

Selecting Parameters

The first step in developing drinking water quality standards is to identify those substances for which limits in drinking water are necessary or desirable. In this regard, the Federal-Provincial Sub-Committee on Drinking Water is continuously updating a list of parameters that it considers should be reviewed for possible inclusion or revision in future guidelines.

Factors considered when selecting parameters for review include:

- potential sources of the parameter
- the frequency of detection in drinking water

- the range of concentrations found in drinking water
- the infectivity, toxicity and/or carcinogenicity of the substance/agent
- aesthetic properties i.e. taste, odour, and colour
- the relative significance of drinking water exposure compared to other exposure routes e.g. food, air and
- other properties that might have adverse or undesirable impacts e.g. corrosive, scale forming, etc.

In general, parameters that are of concern or interest in only one or two locales are not considered a high priority for review by the Sub-Committee; these are addressed by Health and Welfare Canada through health advisories.

An example of the procedures followed in selecting parameters for inclusion in the 1987 Guidelines is provided by the assessment conducted on pesticides. The initial step involved each Sub-Committee member providing information on pesticide use in their jurisdiction and any information on pesticide levels in drinking water or raw water in that member's jurisdiction. Each Sub-Committee member was also requested to identify pesticides which should be considered for inclusion in the Guidelines. A ranking system was then developed by Health and Welfare Canada which considered the following:

- the number of provinces/territories requesting a review of the pesticide
- the quantity used in Canada
- the type of use
- the number of positive detections in Canadian waters
- the maximum levels detected in Canadian waters
- solubility in water
- persistence and leaching potential
- acute toxicity
- chronic toxicity.

Using this ranking system and input from Agriculture Canada, 42 pesticides were identified for inclusion in the Guidelines, and 17 pesticides for which Health Advisories should be developed were also identified.

Detailed Assessment of Selected Parameters

Once a parameter is identified for review and possible subsequent inclusion in the guidelines, a comprehensive scientific assessment is conducted. This assessment covers:

- the physical and chemical properties of the substance
- the occurrence of the substance in air, water and food

- the relative significance of different exposure routes (for drinking water, 1.5 litres is used as the per capita daily consumption)
- the pharmacokinetics of the substance
- toxicological studies including acute and chronic toxicity, teratogenic and reproductive effects, mutagenicity, chronic toxicity and carcinogenicity
- assessment of health risk
- practical quantification limit
- available drinking water removal technologies and capabilities

Based on the assessment, a maximum acceptable concentration is derived using one of following approaches depending on whether the substance is considered carcinogenic, possibly carcinogenic, or not carcinogenic.

Limits for Non-Carcinogens

For substances which are not considered carcinogenic, maximum acceptable concentrations are derived based on a calculated acceptable daily intake (ADI) for the most sensitive health response/impact. The ADI is calculated by dividing the no-observable or lowest observable adverse effect levels (NOAEL or LOAEL) for the health response being considered by uncertainty factors which consider:

- the duration of the NOAEL or LOAEL tests
- intraspecies variations
- interspecies variations
- nature and severity of the effect
- adequacy of the study(ies)
- LOAEL vs NOAEL
- potential interactions with other chemicals

A portion of this ADI is then allocated to drinking water based on actual multi-media exposure data, or in the absence of good exposure data, 20% of the ADI is generally allocated to drinking water. Using a 70 kg adult and a drinking water consumption of 1.5 L, this drinking water ADI is converted to a maximum acceptable concentration.

Limits for Carcinogens

For substances which are considered carcinogenic it is assumed that there is some probability of risk at any level of exposure. By extrapolating dose-response data or using occupational exposure or epidemiological data, an estimate of the risk associated with exposure

to the substance is obtained. The maximum acceptable concentration (MAC) for a carcinogenic substance is then established using the following criteria:

- the upper 95% confidence limit for the lifetime risk associated with the MAC should be less than 10^{-5} to 10^{-6} which is considered to represent a negligible risk (in cases where exposure to the substance via other sources is significant then the MAC is based on a risk level of 10^{-6} or less)
- the MAC must be technically and economically achievable from a water treatment standpoint
- the MAC must be measurable using available analytical methods

If the derived MAC presents a risk of greater than 10^{-5} to 10^{-6} then an "interim MAC" is established until either better treatment or analytical methods are available to allow establishment of a MAC with an associated risk of less 10^{-5} to 10^{-6} .

Limits for Possible Carcinogens

For chemicals which are possibly carcinogenic, the MAC is determined by calculating an acceptable daily intake as is done for non-carcinogenic substances. An additional safety factor of 1 to 10 times is then used to account for the possible carcinogenicity of the substance.

Limits for Aesthetic Parameters

For substances that input taste and/or odour to water, an aesthetic objective is established based on reported aqueous taste and odour thresholds. For substances that have potential health as well as aesthetic impacts a MAC is also established. For compounds such as toluene, ethylbenzene and xylene, however, the taste and odour threshold appears to be several orders of magnitude less than the concentrations having any toxicological significance.

Limits for Radionuclides

The MACs for radionuclides are based on dose-response relationships and Annual Limits of Intake (ALI) recommended by the International Commission on Radiological Protection. A MAC for radionuclides commonly detected is then derived by taking 1% of the ALI for continuous occupational exposure and assuming 2 L of drinking water is consumed (Health and Welfare Canada, 1987). Some modifications to this approach are currently being considered

based on recent recommendations of both the Sub-Committee and the International Commission on Radiological Protection.

Limits for Microbes

Microbiological quality has been, and continues to be, the most important public health issue associated with drinking water. It is not practical, however, to try and identify and quantify all the potentially pathogenic organisms in drinking water. Indicator organisms are therefore used to assess the microbiological quality of drinking water. The purpose of indicator organisms is to indicate the probable presence or absence of disease producing organisms. Total and fecal coliforms and standard plate count organisms are the indicators used in the Guidelines to assess the microbiological quality of drinking water. The recommended frequency of bacteriological sampling to determine compliance with the microbiological MAC is also given in the Guidelines (Health and Welfare Canada, 1987).

APPLICATION OF THE GUIDELINES

The Guidelines for Canadian Drinking Water Quality are not legally enforceable standards unless adopted as such by the appropriate federal, provincial or territorial agency (Health and Welfare Canada, 1987). To date, only Quebec and Alberta have formally adopted the Guidelines through regulation; however, the Guidelines are used by all jurisdictions in Canada as a guide for assessing the quality of drinking water supplies.

Legislated Requirements in Alberta

The Alberta Clean Water Act gives the Lieutenant Governor in Council the authority to regulate the purity of water to be used for human consumption. In this regard, Section 23 of the Clean Water (Municipal Plants) Regulations states that:

"The physical, microbiological, chemical and radiological characteristics of the water in a municipal water supply shall be maintained to meet the health-related concentration limits as contained in the Department of National Health and Welfare (Canada) publication entitled "Guidelines for Canadian Drinking Water Quality 1978."

An amendment to reflect the recent release of the 1987 Guidelines is currently under review. The Clean Water (Municipal Plants) Regulations therefore clearly establish the Guidelines for Canadian Drinking Water Quality as the quality standards for municipal water supplies in Alberta. Non-compliance with this section of these Regulations is not, however, an offence *per se*.

The actual application and enforcement of the Guidelines for Canadian Drinking Water Quality in Alberta is through a Licence to Operate procedure which is established in the Clean Water Act. Under this procedure, Licences to Operate municipal waterworks systems are issued with terms, conditions or requirements which must not be less stringent than those imposed by regulation. A standard condition in all Licences to Operate for municipal waterworks systems is that:

"The water treatment facility shall be maintained and operated to produce treated water which meets the maximum concentration limits as outlined in the latest edition of the Guidelines for Canadian Drinking Water Quality."

Non-compliance with a Licence condition is an offence which is subject to a fine not exceeding \$25,000 and in default of payment to a term of imprisonment not exceeding 3 months.

The "in practice" definition used for a municipal waterworks system to which these requirements apply is:

"Any communal municipal water system providing potable water to either a designated municipality (Hamlet, Metis Settlement, Village, Summer Village, Town, City), or any rural municipal development of two lots or more in which subdivision of the parcel has occurred for the purpose of establishing separate ownership of each lot" (Alberta Environment, 1988).

Non-municipal waterworks systems in Alberta are covered under Public Health Act legislation and fall within the jurisdiction of the Local Health Units.

The application of the Guidelines for Canadian Drinking Water Quality requires that monitoring frequencies be established to determine whether or not compliance is being achieved.

DRINKING WATER QUALITY MONITORING REQUIREMENTS IN ALBERTA

The 1987 Guidelines for Canadian Drinking Water Quality recommend certain sampling frequencies for the various types of parameters covered, i.e. microbiological, chemical and

physical, and radiological, but clearly indicate that the frequency of sampling should depend on (Health and Welfare Canada, 1987):

- the quality of the source water
- the past frequency of unsatisfactory samples
- the type of treatment system
- the presence of specific sources of certain contaminants
- the possibility of seasonal variations in water quality

The minimum frequency of sampling for water supplies in Alberta is specified in the Clean Water (Municipal Plants) Regulations and is as follows:

"The number of water samples to be submitted to the department laboratory for analysis of chemical quality shall be at least

- (a) if the water is obtained from one or more wells, one sample each year from each well connected to the waterworks system, or
- (b) if the water is obtained from a surface water source, two samples of treated water each year, one of which shall be obtained during the winter months and the other obtained during the summer months.

The minimum number of water samples to be submitted to the approved laboratory for analysis for bacteriological quality shall be evenly distributed through the sampling period and shall be in accordance with the latest edition of Canadian Drinking Water Standards and Objectives as published by the Government of Canada."

The frequency of sampling is generally consistent with the frequency recommended in the 1987 Guidelines; however, municipalities are not required to monitor for pesticides, organics and radionuclides.

More extensive monitoring programs associated with analyzing for all the parameters listed in the 1987 Guidelines and other parameters is undertaken by Alberta Environment. For example, non-routine monitoring surveys focusing on parameters such as asbestos and radionuclides have been conducted to determine whether or not additional or on-going monitoring is warranted. A routine Treated Water Survey monitoring program conducted by Alberta Environment involves sampling water from approximately 50 communities at frequencies varying

from once a year to once a month and measuring over 200 chemical parameters. As the number of parameters covered by the Guidelines for Canadian Drinking Water Quality increases; however, it will become more and more difficult to monitor for all regulated parameters on a regular basis, and more directed and site-specific monitoring strategies will have to be developed.

SUMMARY

The Guidelines for Canadian Drinking Water Quality are developed by a Federal-Provincial Sub-Committee. Parameters are selected for inclusion based on frequency of occurrence in water supplies and health or aesthetic significance at the levels detected. The approach used to derive specific limits for a parameter will vary depending on either the possible effect of the parameter (i.e. is it a carcinogen, a chronic toxicant or an aesthetic parameter?) or the general nature of the parameter (i.e. microbiological, chemical, physical or radiological).

The limits specified in the Guidelines for Canadian Drinking Water Quality have been adopted through regulation and facility licensing requirements as the drinking water quality limits applying to municipal and communal water supplies in Alberta. The monitoring requirements established by Alberta Environment to determine compliance with the Guideline limits are similar to those recommended in the Guidelines with some notable exceptions e.g. radionuclides and organics.

REFERENCES

- Alberta Environment. 1988. Standards and Guidelines for Municipal Water Supply, Wastewater, and Storm Drainage Facilities. Alberta Department of Environment, Edmonton, AB.
- Environmental Protection Agency (EPA). 1986. Drinking Water Milestones. EPA J. 12:20.
- Health and Welfare Canada. 1968. Canadian Drinking Water Standards and Objectives 1968. Queen's Printers, Ottawa, ON.
- Health and Welfare Canada. 1978. Guidelines for Canadian Drinking Water Quality 1978. Supply and Services Canada, Ottawa, ON.
- Health and Welfare Canada. 1987. Guidelines for Canadian Drinking Water Quality. Supply and Services Canada, Ottawa, ON.
- Kawata, K. 1986. Evolution of Drinking Water Regulations in the United States. J. Environ. Health 48:206-209.

- Oleckno, W.A. 1982. The National Interim Primary Drinking Water Regulations. Part I - Historical Development. J. Environ. Health 44:236-239.
- Toft, P. 1985. The Control of Organics in Drinking Water in Canada and the United States (Standards, Legislation and Practice). Sci. Total Environ. 47:45-58.

1968 Canadian Drinking Water Standards and Objectives		1978 Guidelines for Canadian Drinking Water Quality		1987 Guidelines for Canadian Drinking Water Quality		
Parameter	Objective (mg/L)	Acceptable Limit (mg L ⁻¹)	Maximum Permissible Limit (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Objective Concentration (mg L ⁻¹)	Aesthetic Objective (mg L ⁻¹)
MBAS	<0.2	0.5				-
Nitrate (as N)				10.0	≤0.001	10.0 (g)
Nitrite (as N)				1.0	≤0.001	1.0
Odour (TON)	0	4	-	-	Inoffensive	Inoffensive
pH (units)	-	6.5-8.3	-	6.5-8.5	-	6.5-8.3
Phenolic Substances as Phenol	ND	0.02		0.002	<0.002	
Phosphates (as PO ₄)	<0.2	0.2				
Selenium	ND	<0.01	0.01	0.01	≤0.002	-
Silver	-	-	0.05	0.05	≤0.005	-
Sodium						(h)
Sulphate (as SO ₄ ⁻²)	<250	500		500	<150	<150
Sulphide (as H ₂ S)	ND	0.3		0.05	<0.05	<0.05
Taste	Inoffensive	Inoffensive		-	Inoffensive	Inoffensive
Temperature (°C)	<10	15		15	<15	<15
Total Dissolved Solids	<500	1000				<500
Turbidity	<1 (JTU)	5 (JTU)		5 (NTU)	<1 (NTU)	<5 (NTU)
Uranium				0.02	≤0.001	0.1
Uranyl Ion (as HO ₂ ⁻)	<1.0	5.0				-
Zinc	<1.0	5.0		5.0	<5.0	<5.0
ORGANICS						
1. Pesticides						
Arochlor						-
Aldicarb						0.009

ND = Not Detectable; JTU = Joday Turbidity Units; NTU = Naphelometric Turbidity Units; IMAC = Interim Maximum Acceptable Concentration

ND = Not Detectable; JTU = Jnday Turbidity Units; NTU = Nephelometric Turbidity Units; IMAC = Interim Maximum Acceptable Concentration

		1968 Canadian Drinking Water Standards and Objectives		1978 Guidelines for Canadian Drinking Water Quality		1987 Guidelines for Canadian Drinking Water Quality	
Parameter	Objective (mg/L)	Acceptable Limit (mg L ⁻¹)	Maximum Permissible Limit (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Objective Concentration (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Aesthetic Objective (mg L ⁻¹)
Aldrin & Dieldrin	ND	ND	0.017	0.0007	<5x10 ⁻⁸	0.0007	-
Atrazine						-(IMAC 0.06)	-
Azinphos-methyl						0.02	-
Bendiocarb						0.04	-
Bromoxynil						-(IMAC 0.005)	-
Carbaryl				0.07	<5x10 ⁻⁸	0.09	-
Carbofuran						0.09	-
Chlordane (total isomers)	ND	ND	0.003	0.007	<5x10 ⁻⁸	0.007	-
Chlorpyrifos						0.09	-
Cyanazine						-(IMAC 0.01)	-
2,4-D	ND	ND	0.100	0.1	<1x10 ⁻³	0.1	-
DDT + metabolites	ND	ND	0.042	0.03	5x10 ⁻⁸	0.03	-
Diazinon				0.014	<1x10 ⁻⁶	0.02	-
Dicamba						0.12	-
Diclofop-methyl						0.009	-
Dieldrin & Aldrin	ND	ND	0.017			0.0007	-
Dimethoate						-(IMAC 0.02)	-
Dinoseb						-	-
Diquat						0.07	-
Diuron						0.15	-
Endrin	ND	ND	0.001	0.0002	<5x10 ⁻⁸	-	-
Glyphosate						-(IMAC 0.28)	-
Heptachlor + Heptachlor epoxide	ND	ND	0.018	0.003	5x10 ⁻⁸	0.003	-
Lindane	ND	ND	0.056	0.004	1x10 ⁻⁷	0.004	-

ND = Not Detectable; JTU = Jeday Turbidity Units; NTU = Naphrelomatic Turbidity Units; IMAC = Interim Maximum Acceptable Concentration

ND = Not Detectable; JTU = Junday Turbidity Units; NTU = Nephelometric Turbidity Units; IMAC = Interim Maximum Acceptable Concentration

Parameter	Objective (mg/L)	1968 Canadian Drinking Water Standards and Objectives		1978 Guidelines for Canadian Drinking Water Quality		1987 Guidelines for Canadian Drinking Water Quality	
		Acceptable Limit (mg L ⁻¹)	Maximum Permissible Limit (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Objective Concentration (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Aesthetic Objective (mg L ⁻¹)
Linuron						-	-
Malathion						0.19	-
MCPA						-	-
Methoxychlor	ND	ND	0.035	0.1	<5x10 ⁻⁸	0.9	-
Methyl-parathion				0.007	<1x10 ⁻⁶	0.007	-
Methalochlor						-(IMAC 0.05)	-
Metribuzin						0.08	-
Organic Phosphates + Carbamates	ND	ND	0.100			-(IMAC 0.01)	-
Paraquat						0.05	-
Parathion				0.035	<1x10 ⁻⁶	0.1	-
Pesticides (total)				0.1		-(IMAC 0.002)	-
Phorate						-	-
Picloram						-(IMAC 0.01)	-
Simazine						0.28	<0.02
2,4,5-T	ND	ND	0.100			-	-
2,4,5-IP	ND	ND	0.100	0.01	<1x10 ⁻³	-	-
Trichloroacetic acid						-	-
Temephos						-(IMAC 0.28)	-
Terbufos						-(IMAC 0.001)	-
Toraphene						-	-
Triallate	ND	ND	0.005	0.005	<5x10 ⁻⁸	0.23	-
Trifluralin						-	-

ND = Not Detectable; JTU = Judo Turbidity Units; NTU = Nephelometric Turbidity Units; IMAC = Interim Maximum Acceptable Concentration

		1968 Canadian Drinking Water Standards and Objectives		1978 Guidelines for Canadian Drinking Water Quality		1987 Guidelines for Canadian Drinking Water Quality	
Parameter	Objective (mg/L)	Acceptable Limit (mg L ⁻¹)	Maximum Permissible Limit (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Objective Concentration (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Aesthetic Objective (mg L ⁻¹)
2. Others							
Benzene						0.005	-
Benzo(a)pyrene						0.00001	-
Carbon tetrachloride						0.005	-
Chlorobenzene; 1,2 di-						0.2	0.003
Chlorobenzene; 1,4 di-						0.005	0.001
Chlorophenol; 2,4 di-						0.9	0.0003
Chlorophenol; penta-						0.08	0.03
Chlorophenol; 2,3,4,6, tetra-						0.1	0.001
Chlorophenol; 2,4,6 tri-						0.005	0.002
1,2-Dichloroethane						-	-
1,1-Dichloroethylene						-	-
Dichloromethane						0.05	-
Dioxins						-	-
Ethylbenzene						-	<0.0024
Furans						-	-
Gasoline						(d)	-
Nitritriacetic acid (NTA)				0.05	≤0.0002	0.05	-
Organics as CCE + CAE	<0.05	0.2		-	-	-	-
PCBs				0.002	<0.002	-	-
Phenols						-	-
Tetrachloroethylene						-	<0.024
Toluene				500	-	-	<500
Total Dissolved Solids				-	-	-	-

ND = Not Detectable; ITU = Iodine Turbidity Units; NTU = Nephelometric Turbidity Units; IMAC = Interim Maximum Acceptable Concentration

Parameter	Objective (mg/L)	1968 Canadian Drinking Water Standards and Objectives		1978 Guidelines for Canadian Drinking Water Quality		1987 Guidelines for Canadian Drinking Water Quality	
		Acceptable Limit (mg L ⁻¹)	Maximum Permissible Limit (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Objective Concentration (mg L ⁻¹)	Maximum Acceptable Concentration (mg L ⁻¹)	Aesthetic Objective (mg L ⁻¹)
Total Organic Carbon	-	-		-		-	-
1,1,1-Trichloroethane						-	-
Trichloroethylene						-	-
Trihalomethanes				0.35	≤0.0005	0.35	-
Xylenes						-	<0.3

ND = Not Detected; JTU = Jday Turbidity Units; NTU = Nephelometric Turbidity Units, IMAC = Interim Maximum Acceptable Concentration

The Importance of Rivers for Urban Recreation: The Case of the Bow River

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INTRODUCTION

Rivers and other water bodies have played an important part in the urban development of Alberta. If you look at a map of the province, you will find that large cities and many smaller communities are situated adjacent to a river or some other body of water. In recent years, many of these communities have undertaken capital projects to develop the recreational capacity of these water bodies in order to enhance the quality of life for their citizens. Now, such features as walking trails, playgrounds, bike paths and boating facilities can commonly be found adjacent to rivers and lakes within municipal boundaries.

The importance of rivers for recreation, particularly in urban areas, has never been disputed. The fact that people are frequently observed in boats and along the shore has been adequate testimony to their importance. However, as industrial and agricultural demands for water are increasing, water resource managers are being forced to question how changes in the quantity or quality of river water would affect urban recreation. Answering these questions requires very basic information about the amount of recreational activity that presently occurs within river valleys and the relative importance and economic value of these activities. Unfortunately, this information does not currently exist for most water bodies within the province, and so the importance of urban river recreation remains an unknown element in water management decisions.

In 1986, concern over the recreational importance of the Bow River near Calgary led the Province of Alberta to commission a study to answer some of these questions. A steering committee consisting of representatives from Alberta Forestry, Lands and Wildlife, Alberta Environment, Tourism Alberta, Alberta Recreation and Parks, and Trout Unlimited was established and consultants were hired to complete the work. Field investigations occurred over a 12 month period starting on May 17, 1986, and a summary of the final results was made public in June of 1988. In this paper the results of the Bow River Recreation Study are described and a discussion is presented on the importance of the study findings.

The Bow River Recreation Study is notable for five reasons:

- (1) The importance of the Bow River valley for water-based and land-based recreation was demonstrated.
- (2) The magnitude of the guided angling industry, which has developed largely because nutrients from treated sewage discharges have enhanced aquatic productivity, was documented.
- (3) A contingent valuation process was used to establish the economic benefits associated with recreational activities. This procedure established that many recreationalists have a box-shaped demand curve, rather than the typical convex-to-the-origin demand curves which are seen for other commodities.
- (4) The existence and option values associated with the river valley were quantified.
- (5) The river surveys employed a unique data extrapolation procedure which provided accurate counts of river users with relatively low sampling effort.

Below each of these five points are briefly discussed.¹

RECREATIONAL IMPORTANCE OF THE BOW RIVER VALLEY

The fact that the Bow River valley between the Bearspaw Dam northwest of Calgary and the Blackfoot Indian Reserve, some 111 km downstream (Fig. 1), is intensively used for recreation should not come as a surprise to anyone. The results of our 12-month monitoring program indicated that an average of about 305 people per day used the river for water-based recreation. This level of activity ranged from a low of only 30 people per day during the winter months to an average of nearly 1500 people per day in June and July. The study found that fishing accounted for 78% of this water-based activity (boating accounted for the remainder); 91% of recreationalists were residents of the Calgary area. Total water-based recreational activity on the Bow River amounted to 111,320 user-days (Table 1).

¹Should readers want more detailed information about the study, copies of both the summary and technical volumes of the study can be obtained by contacting the Information Centre of Alberta Forestry, Lands and Wildlife.

Table 1. Recreational activity in the Bow River Valley.

	Water-Based Recreation	Guided Anglers	Land-Based Recreation	Total Activity
Amount of Activity: (user-days):	111,320 ± 15,200	2,960 na ¹	3,481,050 ± 294,710	3,595,330 ± 295,100
Amount of Activity According to the Origin of the Recreationist:				
Calgary Area	100,850 ± 13,830	0 ± 0	3,481,050 ± 294,710	3,581,900 ± 295,030
Other Alberta	5,740 ± 1,530	130 ± 25	na ²	5,870 ± 1,530
Other Canada	1,250 ± 660	360 ± 70	na ²	1,610 ± 665
U.S. & Other Countries	3,480 ± 1,160	2,470 ± 480	na ²	5,950 ± 1,225

¹Total activity by guided anglers was reported by angling guides and was not an estimate.

²Only land-based activities by Calgary area residents were addressed during the household survey.

Table 2. Economic values associated with recreation on the Bow River (1987 dollars).

Type of Value	Water-Based Recreation	Guided Anglers	Land-Based Recreation	Total Benefits
Expenditures:				
Per user-day	\$19.97 ± 4.19	\$874.36 ± 105.52	\$0.50 ± 0.14	\$1.83 ± 0.15
Total	\$2,224,500 ± 506,800	\$2,588,100 ± 316,000	\$1,755,800 ± 521,500	\$6,568,400 ± 295,100
Consumer's Surplus:				
Per user-day	\$7.61 ± 0.25	na ¹	\$1.02 ± 0.12	\$1.22 ± 0.10
Total	\$847,100 ± 122,400	na ¹	\$3,533,300 ± 517,900	\$4,380,400 ± 532,200
Existence/Option Values				\$636,400 ± 216,700
TOTAL BENEFITS	\$3,071,600 ± 521,400	\$2,588,100 ± 316,000	\$5,289,100 ± 735,000	\$11,585,200 ± 986,180

¹Consumers' surplus was only measured for Alberta Residents and most guided anglers were visitors to the province.

What was surprising, however, was the amount of land-based recreational activity that occurred in the Bow River valley. For example, the study found that nearly 70% of all Calgary households used the Bow River valley for some form of land-based recreation. Total recreational activity during the 12-month period amounted to 3.48 million user-days or about 23 user-days per household. This level of activity was nearly 30 times greater than the corresponding amount

of water-based recreation. Land-based recreation consisted of walking (51%), bicycling (23%), jogging (18%) plus other activities such as picnicking, skating, and hiking. About 29% of land-based activity occurred along the city's network of bicycle and walking paths, another 28% occurred at Fish Creek Provincial Park, and the remainder was distributed among various city parks in the river valley.

As far as we know, this study marks the first time that the land-based recreational activities associated with a water body have been studied in detail. The study's methodology involved monitoring the monthly outdoor recreation patterns of about 450 Calgary households and the resulting estimates of total recreational activity were determined to be accurate to within 8.5% at a 95% level of confidence.

THE GUIDED ANGLING INDUSTRY

The Bow River immediately downstream of Calgary has gained the reputation of being a world-class trout fishery. Treated effluent from the city's two sewage treatment plants has produced a luxuriant growth of aquatic plants in this reach of the river which supports abundant large rainbow and brown trout which now attract anglers from throughout North America and from overseas. This fishery has resulted in the development of a small but valuable guided angler industry in the Calgary area.

One objective of this study was to establish the size and importance of this industry. Through interviews with most guides and a random selection of their clientele, it was determined that during 1987 1,191 anglers used the services of 10 angling guide companies. Total angler effort was 2,960 angler-days and total expenditures by this group amounted to \$2.70 million. Nearly 95% of these people came from the United States and more than 85% of anglers rated the river as above average in terms of angling quality.

The overall economic benefit of guided angling on the Bow River was determined to be \$1.90 million. This represents the amount of money that anglers added to the provincial economy and excludes spending on transportation, spending by provincial residents who used angling guides, and spending by anglers who visited the Calgary area for reasons other than fishing on the river.

THE ECONOMIC VALUE OF RECREATION

While water- and land-based recreational opportunities in the Bow River valley are available as often as people wish to use them, they have no price in the conventional market sense. These opportunities do, however, have a value which is termed "consumers' surplus". Consumers' surplus refers to the amount that people would be willing to pay for a resource over and above what they actually pay to use that same resource. While consumers' surplus values exist for many commodities, such values tend to be quite large for recreational resources.

In the Bow River Recreation Study, a contingent valuation process was used to measure consumers' surplus. Recreationists were asked to estimate the annual amount of fishing that they would do if they had to pay hypothetical access charges ranging from between \$0.50 and \$15.00 per day, assuming that access to all other recreational resources remained free. Their responses were used to derive an aggregate demand curve which allowed estimation of the average value of a day of recreation.

According to the survey results, the average value of a day of water-based recreation was \$7.61; this estimate was found to be accurate to within 3.3%. This amount ranged from \$6.36 per day for residents of the Calgary area to \$20.05 for visitors from other parts of Alberta. Overall, the value of water-based recreation on the river was determined to be \$0.85 million.

A similar approach was used to measure the consumers' surplus benefits associated with land-based recreation. In this case, however, the survey focused on the incremental value of recreation beside the river as opposed to other locations. Average consumers' surplus was found to be only \$1.02 per user-day (about one-seventh of the value of water-based recreation) but the total value amounted to \$3.53 million.

Overall, the study showed that Alberta residents experience the equivalent of about \$4.38 million in consumers' surplus benefits from recreational activities in the Bow River valley. This figure clearly attests to the importance that people place on urban recreation.

One unexpected finding of the study was that the demand curves for many people participating in water-based recreation had a rather unusual shape. Normally, demand curves tend to be convex-to-the-origin in shape, with demand decreasing steadily as commodity prices rise (see Fig. 2). Our results showed that 77% of respondents described a box-shaped demand curve (Fig. 3), where the demand for river recreation remained constant until a certain threshold price was reached. Above this threshold price, recreationists would no longer choose

Figure 2. Example of a demand curve for most commodities.

Shaded area represents the extent to which conventional methods would have under-estimated the value of recreation.

Figure 3. Example of the box-shaped demand curve reported by 77% of the people who use the Bow River for water-based recreation.

to use the Bow River. The reasons for this unusual shape are believed to be the result of recreationists having a fixed amount of time available for recreational pursuits and a lack of easily available alternative recreational resources from which to choose. The importance of this observation concerning the shape of the demand curve lies in the fact that many previous studies of fishing and recreational values have assumed linear demand curves and, by doing so, the real value of the resource may have been seriously underestimated. Had we used a conventional approach for measuring values, the resulting estimates would have been about 20% lower per user-day. Thus, it is imperative that contingent valuation studies determine the shape of demand curves for recreational resources if the true value of those resources are to be determined.

EXISTENCE/OPTION VALUES

Economist John Krutilla (1967) developed the concept of existence and option values to denote the benefits that people might experience simply from knowing that a resource is available for them to use. Such values are believed to be quite large for wilderness resources, but they can also be associated with other types of recreational resources. Unfortunately, existence and options values are difficult to measure and there are very few instances where reliable estimates of these values have been produced.

In the Bow River Recreation Study we attempted to quantify existence and option values for households that did not actually use the Bow River valley for any form of outdoor recreation. Survey results showed that 94% of these households were prepared to pay an annual fee of between \$1 and \$150 to protect the scenic and recreational features of the valley. On average this amount was determined to be \$25 per household, for total existence and options values of about \$0.64 million. This amount is equivalent to about 75% of the consumers' surplus benefits associated with water-based recreation on the Bow River.

EFFICIENT COUNTS OF RIVER USERS

One of the biggest problems facing us in this study was to provide a reliable estimate of the number of water-based recreationists using a 111-km stretch of the Bow River. Most similar studies, like creel censuses, do instantaneous counts by observation, but typically cannot account for people who leave before or arrive after the count was taken. These omissions can be quite large and can bring the reliability of the final counts into question.

For the Bow River Recreation Study, this problem was resolved by establishing daily activity patterns along each reach of the river. By establishing when people arrived, when they left, and the size of their party, we were able to establish what portion of total daily recreation activity occurred during each one hour of the day. For the summer months, for example, we found that 51.96% of people were on the river between the hours of 1 pm and 2 pm and that 16.30% were on the river between 7 pm and 8 pm (Fig. 4). By conducting comprehensive counts of river users during these time intervals, we were able to scale our estimates to account for river users that were excluded from the counts.

The sample design for our counting program incorporated both Latin-square and lattice design to account for factors such as time of day, reach of the river, weekend or weekday, and direction of travel on the river. Sample days were selected at random and the basic design was replicated four times during the summer period. Survey crews were active during 20.8% of possible sampling intervals and the resulting estimate of recreational activity proved accurate to within 14.4% at a 95% level of confidence. Given the amount of variation in the counts, survey crews would have had to undertake counts during at least 80% of sampling intervals in order to have achieved the same degree of accuracy using a conventional counting procedure. Thus, the modified counting procedure proved to be highly effective in producing accurate count data.

CONCLUSIONS

The most significant outcome of the Bow River Recreation Study is the clear demonstration of the importance of rivers as urban recreation resources. The people of Calgary were found to derive benefits equivalent to about \$5.0 million annually from recreation in the Bow River valley, and the guided angling industry generated \$1.9 million for the provincial economy. With this information, water managers are now in a better position to understand some of the recreational implications of their decisions regarding water quality and quantity.

While the Bow River Recreation Study has provided important new knowledge about the importance of river recreation, there is much more that must be known. For example, the effects of changes in water quality and/or quantity on the amount of recreational activity and associated values must be established. In addition, we have no idea about whether the value of recreational changes from one river to another or whether these values change over time. Thus,

he work that we have done on the Bow River has just begun to address questions related to the recreational importance of rivers and their implications for water management decisions.

Figure 4. Daily activity patterns for water-based recreation on the Bow River in 1987.

REFERENCE

Krutilla, J. 1967. Conservation Reconsidered. In: American Economic Review, September, 1967.

Maintenance of Stormwater Retention Lakes

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ABSTRACT

Permanent stormwater retention lakes have become an integral part of the storm sewage collection system in the City of Edmonton. The first lake was constructed in 1978. Ten years later, in 1988, there were eight lakes in operation and two others were nearing completion. The long-range planners estimate that in the next 50 years there will be close to 100 lakes in operation within the boundaries of the City of Edmonton. The major maintenance activities associated with these lakes are related to the growth of weeds and algae during the summer months. The control of these two nuisances is expected to become more difficult as the lakes mature. The Environmental Services Department of the City of Edmonton will have to develop alternative maintenance techniques for preserving an acceptable water quality in these lakes.

Drinking Water Odour Caused by Organic Runoff and Disinfection

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ABSTRACT

A long history of springtime taste and odour problems in Edmonton culminated in a series of severe, short-term odour incidents in the spring of 1985. An independent inquiry into the safety and quality of Edmonton's drinking water supply was commissioned in response to the widespread public concern. This investigation confirmed that the Rossdale Water Treatment Plant was severely affected by contaminated runoff from upstream storm sewer outfalls. However, part of the 1985 odour incident and a repeated incident in 1986 affected both the Rossdale Plant and the upstream E.L. Smith Plant.

These other incidents required some explanation beyond the hydrocarbon contamination which arose from the storm runoff. That explanation was ultimately traced to the presence of relatively high concentrations of some odourous, low molecular weight aldehydes in the treated water. These contaminants were apparently not present in the raw water, suggesting that they were formed in the treatment process. Further investigations have demonstrated that these aldehydes can be efficiently produced by the chlorination and/or chloramination of common amino acids. The relevance of this finding was further demonstrated by analyzing some swimming pool samples which confirmed the presence of the aldehydes in waters known to contain both chlorine and amino acids.

Man-Made and Natural Algal Blooms in Drinking Water Reservoirs

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ABSTRACT

Treatment of phytoplanktonic blooms with non-selective pesticides, such as copper sulphate, severely reduces the ecological stability of the treated reservoirs. The destruction of non-target organisms, such as benthic diatoms and zooplankton, results in the proliferation of fast-growing opportunistic phytoplankton species. These secondary or man-made algal blooms lead to filtration, taste, odour, and stain problems. Destructive pesticides should only be resorted to for specific control of cyanobacterial blooms. The toxins released by the dead cyanobacteria remain in the water for several weeks and water consumption should be restricted following treatment.

These problems are better dealt with using strategies that prevent the formation of cyanobacterial blooms. In addition, suitable treatments should leave the biological control mechanisms intact. Effective prevention may be based on the channelling of nutrients into biomass that does not prevent a nuisance. Alternatively, a decrease in phytoplanktonic photosynthesis may be achieved through reductions in available light.

**TOXICOLOGY AND TREATMENT
OF PULP MILL EFFLUENTS**

Biological Decolourization of Pulping Effluents

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ABSTRACT

Effluents from existing bleached kraft mills in Alberta are subjected to biological secondary treatment in aerated lagoons prior to discharge into surface waters. Although such treatment does an excellent job of reducing BOD and suspended solids, it has virtually no effect upon the characteristic dark brown colour of such effluents. The colour is primarily due to high-molecular-weight lignin and lignin derivatives released during chlorine bleaching of kraft pulp.

A group of wood-rot fungi, the "white-rots", capable of biodegradation of lignin have been shown to be able to decolourize pulping effluents enzymatically. Trials have been conducted on the capacity of fungi of this type to decolourize bleach plant waste streams containing most of the colour before they reach the main sewer. Trials included screening of over 100 isolates for superior decolourization rates and tolerance to high-strength effluents. Experiments employing resting cells of superior cultures, such as *Trametes hirsuta*, have produced better than 85% colour reduction and tolerances of up to 18,000 colour units. Cells of that fungus have been demonstrated to decolourize up to 18 successive batches of effluent over 67 days when supplied with an energy source under non-growth conditions.

INTRODUCTION

Until relatively recently, bleached kraft pulp was the only kind of pulp produced in Alberta. Recent announcements of new projects have changed the picture dramatically but bleached kraft pulp is still the premium product and should enjoy a strong market for the foreseeable future. A source of concern in recent years has been the dark-coloured effluent released from kraft mills to surface waters, such as the Wapiti River near Grande Prairie and the Athabasca River at Hinton.

The main objective in any chemical pulping process is to allow separation of the cellulose fibres in wood by removing the lignin which binds them together. In the kraft process this is done by "cooking" wood chips in an alkaline solution of sodium sulphide or "white liquor" at

elevated temperatures (Casey, 1980). Kraft pulp is bleached to the desired degree of brightness by a multistage bleaching process (Casey, 1980). The pulp is washed between bleaching steps; these washings make up the bulk of the bleachery wastewater. In a conventional bleaching sequence, a dark brown solution of lignin and chlorinated lignin derivatives result from alkaline extraction of the pulp after bleaching with chlorine gas. This waste stream, known as E1 effluent, may account for 70 - 80% of the colour leaving the mill in about 15% of the total flow.

Wastewater generated by bleached kraft mills in Alberta is subjected to biological secondary treatment before release to surface waters. The two existing kraft mills in Alberta use aerated lagoons for this purpose. In such a lagoon or aerated stabilization basin, effluent flows through a shallow pond typically over a period of 7 - 10 days where, assisted by mechanical aeration to provide oxygen, a mixed population of microorganisms oxidizes pollutants before the treated effluent is finally released into a river or stream. These basins were designed for reduction of biological oxygen demand and suspended solids in pulping effluents and generally perform this function very well.

Aerated lagoons do not reduce the brown colour of bleached kraft mill effluent (BKME) due to lignin and lignin derivatives which pass through lagoons virtually unchanged. Despite the lack of hard evidence of negative effects of such coloured effluent to livestock, fish, or humans there has been an adverse public reaction on aesthetic grounds, if nothing else, to the sight of dark plumes entering rivers. Companies have come under increasing pressure in recent years from Alberta Environment to reduce the colour of BKME entering surface waters. More stringent standards have been incorporated into new operating licenses.

Strategies for BKME colour reduction usually focus upon the E1 stream, the most concentrated source of colour. Processes for achieving this objective normally involve the use of physical/chemical methods during the alkaline extraction stage, after the fact upon the E1 effluent stream, or a combination of both. Some of the methods considered have included ultrafiltration, adsorption on resins or direct partial bleaching of E1 streams. The former two methods are rather expensive and the use of chlorine bleaches gives rise to concerns about increased levels of chlorinated organics in the environment.

Another option given serious consideration in recent years is a biological approach to colour reduction. The normal population of bacteria found in biological treatment systems of kraft mills cannot degrade lignin and consequently cannot reduce colour. Commercial mixtures

of "mutant bacteria" have been and still are being sold to mill operators on the basis of claims of lignin breakdown and concomitant colour reduction. Quite simply, they do not work! In a recent review of the literature on the microbial degradation of lignin (Kirk and Farrell, 1987) no evidence was found for the ability of any bacteria to degrade lignin. Lignin degradation appears to be the exclusive property of a group of wood-rot fungi known collectively as the "white-rots" which can enzymatically breakdown both the cellulose and lignin present in wood.

It has also been found that these same fungi can decolourize E1 effluent by breaking down soluble lignin and chlorolignins (Eaton *et al.*, 1982, Royer *et al.*, 1983). In fact, a patented "Mycor" process, has been developed for effluent decolourization employing the white-rot *Phanerochaete chrysosporium* on a rotating biological contactor (RBC) (Chang *et al.*, 1985, 1987). *P. chrysosporium*, has been studied quite intensively as a model organism producing ligninase (Kirk and Farrell, 1987).

Since studies on effluent decolourization by white-rots had concentrated almost exclusively upon one organism, *P. chrysosporium*, it was felt that this field warranted broader-based studies aimed at finding other, possibly superior, systems.

A broad screening procedure was adopted to test the capacity of known white-rot fungi to decolourize E1 effluent from an Alberta bleached kraft pulp mill. E1 effluent, enriched with nutrients to support growth, was adjusted to a standard 15,000 colour units using a scaled-down version of the Canadian industry standard H5P method of colour measurement and buffered to pH 4.5. Flasks of sterile E1 medium were incubated for periods of 2 - 4 weeks after inoculation with test fungi. At the end of this time, biomass was filtered off and colour was measured. In this screening procedure, flasks were not agitated and media was deliberately left nitrogen-deficient, since Kirk and Farrell (1987) had previously reported that agitation inhibited production of lignin-degrading enzymes and that nitrogen starvation acted as a "trigger" for their induction.

RESULTS AND DISCUSSION

A total of 130 white-rot cultures was screened. Some cultures would neither grow nor decolourize in the test medium; others grew but did not reduce colour; a majority of the white-rots displayed differing degrees of activity; some were capable of better than 80% colour reduction. A short list of 14 superior cultures was drawn up (Table 1) from the results of this screening. Interestingly enough, three of them are edible mushroom species: *Pleurotus ostreatus*, *P. sajor-caju* and *Kuhneromyces mutabilis*. As might be expected *P. chrysosporium* appears on the list.

Table 1. White-rot species with superior decolourizing ability.

SPECIES
<i>Arthroconidial basidiomycete</i> ¹
<i>Chrysosporium pruinosum</i>
<i>Kuhneromyces mutabilis</i>
<i>Lenzites betulina</i>
<i>Merulius tremellosus</i>
<i>Phanerochaete chrysosporium</i>
<i>Phlebia merismoides</i>
<i>Ptychogaster rubescens</i>
<i>Pleurotus ostreatus</i>
<i>Pleurotus sajor-caju</i>
<i>Polyporus adustus</i>
<i>Polyporus versicolor</i>
<i>Radulum caesarium</i>
<i>Trametes hispida</i>

¹Unidentified white rot with clamp connections.

When the fungi on this short list were further tested for performance on high concentrations of filter-sterilized effluent (previous trials employed autoclaved media) the list shortened considerably. In all probability this was due to the inhibitory effect of volatile low-molecular-weight organics driven off by autoclaving. When cultural characteristics were

taken into consideration one culture emerged as the best candidate for further study: *Trametes hispida*. Subsequent studies were carried out on that species.

The objective in these studies was to work toward development of a fixed biomass bioreactor illustrated diagrammatically in Fig. 1. Ideally, active fungal cells would be immobilized in the reactor while the effluent passed through it, becoming decolourized in the process.

Despite previous reports (Kirk and Farrell, 1987) that lignin degradation was incompatible with agitation, growth and decolourization took place in shake flasks in nitrogen-deficient medium as long as a surfactant, Tween 80, was incorporated into the medium (Fig. 2). In the absence of Tween 80, growth was suppressed for 120 h and no decolourization took place. In some way the presence of Tween 80 was able to counteract the inhibitory effect of some component of the effluent. Possibly, the inhibitor was a low-molecular-weight volatile component, enough of which was lost by evaporation to permit growth after 120 h.

The effect of nitrogen starvation on decolourization by growing cells of *Trametes* (in the presence of Tween 80) is demonstrated in Fig. 3. In a well-balanced growth medium in which the C:N ratio of added nutrients is adjusted to 12.9, growth continued unabated for the full course of the experiment; decolourization however was insignificant. When growth was limited by reducing nitrogen to one tenth of its previous level, decolourization took place. *Trametes hispida* appears to fit the *P. chrysosporium* model cited by Kirk and Farrell (1987), i.e. enzymatic lignin degradation is triggered by nutrient limitation, such as nitrogen starvation.

The hypothesis of Kirk and Farrell (1987) is that degradation of lignin is an idiophasic event, occurring as part of secondary metabolism when cell growth has ceased or is slowing down. That being so, experimentation moved from growing cell systems to systems of pre-grown washed cells suspended in a medium devoid of nitrogen which would not allow growth (resting cells).

Fungal cells were grown in submerged culture to a pre-determined age, harvested, washed and exposed to E1 effluent supplemented with an energy source and mineral nutrients but no nitrogen. Cells were used in the normal form in which they grow, i.e. small spherical pellets

Figure 1. Schematic representation of fungal decolourization bioreactor.

Figure 2. Effect of 0.1% Tween 80 upon growth of *Trametes hispida* and decolourization of E1 Effluent in submerged agitated culture.

Figure 3. Effect of nutrient nitrogen (C:N ratio) levels upon growth of *Trametes hispida* and decolourization of E1 effluent in submerged agitated culture.

2 - 3 mm in diameter. It was found that resting cell populations were able to rapidly decolourize E1 under vigorous agitation in air and were indifferent to the presence or absence of Tween 80.

Experiments some of which are reported here were carried out to characterize this system. Lignin apparently cannot act as a sole carbon and energy source for lignin-degrading fungi (Kirk and Farrell, 1987); it needs a co-substrate, normally cellulose in the natural environment. The dependence upon an energy source for decolourization by resting cells can be seen in Fig. 4. A low level of decolourization took place in the absence of added glucose, probably using endogenous energy reserves, such as glycogen. A significant stimulation of activity was seen when glucose was supplied. Glucose concentration was limiting up to 0.5% w/v. A wide range of carbohydrates will support decolourization, including maltose, xylose, cellobiose and sucrose.

Lignin degradation does not take place in the absence of molecular oxygen (Kirk and Farrell, 1987). No decolourization by *Trametes* resting cells occurs in the absence of oxygen. Metabolic inhibitors known to block aerobic respiration, such as azide and cyanide, effectively block decolourization by resting cells (Fig. 5).

If a bioreactor treatment system is to be designed to decolourize effluent using a pre-grown, non-proliferating resting cell population, then an important question to be answered is: "What is the useful life of that cell population?"

To answer that question an experiment was set up which began like the standard shake-flask resting cell experiment except that after initial decolourization had taken place, mycelial pellets were allowed to settle, and decolourized effluent was decanted and replaced with fresh effluent plus nutrients. This procedure was repeated at alternating three- and four-day intervals for more than two months. In effect a batch-fed decolourization system was used.

A cell population of 100 mg dry weight biomass in a working volume of 100 mL was able to decolourize 19 successive batches of effluent over 67 days at 30 °C (Fig. 6). After 67 days, activity decreased quite markedly. This trial demonstrated quite clearly that colour reduction was by a chemical change, mediated biologically; it was not simply a matter of absorption and concentration by fungal cells. Even after 19 batches of effluent decolourization, the fungal pellets remained the same pale "cream" colour inside and outside; if they had been merely absorbing and concentrating colour, they would have become extremely dark brown in colour.

Figure 4.

Effect of glucose concentration upon decolourization of E1 effluent by resting cells of *Trametes hispidia*.

Figure 5. Effect of inhibitors upon decolorization of E1 effluent by resting cells of *Trametes hispida*.

Figure 6.

Batch-fed decolourization of E1 effluent by a population of resting cells of *Trametes hispidia*.

Lignin and lignin derivatives responsible for the colour of E1 effluent are macromolecules. These molecules are too large to pass through the cell membrane to be degraded intracellularly; the decolourizing enzyme(s) must act outside the cell membrane. However, the latter experiment (Fig. 6) clearly indicated that the enzyme(s) must be cell-bound. If enzymes were to be released into free solution in the medium, as has been demonstrated with *P. chrysosporium* (Kirk and Farrell, 1987), then it would be lost rapidly as medium was repeatedly replaced (no nitrogen was supplied for synthesis of new enzyme), resulting in a rapid loss of decolourizing activity. Attempts to demonstrate enzyme activity in culture filtrates of decolourized cells of *Trametes hispida* have not met with success. The enzyme(s) presumably are bound to the cell wall or possibly an associated polysaccharide layer or they might occur in the periplasmic space (between the cell membrane and cell wall).

The economic feasibility of biological decolourization is yet unproven but it does have certain advantages over physical/chemical methods. It is relatively clean, producing no chemical sludges for disposal nor problematic stack gases for incineration. Unlike chlorine bleaching, it does not offer the potential of generating additional chlorinated organic compounds. In fact, white-rot fungi, such as *Phanerochaete*, have the remarkable capability of degrading a wide range of polyaromatic hydrocarbons and toxic low-molecular-weight chlorinated organics, such as chlorophenols, PCB, DDT and dioxins (Bumpus *et al.*, 1985; Haemmerli *et al.*, 1986; Huynh *et al.*, 1985). The enzyme system responsible for lignin degradation has been shown to have a broad substrate specificity, showing activity on a wide variety of condensed aromatics. The discovery of this kind of activity has led to serious consideration of the development of bioreactors for detoxification of hazardous wastes by white-rot fungi (Chang *et al.*, 1985).

REFERENCES

- Bumpus, J.A., M. Tien, D. Wright, and S.D. Aust. 1985. Oxidation of persistent environmental pollutants by a white-rot fungus. *Science* 228:1434-1436.
- Casey, J.P. (Ed.). 1980. *Pulp and paper. Chemistry and chemical technology*. 3rd ed. Vol. 1. Wiley and Sons, New York, NY. 820 pp.
- Chang, H-M., T.W. Joyce, and T.K. Kirk. 1985. Process of degrading chloro-organics by white-rot fungi. U.S. Patent No. 4,554,075.

- Chang, H-M., T.W. Joyce, and T.K. Kirk. 1987. Process of treating effluent from a pulp or papermaking operation. U.S. Patent No. 4,655,926.
- Eaton, D.C., H-M. Chang, and T.W. Joyce. 1982. Method obtains fungal reduction of the color of extraction-stage kraft bleach effluents. Tappi J. 65:89-92.
- Haemmerli, S.D., M.S.A. Leisola, D. Sanglard, and A. Fiechter. 1986. Oxidation of benzo(a)pyrene by extracellular ligninases of *Phanerochaete chrysosporium*. Veratryl alcohol and stability of ligninase. J. Biol. Chem. 261:6900-6903.
- Huyn, V-B., H-M. Chang, T.W. Joyce, and T.K. Kirk. 1985. Dechlorination of chloro-organics by a white-rot fungus. Tappi J. 68:98-102.
- Kirk, T.K., and R.L. Farrell. 1987. Enzymatic "combustion": The microbial degradation of lignin. pp. 465-506. In: Annual Review of Microbiology. Vol. 41. (Ornston, L.N., A. Balows, and P. Bauman, Eds.). Annual Reviews Inc., Palo Alto, CA.
- Royer, G., D. Livermoche, M. Desrochers, L. Jurasek, D. Rouleau, and R.C. Mayer. 1983. Decolourization of kraft mill effluent: Kinetics of a continuous process using *Coriolus versicolor*. Biotechnol. Lett. 5:321-326.

Aquatic Toxicity of Kraft Mill Effluents with Emphasis on Chlorinated Organic Substances

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INTRODUCTION

This is intended to be a summary paper, providing some general scientific information and perhaps a point of view. Most of the paper is derived from material collected by the author for a report to the Ontario government, and unsupported statements are derived from data referenced in that report (Bonsor *et al.*, 1988). Some other material was compiled by Alan Colodey of Environment Canada, Vancouver, BC.

KRAFT PULP MILLS

All kraft mills recycle most of their wastes. They must recover at least 90% of their cooking chemicals in order to be profitable, and very well run mills might attain 95% recovery or more. In doing that, the mills also burn more than 90% of the unwanted organic matter from the wood. An ordinary kraft mill that makes brown paper therefore need not cause great problems with regard to water pollution.

Bleaching of the pulp to make white paper is the one process in the mill that is not on a recycling loop. One mill in the world (at Thunder Bay) tried to close that loop and produce an "effluent-free mill" but had to give up the attempt in the early 1980s. From the bleach plant, therefore, there is a waste stream containing various kinds of organically-bound chlorine, up to 6 tonnes per day of such substances from a large bleached kraft mill (measured as the weight of chlorine, not the entire molecule). Various authorities state that 200 to 300 different organochlorines have been found in the waste stream, but that they represent only 3 to 10% of the total weight of organochlorines. There may be, perhaps, 400 to 600 low-molecular-weight organochlorines in the effluent; the high-molecular-weight ones are not very toxic (IPK, 1982; SSVL, 1985; McKague, 1988).

CURRENT REGULATION BY CATEGORY OF POLLUTANT

There are three "old-fashioned" categories of pollutants in pulp mill waste:

- suspended solids
- oxygen-demanding substances
- toxic substances

Those general categories have been regulated in Canada according to federal limits that came into effect in the early 1970s. Various provinces have added particular requirements, for example site-specific ones are used in Alberta, depending on the size and sensitivity of the body of water that might be affected.

Solid pollutants such as wood fibres and particles of bark were once a major type of pollutant. Settled solids sometimes blanketed the bottoms of rivers and lakes, and up to 60 kg of fibres might be discharged for every tonne of pulp produced. Solids are regulated federally on that basis of amount discharged per unit of production (kg t^{-1}). Suspended solids (SS) are no longer a problem at most Canadian mills, since they are easily settled out in ponds. Alberta kraft mills averaged about 10 kg SS t^{-1} in 1985, below the national average of 15.

Oxygen demand (Biochemical oxygen demand or BOD) is created by the decomposition of organic matter and is also regulated federally in terms of kg of BOD t^{-1} of pulp. Kraft mills have much lower BOD loadings because of their recycling, than do sulphite mills and the newer TCMP mills. Still there have been serious problems of low oxygen in the past when mills might discharge as much as 50 kg BOD t^{-1} , compared to the federal regulations of 30 kg t^{-1} . The best mills can produce an effluent with 3 kg BOD t^{-1} , by means of within-mill practice followed by efficient secondary waste treatment, such as the ponds or "lagoons" at the existing kraft mills in Alberta. Oxygen concentrations deserve special attention if there is not much dilution available in the receiving water. For the proposed mill at Athabasca, for example, effluent concentration might be as high as 2% under extreme conditions of low flow in the winter. Still, if that mill achieved BOD of 3 kg t^{-1} , lowering of oxygen in the river would be only about 1 mg L^{-1} compared to saturation levels in the vicinity of 14 mg L^{-1} , even if there was no reaeration.

ACUTE TOXICITY

The third general category of pollutant from a pulp mill is toxic material. In Canada, that has been regulated for the past two decades by a lethality test with rainbow trout at the "end of the pipe" (the requirement is less than 20% mortality of trout in 65% effluent concentration). That is a simple and convincing kind of test and has often been a powerful means of encouraging pollution cleanup. The test has two major drawbacks, however, which suggest that it should either be retired from its role of monitoring pulp mills, or supplemented with other approaches.

The first problem is that the lethality test measures concentration of toxic substances in the effluent, while it is the amount of toxic material mixed into the receiving water that is important to the aquatic ecosystem. The test therefore discourages water conservation within the mill. In fact, reduction in water use should be a primary goal because it is accompanied by efficient design and operation, and by the control of leaks and spills that contribute most pollutants. Government control agencies that wish to continue using the fish lethality test should modify their regulations to include a factor for volume of discharge (i.e. toxicity times volume, the "toxic units" approach).

The second problem with the 4-d lethality test with fish is that it does not measure the discharge of the subtle, persistent, bioaccumulative toxicants that should be of most interest. Current toxicity tests on pulp mill effluent detect, in large part, the action of non-persistent chlorinated or unchlorinated substances that have been cooked out of the wood.

Components Causing Acute Toxicity

Among the non-volatile substances, those causing most of the acute lethal action of bleached kraft mill effluent (bleached kraft mill effluent) are resin and fatty acids, diterpenes, pitch dispersants, and a range of chlorinated substances including lignins, phenolics, and again, the resin and fatty acids (Table 1). All of these major contributors of acute lethality tend to decompose in relatively short times of a few days or weeks.

The chlorinated lignins, resin and fatty acids, and phenolics shown in Table 1 originate in the bleach plant. Some within-mill studies indicate that the bleach plant waste accounts for about half of the acute lethality of the total mill effluent. In one Canadian mill, the bleach plant effluent had a Toxicity Emission Factor of 300 units (Toxic Units $\times$ m^3 of effluent t^{-1} of pulp). That was almost exactly half of the Toxicity Emission Factor for the whole mill effluent

Table 1. Principal constituents in effluents of bleached kraft mills that cause acute lethality to fish, with sub-items listing in approximate order of prominence. From Bonsor *et al.* (1988), based on Leach and Thakore (1973, 1977), supplemented from McKague (1981) and Walden and Howard (1977).

Effluent stream	Major toxic contributors	Less toxic contributors
Debarking	Resin acids Isopimaric Dehydroabietic Abietic Pimaric	Diterpene alcohols Pimarol Isopimarol Dehydroabietal
Pulping (unbleached whitewater)	Resin Acid Soaps Na isopimarate Na abietate Na dehydroabietate Na pimarate	Sodium salts of unsaturated fatty acids Palmitoleic Oleic Linoleic
Acid chlorination	Chlorolignins 4,5-Dichlorocatechol 3,4,5-Trichlorocatechol Tetrachlorocatechol 2,6-Dichlorohydroquinone	Not Known
Caustic extraction	Chlorinated stearic acids Epoxy stearic acid Dichlorostearic acid Chlorinated resin acids Monochlorodehydroabietic acid Dichlorodehydroabietic acid Chlorinated phenolics Tetrachloroguaiacol Trichloroguaiacol	Liquid pitch dispersants

(610 units, Craig *et al.*, 1986). Similarly, bleach plant effluents in another mill had 740 TEF compared to 1400 for the whole mill (Scroggins, 1986). The relatively large contribution is, of course, because the wastewater from the bleach plant cannot be recycled.

Acute Toxicity of Whole Effluents

If a kraft mill effluent has received only primary waste treatment (settling of solids) it will probably be lethal to fish, although the degree of lethality is extremely variable from mill to mill

and also from time to time. Nevertheless these wastes must be considered to be only mildly toxic. Median lethal concentration (LC_{50}) for rainbow trout may range from 3% to more than 100% (McLeay, 1987), but a geometric mean LC_{50} would be in the vicinity of 16% whole effluent. That average LC_{50} does not represent very high toxicity by the usual standards, and direct mortality would not be expected, after even minimal dilution in the receiving water.

The generalization that untreated whole effluents are only mildly lethal is true overall, whether the effluent comes from a bleached or unbleached mill, or whether or not it has received primary waste treatment. Much of the variation between mills is probably accounted for by control of spills and general tightness of operation.

Secondary (biological) waste treatment can be very effective in removing or greatly decreasing the acute lethal action of kraft mill effluent. This is largely because of decomposition of resin and fatty acids, and the disappearance of their potent short-term toxicity.

SUBLETHAL EFFECTS OF WHOLE EFFLUENT

Protection of ecosystems against sublethal toxicity is of greater importance than is acute lethality, in real-world situations. Although secondary waste treatment is part of existing and proposed kraft mills in Alberta, the toxicity results for effluent that has not received secondary treatment ("untreated") also provide useful benchmarks.

Laboratory Studies with Untreated Kraft Effluent

Excellent detailed reviews on sublethal toxicity have been provided by the Canadians Davis (1974, 1976), Kovacs (1986), and McLeay (1987). The various sublethal effects studied in the laboratory include those given in the following tabulation (Kovacs, 1986).

Biochemical metabolites	Blood
Histopathology	Body composition
Respiration and circulation	Growth
Food conversion	Swimming stamina
Early development and reproduction	Behaviour

Generalizing from the reviews mentioned above, many of the sublethal experiments showed No-Observed-Effect Concentrations (NOEC) at concentrations of 1% or 2% whole kraft mill effluent or bleached kraft mill effluent, with an occasional lower value of 0.5% effluent.

Sublethal Studies with Bio-treated Bleached Kraft Mill Effluent

Sublethal effects may occur only at concentrations of 1% or more, and often as high as 10%. McLeay (1987) summarizes results from long-term exposures to biologically treated bleached kraft mill effluent, in laboratory and artificial ecosystem experiments. Relevant items are given below in tabulated form, and beside each category are the concentrations, as a percentage, found by various investigators to cause an effect. If several effects at different concentrations were documented in the same experiment, only the lowest concentration has been used in the tabulation.

Internal physiological effect	<5		
Respiration, circulation	>1	>10	
Swimming performance	>1	>28	
Tissue abnormalities	1	>1	>2
Growth, development	>5	10	>100
Disease resistance	>2		
Reproduction or young stages	1.3	>7	>100
Productivity, lab or outdoor streams	>2		

There is no value less than 1% in the tabulation of effects of bleached kraft mill effluent. The value of 1.3% has been added to McLeay's tabulation and was for abnormal development of oyster larvae, known to be one of the most sensitive tests available (Woelke *et al.*, 1972). Other findings are for fish, except the two values of greater than 100 which are for lack of effect of full-strength effluent on the waterflea *Daphnia* in life-cycle exposures (Weinbauer and Somers, 1982). Fish appear reasonably sensitive to bleached kraft mill effluent compared to other organisms such as *Daphnia* and algae (Bothwell and Stockner, 1980).

North American Field Surveys and Studies in Artificial Streams

Well-treated kraft mill effluent is only mildly toxic, according to numerous studies of this kind. A striking example is an earthen channel containing the full-strength biotreated bleached kraft mill effluent from a Florida mill; it was home to 25 species of fish of which four were common, and had 6,000 invertebrates m⁻², mostly midge larvae (Juul and Shireman, 1978).

An example closer to home, that shows similar slight toxicity, is a study in the Wapiti River of Alberta (Alberta Environmental Centre, 1987). The treated effluent from the bleached kraft mill was generally non-lethal to fish. In the river, fish carried residues of mill-derived chlorinated organic compounds and had liver damage and reduced levels of glycogen in the liver. Nevertheless, no reduction of fish populations or impairment of reproduction was documented,

and there were 12 species of fish downstream of the mill compared to 10 species upstream. Perhaps the most surprising part of the report is that the downstream sampling stations were only 0.1 km and 2.5 km below the mill, presumably in a section where the effluent was at relatively high concentration because it was still mixing. The Alberta Environmental Centre report does not give a statement of the concentrations of effluent at those locations. I attempted calculations based on relative concentrations in the effluent and at the downstream locations, of various conservative variables such as chloride and other ions. Estimates ranged from 8% to 50% concentration of effluent at the downstream stations, most of them closer to the high figure than the low one. Again that illustrates surprisingly little documented effect of apparently high concentrations of treated bleached kraft mill effluent.

U.S. research in artificial streams also indicates little effect of well-treated bleached kraft mill effluent. In the tabulation above, the underlined values are derived from long-term studies of fish allowed to grow in artificial streams under semi-natural conditions, as carried out by the National Council for Air and Stream Improvement (NCASI), a group sponsored by the U.S. pulp and paper industry. The artificial streams showed effects on fish growth, maturation and reproduction, only at concentrations of 7%, 10%, and higher. A report summarizing various experiments states that "largemouth bass, bluegill sunfish, and golden shiners spawned successfully in the experimental streams at effluent concentrations of about 9 percent ...", and that in "fathead minnow full life cycle tests no consistent affect [sic.] on egg production would be expected until effluent concentrations exceed 18 percent ..." (Borton, 1985).

Mutagenicity

This does not seem to be a major problem with bleached kraft mill effluent, but it is worth examining because of a general concern in the media about carcinogenic hazards resulting from organochlorines in kraft paper products and effluents. A mutagenic chemical changes the gene structure, and of course may not only affect the functioning of an organism, but may also cause changes that are passed on to the next generation. A mutagenic substance is also likely to be a carcinogenic one, but that has to be substantiated for each substance, and the list of confirmed carcinogens is smaller than might be thought. Similarly, some carcinogens are not mutagens, for example the most toxic "dioxin" (2,3,7,8-tetrachlorodibenzo-*p*-dioxin) promotes cancer but is not mutagenic.

Untreated whole bleached kraft mill effluent is, indeed, usually weakly mutagenic (reviewed by Stockman *et al.*, 1980; Langi and Priha, 1988). The strongest mutagenic component is generally the first chlorination effluent, while the caustic extraction effluent seldom carries mutagens (B.C. Research, 1979; Hglund *et al.*, 1979; Nazar and Rapson, 1980; Langi and Priha, 1988). Aerated lagoons remove most or all mutagenicity from bleached kraft mill effluent (B.C. Research, 1979; Hglund *et al.*, 1979).

Substitution of chlorine dioxide for chlorine reduces mutagenicity of bleached kraft mill effluent, and waste from a chlorine dioxide stage of bleaching may show none (Rannug *et al.*, 1981). In experiments with softwood pulp, strong mutagenicity dropped off linearly with substitution, until at 100% ClO₂, mutants were no greater than in the control (Eriksson *et al.*, 1979; Nazar and Rapson, 1980). Nor does waste from a hypochlorite stage show mutagenicity (Eriksson *et al.*, 1979), although it can be revealed by concentrating the effluent (Rannug *et al.*, 1981).

Tainting of Fish

Untreated whole kraft mill effluent may taint fish, but only at concentrations similar to those causing sublethal effects on the fish themselves. Laboratory exposures generally estimate tainting thresholds in the 1% to 5% range of bleached kraft mill effluent concentrations, with some values as low as 0.5% (Shumway and Chadwick, 1971; Cook *et al.*, 1973; Liem *et al.*, 1977; Whittle and Flood, 1977; Gordon *et al.*, 1980). Field studies confirm that tainting may occur at those concentrations, although some results appear to be confounded with natural off-flavours.

Bleaching has not been considered as a significant contributor to tainting, with general opinion that chlorinated phenolics do not have an effect (Kovacs *et al.*, 1984). However, recent work indicates that tri- and tetra-chloroguaiacol can be transformed by bacterial O-methylation to tri- and tetra-chloroveratrole, which have lethal and severe sublethal effects on aquatic organisms (Neilson *et al.*, 1984). Paasivirta (1988) generalizes from his work that chlorophenols from pulp mills are biomethylated into chloroveratroles and chloroanisoles, which bioaccumulate in fish and have a high off-flavour potential.

Secondary waste treatment, or even aeration, greatly reduces tainting (Cook *et al.*, 1973), with biological treatment of bleached kraft mill effluent causing a 2-fold drop (Miettinen *et al.*, 1982), or a 4 to 10-fold drop (Gordon *et al.*, 1980).

SWEDISH RESEARCH ON SUBLETHAL EFFECT

Swedish research indicates that the degree of harmful effects increases with the use of chlorine in bleaching, and that bleached kraft mill effluent may cause toxicity at dilution of 1 in a thousand (0.1%), one or two orders of magnitude more toxic than indicated by the sublethal studies summarized in sections above.

Recent Swedish results come from wide-spectrum field studies under a program "Environmental impact of bleach plant effluent" from 1982 to 1985 (SSVL, 1985), and another called "Environment/Cellulose", from 1983 to 1986. There were also smaller similar projects and cooperative projects involving scientists from Finland and Norway. The multi-disciplinary studies were largely reported at a 1987 conference in Tampere, Finland, and most of the ecological papers from the conference were then published in *Water, Science and Technology*, Vol. 20(2), 1988. The research compared biological effects of bleached effluent from one kraft mill with unbleached effluent from another mill, and with control locations. This was in the Baltic Sea, which has a very low salinity and is populated by both marine species and by species that we usually think of as freshwater.

Effects found near the bleached kraft mill but not at the unbleached mill were:

- absence of fish within 1 km of the discharge
- a superabundance of small and shallow-water fish at 2-3 km distance
- absence of larger deep-water fish within 4 km
- failure of sexual maturation among perch, (see below)
- faster growth of perch (see below)
- fin erosion up to 4 km distance
- deformed skulls in pike
- disturbance of many biochemical and physiological functions.

Some of those effects, and others for which there was no comparison with an unbleached mill, were detected at 10 km from the bleached kraft mill, where dilution of effluent was over 1000-fold. Residues of organochlorines in sediments and in tissues of fish showed gradients with distance from the mill, parallel to the gradients in biological effects.

Some further details are warranted about depressed sexual maturation and growth changes. At 2 km distance from the bleached kraft mill, only about 50% of "potentially mature" female perch had gonads developing, compared to 80% at great distance and at a control location (Sandstrom *et al.*, 1988). Even at 10 km from the mill, the gonads were smaller in relation to body size (Larsson *et al.*, 1988). Perch did not show such a problem near the unbleached mill. However, the perch grew faster near the BK mill than away from it, again a phenomenon that was not seen at the unbleached mill. Sandstrom *et al.* (1988) felt, from other work, that toxicants had disturbed the allocation of energy by the fish. Evidence of such disturbed body chemistry was provided by the biochemical and physiological tests on perch (Larsson *et al.*, 1988). Some of the more important observations were enlarged liver, strong induction of mixed function oxidases ("defence enzymes") of the liver, strongly abnormal metabolism of carbohydrates, and drop in white blood cell count.

Other research involved studies on spinal deformities, artificial ecosystems dosed with effluents of mills using a variety of bleaching techniques, surveys of benthic invertebrates, chemical surveys of organochlorine substances in sediments and biota, and other approaches.

The wide-ranging and intensive Scandinavian research ranks among the most thorough attempts to ascertain the effects of bleached kraft mill effluent. The Swedes at least, are convinced from the results, that chlorinated organic substances add a special and worrisome kind of toxicity to kraft mill effluent. Sweden embarked some years ago on a vigorous program to reduce the use of chlorine in mills, and in the years 1970 to 1985 the amounts of chlorine declined from 85 kg Cl t⁻¹ of pulp to 40 kg t⁻¹. Sweden is currently attempting to persuade other countries to follow the same path.

CRITIQUE OF SWEDISH AND NORTH AMERICAN FINDINGS

Much of the research on sublethal toxicity summarized in sections under "Sublethal Effects of Whole Effluents" is from Canada and U.S.A., and as mentioned, Swedish field studies found effects at concentrations one or two orders of magnitude lower. In particular, NCASI carried out careful and intensive studies with organisms exposed for long periods in artificial streams, and found relatively mild effects. Having read much of the literature, I am convinced that the Swedes are correct about the general harmfulness of organochlorines and the desirability of reducing them. There are, however, some difficulties in interpreting their work, and it is not

at all clear that their effect-concentrations apply to the North American scene. Listed below are some of the questions that arise when interpreting some of the Scandinavian research.

(1) Other toxicants in the Baltic Sea may be acting in concert with those from pulp mills, and lowering the apparent effect-concentrations. Certainly the Baltic has many sources of pollution from the five countries that discharge a diversity of industrial and agricultural wastes into it. There have, in fact, been disturbing reports over the last two decades, of reproductive declines and/or abnormalities of organs in marine mammals and predatory birds from the Baltic region. High levels of some persistent toxicants such as PCBs have been found simultaneously.

(2) One clear reason that a great difference might be expected between the Swedish and NCASI results is the high degree of secondary treatment given to effluent used in the American studies. At the main Swedish mill studied, the effluent received only primary treatment while the effluent studied by NCASI had received 12 - 14 days of treatment including an aerated lagoon (Borton, 1985).

(3) In order to document the particular effects of the bleached component of kraft mill effluent, it is necessary to make a careful and parallel assessment of effects from an unbleached effluent, but sometimes that comparison was not included in the Swedish work. For example, specific assessments of mortality rates in perch populations were done at the bleached mill but not at the unbleached one. Similarly, Larsson *et al.* (1988) found physiological disturbances in fish near the mill that bleached but made the following general statement about studies near the unbleached mill: "... a limited investigation on perch caught in the receiving body of water of a pulp mill without bleaching processes showed no or considerably lower effects on most physiological parameters ..." but the details have not yet been published.

(4) There have been delays in publication of some sets of data. Although many of the results have now been published in refereed journals, some of the information seems to have been reported only at conferences, in summary reports, or non-refereed reports. In some cases, fairly major and often-repeated general findings have not been supported by published data giving the details necessary for an outside observer to evaluate the conclusions. Even the journal publication of the important papers from the Tampere Conference (section on Swedish Research on Sublethal Effects) actually resulted in printing the pre-prints from the conference with little or no change apparent, and certainly without increasing the details of data presented.

The same problem applies to some of the North American research, which does not seem to find its way into refereed journals. That is particularly true of the NCASI studies in artificial streams. However, the detailed findings are presented in reports, which are available in libraries or can be obtained without problem.

(5) The lack of substantiating details is important for evaluating some findings. For example, failure of fish to mature was mentioned as one effect at the bleached mill but not at the unbleached one. We may link that with the observation that fish grew faster near the bleaching mill. Were the gonads small in relation to body size, or age of the fish, or both? If the conclusion about failure to mature was based on ratios of gonad to body size, it could be that the fish were big enough to mature, but not old enough. Evaluating that question needs the detailed raw data and the methods of analysis, which are not usually presented in summary papers or conference papers. The detailed manuscript reports in Swedish are not always readily available.

(6) Sometimes when details of the Swedish research are available, there is more variability than might be supposed. For example the microsomal defence enzyme EROD in the livers of Baltic fish increases regularly with closeness to the bleached kraft mill, and is often presented as a histogram of convincing averages. However, the individual histograms for various months show a great deal of variation and are rather less convincing.

The above items do not invalidate the Scandinavian findings, but perhaps they give pause about accepting concentrations of 0.1% bleached kraft mill effluent as causing harmful effects under North American conditions.

ORGANOCHLORINES AS A CLASS OF TOXIC SUBSTANCES

There is concern about organochlorines produced in kraft mills because of the Swedish results and because of the emphasis in the news media about "dioxins" in paper products. The concern is warranted in a general way. Chlorination of organic molecules tends to make them (a) more toxic and in particular more mutagenic, (b) less biodegradable, and (c) more lipophilic so that they are more likely to bioaccumulate in living organisms.

As mentioned above, a large bleached kraft mill might produce 6 t d^{-1} of organically-bound chlorine, mostly unidentified. Chlorinated dioxins and furans represent an infinitesimally small part of the total, and the concern is that there could be other potent

toxicants, as yet unidentified. Because of the general tendency of these substances to be toxic, the uncertainty about exactly which ones are present, and the impracticality of regulating hundreds of substances, it is logical to attempt to control chlorinated organics in pulp mill effluent, as a class of substances.

Amounts of organochlorines are usually measured as the mass of organically-bound chlorine or halogens. Adsorbable Organic Halogen (AOX) and its near-equivalent Total Organic Halogen (TOX) are becoming the most frequent measurements. Adsorbable refers to adsorption onto charcoal, and in kraft mills the halogen ("X") is almost entirely chlorine. Total Organochlorine (TOCl) has been frequently used, and is often only about 75% of the AOX value.

CONTROLLING ORGANOCHLORINES

Some European countries or regions have established schedules for reducing discharge of chlorinated compounds from pulp mills, by means of regulations for TOCl (total organic chlorine) as noted below:

Location	TOCl Discharge (kg t ⁻¹)	Target Date
West Germany	1.0	1980
Finland	1.5	2000
Norway	2-2.5	1991
Baltic*	2.5	1994
Sweden	1.3-5.5	1988
Sweden	1.5	1992
Sweden	0.1	ca. 2010

*Baltic Sea Environment Ministers (Source: U.B. Fallenius, Swedish Environment Protection Board, Stockholm, communication to Alan G. Colodey, Environment Canada).

The last line of the above tabulation may perhaps be considered as a desirable goal that cannot yet be met if we wish to have kraft paper that is white. There are many ways of reducing the amounts of chlorine used, or the amounts that come out with the effluent (see below), but it is still necessary to use some chlorine to obtain white paper from a kraft mill.

In Canada, recommendations were made to the Ontario government in April 1988 that mills should achieve 2.5 kg AOX t⁻¹ of pulp in 2 or 3 years, and 1.5 kg t⁻¹ by about 1993. No regulatory action has been taken by Ontario at the date of writing, although one mill already meets the proposed 1993 level, of its own volition and apparently partly for economic reasons (E.B. Eddy at Espanola, ON). There have been rumours that Environment Canada would adopt regulations on AOX but again no action has been taken to date.

Alberta seems to have taken the lead in Canada to judge from the characteristics of the proposed new kraft mills. The design for the intended mill near Athabasca incorporates six important features that minimize organochlorines in the waste.

- Hardwood as a major source of wood. (A large component of hardwood forms lower amounts of organochlorines.)
- Extended cooking. (This removes more of the lignins and reduces the need for bleaching.)
- Extensive, well-designed washing of pulp. (This also removes lignin.)
- Oxygen delignification. (Another method of reducing the need for bleaching to remove the lignin.)
- 70% chlorine dioxide substitution (for chlorine) in the first stage of bleaching. (Reduces AOX in the effluent.)
- Secondary waste treatment with total of 15 days of retention. (Although it is better to achieve the desired end by processes within the mill, rather than produce a waste and then treat it, the lagoon should reduce the discharge of organochlorines by at least one third, and will also reduce BOD of the waste, etc.)

Other features would also contribute to reduced discharge of organochlorine. Specific design for good mixing of the pulp with the bleaching chemicals would reduce the need to "over-chlorinate" and would eliminate pockets of high-concentration chlorine. An intended water use of 55 - 65 m³ t⁻¹ of pulp is low for the industry and bespeaks good design of processes. A spill pond would hold wastewater in case of accident and retain it for later treatment. A diffuser over 40% of the river's width would achieve rapid dilution.

If such a mill were built and achieved its design parameters, it would have AOX discharge below 1 kg t⁻¹. That is not an elimination of organochlorine production, but it at least represents the current state of the art in reducing the amounts produced, and it betters the Swedish objectives for 1992.

REFERENCES

- Alberta Environmental Centre. 1987. Toxicity and environmental chemistry of wastewater from a kraft pulp and paper mill: fish toxicity studies. Alberta Environmental Centre, Vegreville, AB. Report No. AECV87-R4. 67 pp.
- Bonsor, N., N. McCubbin, and J.B. Sprague. 1988. Kraft mill effluents in Ontario. Report of the Expert Committee on Kraft Mill Toxicity to Pulp and Paper Sector, MISA, Ontario Ministry of the Environment, Toronto, ON. 260 pp.
- Borton, D.L. 1985. Effects of biologically treated bleached kraft mill effluent during early life stage and full life cycle studies with fish. National Council of the Paper Industry for Air and Stream Improvement, Inc., New York, N.Y. NCASI Tech. Bull. No. 475. 105 pp.
- Bothwell, M.L., and J.G. Stockner. 1980. Influence of secondarily treated kraft mill effluent on the accumulation rate of attached algae in experimental continuous-flow troughs. *Can. J. Fish. Aq. Sci.* 37:248-254.
- B.C. Research. 1979. Biological characteristics of pulp mill effluents. [British Columbia Research Council] Environment Canada, CPAR Secretariat, Ottawa, ON. CPAR Project Report 678. 178 pp.
- Cook, W.H., F.A. Farmer, O.E. Kristiansen, K. Reid, J. Reid, and R. Rowbottom. 1973. The effect of pulp and paper mill effluents on the taste and odour of the receiving water and the fish therein. *Pulp & Paper Magazine Canada* 74:97-106.
- Craig, G.R., M.F. Holloran, K. Schiefer, R.W. Wilson, and A. Burt. 1986. Study of in-mill toxicity and the impact of mill effluent on Lake Superior. A report for James River-Marathon Ltd., Marathon, ON. Submitted to Ontario Ministry of the Environment. Beak Consultants Ltd., Mississauga, ON. Report No. 2243.2.
- Davis, J.C. 1974. Bioassay procedures and sublethal effect studies with bleached kraft pulp mill effluent and pacific salmon. *Pulp & Paper Magazine Canada*, July 1974:D13-D15.
- Davis, J.C. 1976. Progress in sublethal effect studies with kraft pulpmill effluent and salmonids. *J. Fish. Res. Bd. Can.* 33:2031-2035.
- Eriksson, K.E, M.C. Kolar, and K. Kringstad, 1979. Studies on the mutagenic properties of bleaching effluents. Part 2. *Svensk Papperstidning* 82:95-104.
- Hglund, C., A.S. Allard, A.H. Neilson, and L. Landner. 1979. Is the mutagenic activity of bleach plant effluents persistent in the environment? *Svensk Papperstidning* 82:447-449.
- Gordon, M.R., J.C. Mueller, and C.C. Walden. 1980. Effect of biotreatment on fish tainting propensity of bleached kraft whole mill effluent. *Canadian Pulp & Paper Assoc., Trans. Tech. Section*, March 1980. 6:TR2-TR8.

- IPK. 1982. [Industrins Processkonsult AB] Miljvnlig tillverkning av blekt massa. Slutrapport. Projektet utfordes 1977-81. [Environmental harmonization of manufacture of bleached pulp. Final report. Project carried out 1977-81]. IPK, Box 8309, 104 20 Stockholm. 189 pp.
- Juul, R.B., and J.V. Shireman. 1978. A biological assessment of fish and benthic populations inhabiting a kraft mill effluent channel. *Wat. Res.* 12:691-701.
- Kovacs, T.G. 1986. Effects of bleached kraft mill effluents on freshwater fish: a Canadian perspective. *Wat. Poll. Res. J. Can.* 21:91-118.
- Kovacs, T.G., R.H. Voss, and A. Wong. 1984. Chlorinated phenolics of bleached kraft mill origin. An olfactory evaluation. *Wat. Res.* 18:911-916.
- Langi A., and M. Priha. 1988. Mutagenicity in pulp and paper mill effluents and in recipient. *Wat. Sci. Tech.* 20:143-152.
- Larsson, L., T. Andersson, L. Frin, and J. Hrdig. 1988. Physiological disturbances in fish exposed to bleached kraft mill effluents. *Wat. Sci. Tech.* 20:67-76.
- Leach, J.M., and A.N. Thakore. 1973. Identification of the constituents of kraft pulping effluent that are toxic to juvenile coho salmon (*Oncorhynchus kisutch*). *J. Fish. Res. Bd. Can.* 30:479-484.
- Leach, J.M., and A.N. Thakore. 1977. Compounds toxic to fish in pulp mill waste streams. *Prog. Wat. Tech.* 9:787-798.
- Liem, A.J., V.A. Naish, and R.S. Rowbottom. 1977. Evaluation of the effect of inplant treatment systems on the abatement of air and water pollution from a hardwood kraft pulp mill. Environment Canada, CPAR Secretariat, Ottawa, ON. CPAR Project Report 484. 81 pp.
- McKague, A.B. 1981. Some toxic constituents of chlorination-stage effluents from bleached kraft pulp mills. *Can. J. Fish. Aq. Sci.* 38:739-743.
- McKague, A.B. 1988. Characterization and identification of organic chlorine compounds in bleach plant effluents. *Proc. Colloquium on Measurement of Organochlorines, Pulp & Paper Res. Centre, Univ. Toronto, Toronto, ON, Feb. 16-17, 1988.*
- McLeay, D.J. [D. McLeay and Associates Ltd.]. 1987. Aquatic toxicity of pulp and paper mill effluent: a review. Environment Canada, Ottawa, ON. Report EPS 4/PF/1. 191 pp.
- Miettinen, V., B.E. Lnn, and A. Oikari. 1982. Effects of biological treatment on the toxicity for fish of combined debarking and kraft pulp bleaching effluent. *Paperi ja Puu* 64:(Special no. 4):251-254.
- Nazar, M.A., and W.H. Rapson. 1980. Elimination of the mutagenicity of bleach plant effluents. *Pulp & Paper Canada* 81:T191-T196.

- Neilson, A.H., A.S. Allard, S. Reiland, M. Remberger, A. Trnholm, T. Viktor, and L. Landner. 1984. Tri- and tetra-chloroveratrole, metabolites produced by bacterial O-methylation of tri- and tetra-chloroguaiacol: an assessment of their bioconcentration potential and their effects on fish reproduction. *Can. J. Fish. Aq. Sci.* 41:1502-1512.
- Paasivirta, J. 1988. Organochlorine compounds in the environment. *Wat. Sci. Tech.* 20:119-129.
- Rannug, U., D. Jenssen, C. Ramel, K.E. Eriksson, and K. Kringstad. 1981. Mutagenic effects of effluents from chlorine bleaching of pulp. *J. Toxicol. Environ. Health* 7:33-47.
- Sandstrom, O., E. Neuman, and P. Kars. 1988. Effects of a bleached pulp mill effluent on growth and gonad function in Baltic coastal fish. *Wat. Sci. Tech.* 20:107-118.
- Scroggins, R.P. 1986. In-plant toxicity balances for a bleached kraft pulp mill. *Pulp & Paper Canada* 87:T344-T348.
- Shumway, D.L., and G.G. Chadwick. 1971. Influence of kraft mill effluent on the flavor of salmon flesh. *Wat. Res.* 5:997-1003.
- SSVL. 1985. [Stiftelsen Skogsindustriernas Vatten och Luftvdrdsfo-rskning] Risker fr pverkan p miljn genom utslp av blekeriavlopp. Projektet utfdrdes 1982-1985. SSVL-85 Rapport Nr 44, Slutrapport delprojekt 4, Framstllning av blekt massa. F-IPK, Box 8309, 104 20 Stockholm, 84 pp. [Environmental impact of bleach plant effluent. Project carried out 1982-1985. SSVL-85 Report No. 44, Final Report subproject 4, Manufacture of bleached pulp.] [Read in the original and in the translation by Ralph McElroy Co., P.O. Box 4828, Austin, TX 78765, USA].
- Stockman, L., L. Stromberg, and F. de Sousa. 1980. Mutagenic properties of bleach plant effluents: present state of knowledge. *Cellulose Chem. Technol.* 14:517-526.
- Walden, C.C., and T.E. Howard. 1977. Toxicity of pulp and paper mill effluents. A review of regulations and research. *Tappi J.* 60:122-125.
- Weinbauer, J.D., and L.S. Somers. 1982. Chronic toxicity testing of treated pulp and paper mill effluent. pp. 21-27. *Proc. 1982 Tappi Environ. Conf.*
- Woelke, C.E., T. Schink, and E. Sanborn. 1972. Effect of biological treatment on the toxicity of three types of pulping wastes to Pacific oyster embryos. U.S. Envir. Protection Agency Contract No. 68-01-377. Washington State Dept. Fisheries, Olympia, WA. [Cited through McLeay, 1987.)
- Whittle, D.M., and K.W. Flood. 1977. Assessment of the acute toxicity, growth impairment and flesh tainting potential of a bleached kraft mill effluent on rainbow trout (*Salmo gairdneri*). *J. Fish. Res. Bd. Can.* 34:869-878.

Toxicity and Environmental Chemistry of Wastewater from a Kraft Pulp and Paper Mill

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ABSTRACT

The purpose of this investigation was to determine if effluent discharged from the Procter and Gamble Cellulose Ltd. (Grande Prairie) kraft process pulp and paper mill was deleterious to fish in the Wapiti River. The presence and concentrations of organic and inorganic chemicals in effluent, river water and selected fish tissues were determined. In addition, selected tissues from fish resident in the river adjacent to the mill were examined for evidence of pathological changes. Measurement of the chemical constituents of upstream and downstream river water and mill effluent in this study was done for the purpose of correlating the chemical constituents of the water with residues in fish. As such, it does not constitute a complete study of the chemistry of either the water or the mill effluent and may not represent the chemical status of the river downstream of the immediate zone of influence of the effluent. However, the results reported herein represent a more in-depth assessment of water and effluent chemistry than is normally carried out for monitoring purposes. For a more complete assessment of downstream water quality, reference should be made to existing water quality and bioassay data.

It has been concluded that:

1. Effluent from the mill was not usually acutely toxic to fish under natural or laboratory conditions.

Twelve species of fish were found immediately below the waste discharge point compared with ten species in the upstream control area. The age distribution of the species caught above and below the discharge point was similar. Based on the presence of young-of-the-year, at least seven species of fish appeared to reproduce in the vicinity of the mill. Acute toxicity tests of whole mill effluent conducted over a period of several years were usually negative.

2. Effluent from the mill contained alkanes, fatty acids, phenolic compounds, dissolved solids, and high chemical oxygen demand all commonly associated with

pulp mill effluent. Discharge of these constituents lead to a measurable increase in their concentrations in the Wapiti River immediately below the mill.

3. Fish captured between 0.1 - 2.5 km below the mill contained alkanes, fatty acids, chlorophenol and guaiacols, and α - α' -dichlorodimethylsulphone in their tissues, at concentrations ranging from trace (detected but not quantifiable) to $5 \mu\text{g kg}^{-1}$ (wet weight, whole fish), except for α - α' -dichlorodimethylsulphone which was detected at a concentration of $26 \mu\text{g kg}^{-1}$. In most cases these compounds were also detected in mill effluent; in a few cases they were also detected in upstream water samples and in fish taken upstream of the mill.
4. Fish captured upstream and downstream of the mill contained concentrations of polychlorinated biphenyls in intestinal fat (but not muscle tissue) at concentrations ranging from trace (detected but not quantifiable) to $5 \mu\text{g kg}^{-1}$. Intestinal fat samples of fish captured downstream of the mill also contained similar concentrations of DDT and DDE. These compounds were not detected in mill effluent and the concentrations detected in fish tissues were well below those currently accepted as safe for human consumption.
5. Histopathological examination of tissues of small fish resident downstream of the discharge point revealed a slight increase in observable lipid in the livers. This finding suggests a mild unspecified impairment of lipid metabolism and possibly of protein synthesis. The effect of this change on fish production in the river is not known.

CONTRIBUTED PAPERS

**Mitigating the Loss of Arctic Grayling (*Thymallus arcticus*)
Spawning Habitat in Southern Montana**

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ABSTRACT

I am reporting about the well-being of an Arctic grayling (*Thymallus arcticus*) population from southern Montana. The grayling population in question was cohabiting a reservoir with cutthroat trout (*Oncorhynchus clarki*), brook trout (*Salvelinus fontinalis*) and cutthroat-rainbow trout hybrids. The dam at the outlet of the reservoir had been classified as unsafe by the U.S. Corp of Engineers. The cost of rehabilitating the dam, which is owned by the State of Montana and administered by the Montana Department of Natural Resources and Conservation, had to be recovered by increasing storage. In Montana, water is sold by the gallon as opposed to being allocated by water licenses as is the practice in Alberta. The cost of rehabilitating the dam was therefore placed squarely on the shoulders of the water users.

The simple solution of increasing storage caused some serious concern. It was the opinion of the Montana Department of Fish and Game that approximately 80% of the Arctic grayling spawning habitat in Middle Creek, which is the spawning stream for this species, would be inundated by increasing the full supply level of the reservoir. Arctic grayling is considered a Class A Montana Fish of Special Concern, and has been nominated as a rare and endangered species in the state. The loss of spawning habitat was therefore considered an unacceptable impact of reservoir rehabilitation. In the spring of 1986, we carried out an investigation which confirmed the majority of Arctic grayling spawning habitat would in fact be lost if the reservoir level was raised (Fernet and Hunter, 1986). In analyzing the situation, the mitigative solution of creating spawning habitat was offered. However, there had been several bad experiences in Montana with habitat creation and/or enhancement activities. This together with the special status of Arctic grayling led to a cool reception for the proposed mitigative option.

We were therefore directed to create artificial spawning habitat for the Arctic grayling in Middle Creek, and in addition to documenting use of this habitat, we were requested to document that recruitment from these spawning areas at least equalled that of the natural spawning areas.

To this end, spawning habitat utilization criteria for the Arctic grayling in Middle Creek were developed from measurements of depth, velocity and substrate recorded at the spawning site used in 1986. A location for experimental enhancement was selected which was near the creek mouth, so we would be sure the fish at least considered the habitat. We were concerned that if the habitat was created upstream of the existing spawning areas, the spawners would not ascend to the newly created habitat. Virtually all the lower region of the creek has gravels of suitable size for spawning, so substrate was not an issue. We did, however, identify an area where there was virtually no spawning in 1986.

Rather than designing the habitat enhancement structure, we provided depth and velocity criteria to the engineering company, HKM Associates of Billings, Montana, who were charged with developing the instream mitigation. In the region of the creek which was chosen for the enhancement, the creek was too shallow and fast for spawning use by Arctic grayling. The relatively simple solution the engineering company employed to meet our depth and velocity criteria was the installation of a sill, in the form of a downstream oriented "V". This was a practical solution as all the mitigation had to be done by hand, with materials which were readily available due to the poor access to this region of the creek.

During the spring of 1987, the performance of this structure was evaluated (Fernet and Hunter, 1987). We had several concerns, one of which was the very low flow which occurred in the creek. As a result of the low stage, the depths and velocities created upstream of the structure did not meet our criteria, but were fairly close. We were also quite concerned about the sediment accumulation which occurred upstream of the weir. It was unfortunate that the "V" was not oriented in the upstream direction.

Spawning did occur in this habitat in 1987, and because it was such a low runoff year, the majority of spawning in Middle Creek actually occurred in the enhancement area. Our concern was then the issue of the sediment in the structure, and whether this would result in adverse effects on recruitment.

Two types of traps were employed to evaluate recruitment from the structure for comparison with that which occurred at the natural spawning areas. Emergence traps were

installed over seven spawning sites, two of which were situated in the newly created spawning habitat. These traps were cleared twice a day over a ten-day period in mid July (07-17 July). In addition to the emergence traps, two drift nets were installed in the stream. One of these traps was installed at a location which was used in 1975, using traps of the same design as those employed in 1975 (Wells, 1976), to evaluate any changes in overall recruitment which may have resulted from use of the structure. The greatest number of fry (693) was collected from one of the two emergence traps installed in the enhancement structure. A total of 801 fry were collected from the two emergence traps in the newly created spawning habitat, while only 387 were taken from the other five emergence traps located in natural spawning areas. In the drift net, a total of 1,054 fry were taken, which resulted in an average of 95 fry d⁻¹ over the 11-d sampling period. This compared favourably with the average catch per day of 109 fry recorded in 1975 (Wells, 1976). It is also worth noting that the emergence traps resulted in 15% mortality, compared to 94% mortality recorded in the drift nets. Upon emergence, Arctic grayling from this population exhibit the behaviour which is the norm for the species, which is passive drift.

We therefore demonstrated that Arctic grayling spawning habitat could be created, and successfully used. The sediment accumulation in the spawning area did not apparently have serious negative effects on recruitment. Permanent spawning habitat will ultimately be created for this Arctic grayling population, in the form of a side-channel for spawning, with an upstream headgate to control flows in the channel.

REFERENCES

- Fernet, D.A., and C. Hunter. 1986. Hyalite Reservoir Arctic grayling study, 1986. Prepared by Environmental Management Associates, Calgary, AB and OEA Research, Helena, MT. Prepared for HKM Associates, Billings, MT. 44 pp.
- Fernet, D.A., and C. Hunter. 1987. Hyalite Reservoir Arctic grayling study, 1987. Prepared by Environmental Management Associates, Calgary, AB and OEA Research, Helena, MT. Prepared for HKM Associates, Billings, MT. 35 pp.
- Wells, J.D. 1976. The fishery of Hyalite Reservoir in 1974 and 1975. M.Sc. Thesis, Montana State University, Bozeman, MT. 57 pp.

Effects of Glyphosate on the Water Chemistry and Plant Productivity of Boreal Forest Ponds

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ABSTRACT

In 1984, the Province of Manitoba entered into an agreement with the federal government which mandated the replanting of clearcut forested areas. As a result, forestry companies involved in clearcut operations began evaluating various means to remove silviculturally undesirable vegetation from these areas so as to reduce competition to newly planted black spruce and jack pine seedlings. One treatment involved the application of the broad spectrum herbicide Roundup. This herbicide can be applied aerially and is reportedly phytotoxic to most terrestrial plant groups; however there are relatively few data on its effects upon nontarget organisms. Consequently, one aspect of the agreement sought to investigate possible deleterious effects on aquatic ecosystems as a result of chance contamination by airborne herbicide residues.

There were two basic objectives for this four year study which commenced in 1985. The first objective was to select a series of small ponds for manipulative studies, and to characterize baseline physical, chemical, and botanical (phytoplankton, periphyton, macrophytes) differences between them over a period of one and a half years (1985-1986). Following this characterization, the second objective was to select ponds to be oversprayed with Roundup (active ingredient glyphosate) and to monitor water chemistry, aquatic plant productivity and standing crop in treated and untreated ponds for two and a half years (1986-1988). A study area was selected near the western limit of the Canadian Precambrian Shield, north of the confluence of the Winnipeg River with Lake Winnipeg. Six small ponds within this study area (named Spruce, Raspberry, Pine, Golden, Hike and Birch for convenience) were chosen in 1985, and two additional ponds (Tamarack and Alder) were added in 1987. The pond basins were excavated in the late 1950s during the construction of a nearby highway. Since then, all have become filled with water and colonized by submerged and emergent aquatic plants. Native forest encroaches on the shores of several of the ponds. The mean depths of the ponds vary between 0.9 and 1.5 m, with maximum depths in the range of 2.9 to 4.2 m. All ponds are <1 ha in area. None

of the ponds have any permanent overland source of inflow or outflow, although they receive periodic recharge from inflow following rainfall.

Several physical, chemical and biological parameters were monitored in each of the study ponds at weekly or biweekly intervals throughout the ice-free seasons of 1985 and 1986. In late August, 1986, three ponds (Spruce, Hike and Birch) were sprayed at a rate of 2.5 L Roundup ha⁻¹ (0.9 kg glyphosate ha⁻¹). A second 6.0 L ha⁻¹ Roundup treatment was applied to Spruce and Hike ponds in late August, 1987. Birch pond was not resprayed in 1987 but Tamarack pond, which had not been treated in 1986, was sprayed with 6.0 L ha⁻¹ Roundup in 1987. None of the remaining four ponds were sprayed, and water samples collected from these four ponds immediately after treatment showed no evidence of cross-contamination by Roundup.

No evidence of immediate phytotoxic effects on terrestrial plants around the periphery of treated ponds was found after the 1986 and 1987 treatments. The range of effects observed in the spring of the year following each treatment appeared to vary with the position of the plants in relation to the bearing of the spray aircraft, and the type of plant. Few trees were killed as a result of either treatment.

Water samples were collected from the treated ponds for herbicide residue analysis immediately after application and at regular intervals for several weeks afterwards. Glyphosate concentrations in those ponds sprayed in 1986 decreased according to first order kinetics with half-lives of between 3 and 5 d.

Seasonal means of total Kjeldahl nitrogen and total phosphorus in water collected during the ice-free season from the study ponds showed no apparent change as a result of herbicide treatments in 1986 or 1987. There was, however, marked inter-pond variation in both parameters, which substantiated the need for background characterization prior to any herbicide treatment. Seasonal variation in total nitrogen and phosphorus within given ponds was also marked, particularly for Birch, Raspberry, Pine and Golden ponds in 1988. These ponds experienced severe drawdown due to extended periods of low precipitation.

Mean seasonal phytoplankton biomass, as indicated by total chlorophyll *a* concentration, did not appear to change significantly in treated ponds following treatment although there was marked variation both between individual ponds, and between years from a given pond. Variation between ponds, however, appeared to correlate with the total phosphorus concentration of their respective waters.

Mean seasonal periphyton biomass, as indicated by total chlorophyll *a* concentration, did not appear to decrease significantly in treated ponds. In 1987 for Spruce and Hike ponds, and in 1988 for Hike pond, there was an increase in overall periphyton biomass. Data plotted seasonally for these ponds, indicated that periphyton biomass increased in the spring of the year following the 2.5 L ha⁻¹ treatment of both ponds, and was slightly higher in Hike pond in 1988 following the 6.0 L ha⁻¹ treatment. However, in the latter case, periphyton biomass in 1987 decreased immediately after the 6.0 L ha⁻¹ treatment.

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Sludge Capping for the Disposal of Oil Sands Sludge

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INTRODUCTION

The Syncrude Canada Ltd. (SCL) oil sands plant near Fort McMurray, Alberta, produces large quantities of fluid wastes (water and sludge) during the production of synthetic crude oil. These wastes are currently held in a large pond being developed on the northern portion of the lease (Fig. 1). The tailings pond is stratified by density into a low solids surface zone overlying a high solids sludge zone (Fig. 2). The tailings pond operates both as a fluid waste storage area and a settling basin for the large volumes of tailings slurry that are input to the pond. An important aspect of pond operation is the reuse of water released during sludge densification. In 1988 over 70% of the water needs of the plant were met from this recycled water.

SCL is considering various options for the ultimate reclamation of both the water and the sludge currently being stored within the pond. One such option being evaluated by Syncrude is disposal of the mature sludge in the mined-out pit. A capping layer of clean water will then be added to allow the establishment of a viable biological community. Because of the rheological properties of the sludge and the large density difference between the two layers, the amount of resuspension of the sludge into the water column will not be sufficient to be detrimental to the maintenance of a stable, productive ecosystem.

Sludge capping is analogous to land reclamation where barren overburden or tailings sand is capped with a sufficient layer of productive soils to establish a self-sustaining terrestrial community.

Though much work remains to be done, research to date continues to support the concept of capping sludge with a water layer. Under the present plans, the primary source of the capping water will be natural surface water drawn from outside the tailings pond system. It is expected that the capping zone will evolve into an aquatic system with a biological productivity similar to other natural waterbodies in the Fort McMurray region. The rate of this development, its

Figure 1. Location of Syncrude site on Leases 17 and 22.

Figure 2. Schematic representation of cross-section of tailings pond (1988). Vertical exaggeration is about 80:1.

productivity and the stability of the resulting aquatic habitat need to be addressed to demonstrate the feasibility of this concept.

BACKGROUND

Syncrude Canada Ltd. has operated its Mildred Lake oil sands project in northeastern Alberta since 1978. Since start-up more than $60 \times 10^6 \text{ m}^3$ of synthetic crude oil have been produced from the processing of over $800 \times 10^6 \text{ t}$ of oil sand. Large quantities of water and sludge are generated during the production of synthetic crude oil. Under Syncrude's zero discharge policy, no process-affected water is released to the surrounding environment. Tailings, resulting from the extraction of the oil sand, are transported as a slurry of water, solids, and unrecovered bitumen (55% solids, 0.5% bitumen) to a large containment pond to the north of the plant site (Fig. 1). Most of the coarse solids are deposited in cells and on beaches which form the dykes surrounding the pond. The remaining water, solids, and bitumen runoff into the pond. About 6 - 8% of the solids in the oil sand enter the pond and become incorporated in the sludge where settling and consolidation proceeds (Fig. 2). The rates of densification of the sludge are slow, with full consolidation likely requiring many decades (Devenny, 1975; Yong *et al.*, 1983; Scott *et al.*, 1985).

By 1988 there were over $230 \times 10^6 \text{ m}^3$ of fluid wastes within the pond. The sludge zone accounted for approximately $130 \times 10^6 \text{ m}^3$ of total pond volume. Sludge volumes are growing at a rate of about 0.15 to $0.20 \text{ m}^3 \text{ t}^{-1}$ of oil-sand feed.

Treatment of low solids water in the surface zone of the tailings pond to an acceptable quality for other uses has been reported (MacKinnon and Retallack, 1982; MacKinnon and Boerger, 1986). However, effective methods for the reclamation of the sludge layer are still being developed. Syncrude is pursuing two approaches for reclamation of tailings sludge: one involves treatment for disposal as a dry landscape and the other will lead to disposal within an aquatic habitat. At present, no known technology exists which would allow Syncrude to use a dry disposal scheme as a basis for planning. While Syncrude's plans are currently based upon "wet" sludge disposal, research on "dry" sludge disposal continues. Some of the dry disposal options being examined include: chemical treatment, sludge capture with sand or overburden solids, and desiccation by evaporation, evapotranspiration or freeze-thaw.

In the "wet" disposal options, consideration is being given to the feasibility of management of the sludge by capping it with a water layer in which an aquatic ecosystem will evolve and be maintained. At this stage in the development of the sludge capping plan, research has been directed at the general aspects of the concept. Ultimate acceptance of the sludge capping concept as a viable fluid waste disposal method will require more research and demonstration of larger-scale viability. Prior to proceeding with detailed research, some basic questions must first be addressed. The factors potentially influencing sludge capping are outlined in Fig. 3 and include:

- type of capping water
- depth of capping layer
- effect of released water on capping layer quality
- potential for mixing of layers
- hydrologic influences
- role of chemical and biological factors in maintaining a stable ecosystem
- littoral zone development

This paper will focus upon some of Syncrude's progress to date in development of the option for disposal of sludge as part of an aquatic habitat.

PHYSICAL ASPECTS

The physical makeup of the final capped sludge pond will have a direct influence on the biological colonization of the pond and its ultimate development as an aesthetic and viable lake community. Beginning in the late 1990's, Syncrude plans to begin pumping mature sludge from the tailings pond into the mined-out pit. Pumping is expected to continue for approximately 20 years. The resulting pond will have a depth of about 60 m of sludge, an area of 5 to 6 km² and will be approximately 3 km long from north to south.

Wind Generated Mixing

Ideally, there should be no active mixing between the surface water zone and the sludge which it is capping. It is assumed that because of density differences between the layers and the rheological properties of mature sludge (shear strength, and viscosity), the interface between the zones will act as an effective barrier which will prevent sludge solids from becoming resuspended in the overlying water column. The main potential source of energy which will cause mixing of the sludge into the surface layer will be wind generated currents or waves. The degree of

FACTORS POTENTIALLY AFFECTING SLUDGE CAPPING

Figure 3. Factors which may influence the concept of capping oil sand sludge with a shallow lake system.

turbulence is a function of the cap water depth, wind speed, fetch (distance of wind across surface), threshold bed velocity (minimum velocity of water required to disturb the sediment surface), and wave height and period. The projected water cap depths, as functions of fetch and threshold velocity at the sludge interface, are plotted for a major storm event (Fig. 4a and 4b).

Preliminary modelling experiments have been carried out using flume tests to assess the impact of surface layer agitation caused by wind and current flows. Results from these studies have shown that the bed velocities, which are produced by the wind generated wave action, have more erosional power than similar velocities produced under current flows. It was found that a threshold velocity of about 0.04 m s^{-1} at the sludge interface produced by waves, caused resuspension of the sludge resulting in water turbidities above the sludge zone of 50 to 100 NTU (less than 100 mg L^{-1}).

Critical water depths required to prevent sludge suspension under various wind storm events during the open water period are indicated in Table 1. The average annual storm events are based on actual Environment Canada wind data collected at the Fort McMurray Airport between 1978 and 1987. The predicted capping depths have been calculated using threshold velocity (U_t) of 0.04 m s^{-1} (determined using flume tests) and a fetch of 4 km (Fig. 5). Since the proposed in-pit pond will have a fetch of less than 4 km, the predicted cap water depth can be considered a worst case or conservatively high estimate. Changes in fetch have a direct influence on the amount of energy imparted to the sludge layer by surface winds. With a lower fetch of 3 km the depth of a capping layer required to isolate the sludge for a 10-year storm event (i.e. 15 m s^{-1}) will be reduced by about 20%.

Based on the laboratory flume tests and historical data on wind velocity a capping layer of clean water of as much as 5 m is expected to effectively isolate the sludge layer from active mixing in all but the most severe wind conditions. Further field and modelling tests are expected to reduce significantly the required depth of the capping layer and define more adequately the resuspension and resettling processes.

Figure 4a. Depth of water cap as a function of fetch.

Figure 4b. Depth of water cap as a function of the threshold bed velocity (U_t).

MINIMUM DEPTH AS A FUNCTION OF WIND VELOCITY

Figure 5. Depth of water cap as a function of wind velocity.

Table 1. Predicted water layer depths above which significant sludge suspension will not occur based on flume tests (sludge suspension threshold velocity of 0.04 m s^{-1} and a fetch of 4 km).

	Average Wind Speed of 4-hour storm events (m s^{-1})	Predicted Depth of Water Cap Layer to Prevent Sludge Suspension*
1-year storm**	11.9	3.8
10-year storm***	15.3	4.8
20-year storm***	16.4	5.1
100-year storm**	18.3	5.5

* Sludge of solids content greater than 25%.

** Average of annual storms between 1978 to 1987.

***Predicted probability of storm events based on observational data between 1978 to 1987.

Shoreline Erosion

Slopes of dykes surrounding the sludge storage pond are presently projected to be between 8 and 10%. Wind generated waves could promote erosion of these dykes leading to an increase in turbidity of the surface water. Terrestrial reclamation of the exposed sand and overburden dykes is scheduled as part of Syncrude's ongoing reclamation of disturbed areas. A similar reclamation program will be required for the beaches of the in-pit sludge pond area. This will entail the amendment of dyke slopes similar to existing activities and this will assist in the development of littoral zone and shoreline vegetation. Research into methods of improving shoreline and littoral zone stability and development will proceed as Syncrude carries out larger scale field tests of the feasibility of the sludge capping concept.

Seasonal Effects

There is a potential for mixing of the two layers during spring and fall periods when the surface cap zone of the in-pit pond will become isothermal. However, due to the large density differences between the high solids sludge and lower solids surface water, the possibility of mixing during the seasonal turnover periods is small. At present no disturbance of the sludge has been observed in the existing tailings pond during such periods.

Hydrologic Cycle

Water depth of the capping zone must be maintained above a critical level to minimize wind generated mixing (Table 1). In the Syncrude area, there is a slight net evaporation (about 5 - 10 cm yr⁻¹). As a result, some system to stabilize water depths may be required until a balance between the precipitation collected by the catchment area and losses is obtained. Over time, the sludge zone in the pond will continue to densify although the rate is expected to be slow (estimated at 10 cm yr⁻¹). As a result, a constant release of water is expected. As a worst case, positive replenishment of the capping layer from an external source may be necessary for the early period of evolution of a lake environment.

CHEMICAL ASPECTS

The sludge that will be transferred from the tailings pond into the mined-out pit is classified as "mature" sludge (about 35% solids by weight). Once in pit, the mature sludge will continue to consolidate slowly. Prior to capping with water, the transferred sludge will have as much as 20 years to age and densify. As the sludge densification processes continue, there will be a release of the sludge pore water into the overlying surface water. The rate at which the water is released will be slow and will be accompanied by a corresponding drop in the level of the sludge interface. The clean water overlying the sludge in the final capped sludge pond could be influenced by the upward migration of interstitial sludge water during consolidation.

Effect of Released Water on Sludge

The properties of the water in the present tailings pond are a good indication of the quality to be expected of released water. In Table 2 the composition of the surface zone and sludge zone waters are compared. Some minor differences between the two water types are seen. Tailings pond waters are acutely toxic to aquatic organisms, with most of the toxic character being related to polar organic leachates resulting from the oil sands processing (MacKinnon and Boerger, 1986). Degradation of potential toxicants, once expressed into the surface water, is expected to occur fairly quickly through processes of oxidation, hydrolysis, biodegradation, and volatilization. This natural reduction of toxicity of Syncrude's tailings pond water has been demonstrated in experimental pits where tailings pond water has been monitored for periods greater than 12 months (Boerger and Aleksuk, 1987). This process can be accelerated by

Table 2. Composition of tailings pond water from the surface zone and mature sludge zone, 1988. All concentrations are in mg L⁻¹ unless shown otherwise.

Variable	Surface Zone (0 - 8 m)	Mature Sludge Zone (17 - 23 m)
pH	7.8 - 8.1	8.3
Conductivity ($\mu\text{S cm}^{-1}$)	1750 - 1900	1150 - 1250
Suspended Solids (%)	0.03 - 0.07	27 - 35
Dissolved Solids	1300 - 1500	1200 - 1300
Dissolved Organic Carbon (mg C L ⁻¹)	40 - 60	50 - 70
Dissolved Oxygen	2 - 4	<0.2
COD	250 - 300	300 - 600
BOD	20 - 50	50 - 200
Phenols	0.05 - 0.15	0.05 - 0.10
Oil and Grease	10 - 40	15000 - 40000
Sulphide	<0.01	<0.01
Cyanide	0.2 - 0.3	0.005 - 0.010
Redox Potential (mV)	+50 - +100	-150 - -300
Surfactants	1.5 - 3.0	1.5 - 3.0
Tannin and Lignin	2 - 4	4 - 8
Major Ions		
<u>Cations</u>		
Sodium	440 - 460	400 - 440
Potassium	10 - 15	10 - 15
Magnesium	5 - 10	2 - 6
Calcium	5 - 10	5 - 10
<u>Anions</u>		
Fluoride	<1	<1
Chloride	135 - 145	130 - 140
Sulphate	200 - 250	<10
Bicarbonate	700 - 750	900 - 1000
Alkalinity	550 - 600	750 - 850

Variable	Surface Zone (0 - 8 m)	Mature Sludge Zone (17 - 23 m)
Hardness	40 - 50	25 - 35
<u>Nutrients</u>		
Nitrite + Nitrate (mg N L ⁻¹)	<0.05	<0.02
Ammonia (mg N L ⁻¹)	3 - 5	3 - 6
o-Phosphate (mg P L ⁻¹)	<0.03	<0.3
<u>Acute Toxicity</u>		
Trout LC ₅₀ (%)	<10	-
Microtox EC ₅₀ (%)	25 - 40	25 - 40
<u>Minor Elements</u>		
Al	0.10	0.91
Sb	0.0007	0.0005
As	0.0080	0.0360
Ba	0.07	0.08
Be	0.002	0.001
B	1.5 - 3.0	2.0 - 3.0
Cd	<0.001	<0.001
Cr	<0.001	0.003
Cr ⁵⁺	<0.003	<0.003
Co	0.005	0.008
Cu	<0.001	0.005
Fe	0.17	0.82
Pb	0.015	0.020
Mn	0.035	0.009
Hg	<0.0005	<0.0005
Mo	0.120	0.063
Ni	0.020	0.015
Se	0.013	<0.0002
Si	9.90	13.80

Variable	Surface Zone (0 - 8 m)	Mature Sludge Zone (17 - 23 m)
Ag	0.002	0.002
Sr	0.25	0.17
Sn	<0.01	<0.01
V	0.003	0.019
Zn	1.15	0.025

aeration. In Fig. 6, the acute toxic character of various sludge capping water is plotted. The initial acute toxicity of the tailings pond water (EC_{50} 30%) was removed in four months under an aerobic environment (resulting $EC_{50} > 100\%$).

Currently, experiments are under way where water layers are maintained over sludge in a series of indoor chambers (0.3 m x 0.3 m x 2.5 m). Preliminary results indicate changes in the chemistry of overlying water are primarily the result of the more soluble and conservative species (Table 3). As shown in Table 2, the trace metal concentrations in the oil sand waste water are low. Qualitative analysis of dissolved organics in tailings water have found no detectable levels of priority pollutants. As a result, the water quality of the capping layer has remained high. Increases in sodium as well as chloride and bicarbonate have been monitored over a 12-month period (Fig. 7a and 7b).

The changes in chemistry are occurring slowly as a result of the gradual release of sludge interstitial water. Generally, no negative effects of this released water on a biological community have been observed. In the Mildred Lake cap water experiment, acute toxicity values (Microtox EC_{50}) have remained greater than 100% (Table 3). This indicates that no significant increase in toxicity is occurring due to effects of the water released from the sludge zone (Fig. 8). Decreases in toxicity were also noted in other tests involving a tailings water cap over the sludge layer (Fig. 6). While trends showed increasing levels of the more conservative ions in these experiments, no obvious increase was noted in dissolved organic carbon levels (Fig. 8) nor in levels of other potential materials of concern such as cyanide or phenol.

Figure 6. Change in acute toxicity (Microtox EC₅₀) measured in various sludge capping waters. Ratio of initial cap water volume to sludge volume = 4:6.

- Star = Tailings Surface Pond Water (aerated)
- Triangle = Mildred Lake Water (aerated)
- Circle = Mildred Lake Water (no aeration)
- Square = Mildred Lake Water with Sediment Added (aerated)

Table 3. Composition of sludge and water used to cap the sludge. Changes with time are shown in the composition of the capping water maintained under aerobic conditions by aeration over a 12-month period.

Variable	Sludge	Mildred Lake Water (Capping)*			
	T ₀	T ₀	T _{1 month}	T _{6 months}	T _{12 months}
pH	8.3	7.9	8.4	8.4	8.5
Conductivity ($\mu\text{S cm}^{-1}$)	1500	285	340	575	685
Dissolved Oxygen (mg L^{-1})	<0.2	9	8	6	5
Dissolved Solids (mg L^{-1})	1400	250	250	370	405
Suspended Solids (mg L^{-1})	30000	29	10	<10	10
Dissolved Organic Carbon (mg C L^{-1})	64	25	16	26	16
Phenols (mg L^{-1})	0.065	0.003	0.006	0.004	0.003
Cyanide (mg L^{-1})	0.006	0.001	0.001	0.001	0.001
Chemical Oxygen Demand (mg L^{-1})	>200	20	48	58	55
Release Water from Sludge** (% of Cap Zone)	-	-	1	20	29
Major Ions (mg L^{-1})					
<u>Cations</u>					
Sodium	450	15	40	106	129
Potassium	11	1.3	2.2	3.4	5.0
Magnesium	4.0	10	9.6	8.8	9.2
Calcium	4.0	38	34	20	16
<u>Anions</u>					
Chloride	130	7.8	13.2	33.6	41.5
Sulphate	4	23	22	22	21
Bicarbonate	940	160	198	280	270
Hardness (mg of CaCO_3)	27	135	125	87	78
Ratio Na/Cl (meq/meq)	5.3	3.0	4.7	4.9	4.8
<u>Nutrients</u>					
$\text{NO}_2^- + \text{NO}_3^-$	0.04	0.03	0.02	<0.01	0.31
Ammonia	3.8	0.01	0.20	<0.01	0.02
O- PO_4	0.14	0.03	0.02	0.02	0.02

Variable	Sludge	Mildred Lake Water (Capping)*			
	T ₀	T ₀	T _{1 month}	T _{6 months}	T _{12 months}
<u>Acute Toxicity</u>					
Microtox					
EC ₅₀	30	100	100	100	100
EC ₂₀	10	100	100	100	100
<u>Minor Elements</u> (mg L ⁻¹)					
As	0.0079	0.0010	0.0018	0.0020	0.0010
B	2.45	0.06	0.16	0.43	-
Cd	<0.001	<0.001	<0.001	0.002	-
Cr	0.016	0.003	0.012	0.007	-
Co	<0.001	<0.001	0.013	<0.001	-
Cu	0.013	0.012	0.003	0.003	-
Pb	<0.002	<0.002	0.015	0.015	-
Mo	0.058	<0.001	0.017	0.009	-
Ni	0.013	<0.001	0.048	<0.001	-
Sr	0.21	0.23	0.24	0.24	-
V	0.023	0.006	0.002	0.001	-

* Ratio of capping layer to sludge layer = 2:3. Capping layer = Mildred Lake Water collected September, 1987.

**Volume of release water from sludge as a percent of original volume of capping layer.

a) Cations

Figure 7a. Cations. Change with time of major ion content (meq L⁻¹) in Mildred Lake water (aerated) used in capping mature sludge. Ratio of initial cap water volume to sludge volume = 4:6.

b) Anions

Figure 7b. Anions. Change with time of major ion content (meq L⁻¹) in Mildred Lake water (aerated) used in capping mature sludge. Ratio of initial cap water volume to sludge volume = 4:6.

Nutrients

The transport of nutrients into the surface zone would be expected to occur along with other compounds during the release of sludge water. However, the levels of phosphate and oxygenated nitrogen forms are low in the sludge waters (Table 2). High levels of ammonia are found in the sludge water. As a result, some nitrogen in the form of ammonia is input to the capping water where it is oxidized (Table 2). The phosphate levels are low in the sludge and the system can be considered to be phosphate limited.

One strategy for improving the nutrient regime would involve the redirection of natural surface drainage into the surface water of the capped pond. This would provide a continual input of nutrients and thereby provide significant benefit to the aquatic regime. An inflow program may be required during the initial stage of the development to maintain water levels. By controlling the input of nutrients into the sludge cap lake, the eventual productivity of the ecosystem can be established.

Dissolved Oxygen

Dissolved oxygen concentrations in the surface layer of the capped pond are likely to be maintained high enough by wind mixing to sustain aerobic conditions during the open water season. However, the water released from the sludge has a high chemical oxygen demand. During the 12-month experimental period, a slight increase in the COD of the capping layer has been found (Fig. 8). During winter, when the water is not subject to wind induced mixing, chemical and biological oxygen demands are expected to produce deficient conditions similar to those found in many of the regional lakes. The seasonal variation of oxygen levels within the surface water is part of the natural regional characteristic and is not seen as a serious limitation to the success of the sludge capping concept.

BIOLOGICAL ASPECTS

At present, it is expected that most of the capping water for the proposed sludge pond will be drawn from natural waterbodies in the area. This water will be of high quality and biological nutrient content should be high. Migration rates of water and dissolved components from the sludge layer into the overlying surface layer are expected to be low. Biological colonization of the pond is expected to occur swiftly (Retallack *et al.*, 1981).

Figure 8. Changes with time of properties of Mildred Lake water (aerated) used in capping mature sludge. Ratio of initial cap water volume to sludge volume = 4:6.

Triangle = Acute toxicity (Microtox EC₅₀)
 Star = Dissolved Organic Carbon (mg C L⁻¹)
 Square = Chemical Oxygen Demand (mg L⁻¹)

Table 4. Toxicity values of low solids content in tailings pond water stored in pits (10 m x 10 m x 2 m) under natural conditions for 12 months (MacKinnon and Boerger, 1986).

Bioassay Test	Initial	After 12 Months
Microtox EC ₅₀ (%)	30%	100%
96-hour Trout Bioassay, LC ₅₀ (%)	10%	100%
<i>Daphnia magna</i> ; LC ₅₀ (%)	10%	100%

To date biological monitoring of the sludge cap water has been limited primarily to bench scale experiments in which clean water has been deposited over mature sludge. Information on biological quality and colonization of fresh and aged tailings pond water has been reported (Boerger *et al.*, 1986; MacKinnon and Boerger, 1986; Boerger and Aleksik, 1987). These positive results on the development of biological communities in tailings pond water support the proposed sludge capping project since such water quality represents a worst case estimate of what can be expected in the final capped pond. The cap water quality is predicted to be of better quality than present tailings pond water. As a result, biological colonization should proceed at a rate higher than reported in these earlier studies (Boerger *et al.*, 1988).

Syncrude has demonstrated that tailings pond water exhibits an acute toxicity of about 30% EC₅₀ (MacKinnon and Boerger, 1986). The biological community of the tailings pond is limited to microbial populations (Foght *et al.*, 1985). Detoxification of the pond water has been shown to occur naturally over time (Boerger and Aleksik, 1987). In pits where the active input of tailings material has ceased, a marked increase in biological activity has occurred. In experimental pits, Syncrude Tailings Pond water was monitored in situ for a 12-month period. During this time a reduction of toxicity values was evident (Table 4). This is supported by the observations reported in Fig. 6.

Suncor Inc. has carried out research using larger pits supplemented by nutrient additions. Suncor noted substantial increases in phytoplankton and zooplankton populations after the addition of nutrients (Nix and Martin, 1985). In addition, they noted decreases in acute and chronic toxicity, and marked increases in the growth of pond vegetation. Research by the Alberta Environment Centre in Vegreville shows that tailings sludge and tailings pond water are apparently not phytotoxic (R. Johnson, pers. comm.). In fact, aquatic macrophyte communities have developed in aquatic systems on the Syncrude lease which receive process water inputs by way of the seepage control system. Plants observed include *Carex*, *Sparganium*, *Betula*, *Populus* and others. In situ bioassays using *Lemna minor* have shown no acute toxic effects. The absence of such plants in active tailings ponds is not considered to be due to the toxic nature of the water but rather the presence of bitumen globules, hydrocarbons or other physical factors associated with an active pond area.

Experiments have shown aerobic bacterial degradation to be the driving force behind detoxification of low solids tailings pond water (Boerger and Aleksik, 1987; MacKinnon and

Boerger, 1986). Biological activity in the sludge appears to be limited to anaerobic microbiota such as sulphate reducing bacteria (Foght *et al.*, 1985). While methanogenic bacteria have been identified in sludge, the conditions (i.e. redox potential) of the tailings pond are not conducive to that methane generation in Syncrude's sludge and it is not considered a problem.

Experiments currently underway involve bench scale representations of the water over sludge system. These experiments are designed to assess sludge consolidation, biological colonization, water quality trends and aeration effects. Larger scale field tests are planned which will more accurately model the final capped sludge pond. These will improve our knowledge of the sludge capping system.

CONCLUSIONS

Evidence to date supports the feasibility of the sludge capping option. The resulting waterbody would have an acceptable water quality and in time would result in a self sustaining waterbody, similar to other lakes in the region.

The in-pit pond, as currently designed, will consist of a surface layer of clean water overlying a thick layer of sludge. The overall final depth of the capped sludge pond is expected to be about 60 m. Surface water depths adequate to mitigate the effect of wind initiated resuspension of solids will be required. Preliminary research based on worst case conditions indicate that as much as 5 m of capping water should be sufficient to isolate the sludge from wind generated mixing in a 100-year storm event.

Erosion of shorelines may occur initially due to the anticipated steep and exposed shore. As shorelines stabilize and vegetative cover develops, erosion can be expected to decrease.

The movement of interstitial water from the sludge to the surface water layers will proceed through consolidation and diffusion processes. However, due to the expected low rates of this migration, biological degradation of these water should mitigate any toxic effects in the water column.

Release water is anoxic and has high chemical oxygen demand. Because of mixing, high dissolved oxygen levels in the surface water will be maintained during open water periods. During ice cover, oxygen levels will likely decline to very low levels. Anaerobic conditions during winter may restrict the ultimate establishment of populations of some aquatic species. This is consistent with the situation in many other lakes in the region.

FUTURE WORK

Syncrude has identified several questions concerning the concept of sludge capping which require further research. A program has been established to continue addressing these concerns and demonstrate the viability of the concept. Research planned for the future includes the construction of larger test pits which are intended to reflect better the scale and design of the proposed sludge capping facility. These pits will provide further information on wind and wave action as it relates to sludge resuspension and shoreline erosion. Chemical and nutrient mobility between sludge and surface water is to be examined. As well, overall biological activity and the rates of natural and induced colonization processes will be evaluated.

Syncrude plans to continue its work in the investigation of a sludge capping reclamation strategy.

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REFERENCES

- Boerger, H., and M. Aleksuik. 1987. Natural detoxification and colonization of oil sands tailings water in experimental pits. In: Proceedings of the Symposium of Oil Pollution in Fresh Water. Edmonton, AB.
- Boerger, H., M. MacKinnon, and M. Aleksuik. 1986. Use of bioassay techniques to evaluate the effectiveness of natural and chemical detoxification of tar sands tailings waters. In: Proceedings of the 11th Annual Aquatic Toxicity Workshop, Vancouver, BC. November 13-15, 1984. Can. Tech. Rep. Fish. Aq. Sci. No. 1480.
- Devenny, D.W. 1975. Subsidence problems associated with reclamation at oil sands mines. Canadian Rock Symp. September 2-4, 1975. pp. 161-179.

- Foght, J.M., P.M. Fedorak, D.W.S. Westlake, and H.J. Boerger. 1985. Microbial content and metabolic activities in the Syncrude tailings pond. *AOSTRA J. Res.* 1(3):139-146.
- MacKinnon, M.D., and H. Boerger. 1986. Description of two treatment methods for detoxifying oil sands tailings pond water. *Water Poll. Res. J. Can.* 21(4):496-512.
- MacKinnon, M.D., and J.T. Retallack. 1982. Preliminary characterization and detoxification of tailings pond water at the Syncrude Canada Ltd. oil sands plant. pp. 185-210. In: *Land and Water Issues Related to Energy Development. Proceedings of the 4th Annual Meeting of the International Society of Petroleum Industry Biologists.* Rand, P.J. (Ed). Denver, CO. September 22-25, 1981. Arbor Science.
- Nix, P.G., and R.W. Martin. 1985. Biological detoxification of effluent water from a tar sands process. Canadian Patent 1-181-537 issued to Suncor Inc., Canada. 23 pp.
- Retallack, J.T., P.T.P. Tsui, and M. Aleksuk. 1981. Natural colonization of an artificial stream bed in the Athabasca oil sands area of Alberta, Canada. *Verh. Internat. Verein. Limnol.* 21:799-803.
- Scott, J.D., M.B. Dusseault, and W.D. Carrier. 1985. Behaviour of the clay/bitumen/water sludge system from an oil sands extraction plant. *App. Clay Science* 1:107.
- Yong, R.N., S.K.H. Siu, and D.E. Sheeran. 1983. On the stability and settling of suspended solids in settling ponds. Part I. Piece-wise linear consolidation analysis of sediment layer. *Can. Geotechnical J.* 20:817-826.

**Fisheries Habitat Mitigation for
the Oldman River Dam Project**

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ABSTRACT

The Government of Alberta has set an objective of "no net loss of recreational fishing opportunities as a result of the Oldman River Dam project". In May of 1988, a further commitment was made by the government to fund the fisheries mitigation plant to its completion, over an anticipated ten-year program.

Fish population and habitat inventory surveys were carried out upstream of the damsite for a total of 200 km of the mainstem of the Oldman River and its three main tributaries: the Oldman North Fork, Crowsnest and Castle Rivers. The inventory of physical habitat facilitated: (1) quantification of habitat losses for the different life stages of rainbow trout, brown trout and mountain whitefish; and (2) identification of potential reaches for habitat enhancement above reservoir full supply level (FSL).

The primary assumption of the upstream mitigation strategy is that the "no net loss" objective means chiefly the replacement, above reservoir FSL, of the high quality riverine habitat which will be lost to flooding. This is accomplished by upgrading habitat of a lower quality to a higher quality through the installation of instream features such as boulder gardens, flow deflectors, rock weirs and pools, and overhanging cover.

Aquatic Weed Control in Alberta Irrigation Canals Using the Triploid Grass Carp

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ABSTRACT

Almost since irrigation began in Alberta in the early 1900s, aquatic weed growth has impeded water flow causing serious delivery shortages during critical crop growing periods. As an alternative to the costly chemical and mechanical control of submersed aquatic macrophytes in canal systems, the sterile grass carp (*Ctenopharyngodon idella*) is a very good candidate biological control. The five-year research study, which began in 1988, will evaluate this fish species for its potential as a biological control and for possible environmental impacts.

Objectives of this study are to determine: (1) the triploid grass carp can function and survive in the canal systems, (2) potential cost and benefits, (3) sites suitable for this type of weed control, (4) possible fate scenarios for escaped fish, (5) a list of potential diseases and parasites associated with the fish, (6) requirements for private culture of this fish in the province, and (7) other benefits of the use of triploid grass carp to agriculture across Canada.

Triploid Grass Carp Maintenance and Chromosome Evaluation

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ABSTRACT

Grass carp (*Ctenopharyngodon idella*) are used in many parts of the world for control of aquatic weeds. Grass carp were first imported into Canada by the Alberta Environmental Centre in 1988 in support of the experimental evaluation study "Elimination of Aquatic Weeds in Irrigation Canals Using Triploid Grass Carp", undertaken by Alberta Agriculture, Irrigation Secretariat. The fish are maintained under quarantine conditions in the laboratory. Dietary and thermal conditions have been manipulated to induce adequate growth. The evaluation of bacterial and parasitic agents has produced negative findings to date. Sexually sterile triploid grass carp have 72 chromosomes, while the diploid grass carp, capable of reproduction, have 48 chromosomes. Two methods are being used to determine the diploid/triploid condition of the fish prior to release from the laboratory. One method involves the measuring of the nucleus volume of red blood cells, while the other procedure involves conducting actual chromosome counts.

Diet Overlap in Hasse Lake Fish

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ABSTRACT

There are three main species of fish found in Hasse Lake: rainbow trout (*Oncorhynchus mykiss*), threespine stickleback (*Gasterosteus aculeatus*) and brook stickleback (*Culea inconstans*). Competition between young-of-the-year (YOY) trout and the other fish in the lake has been suggested as a possible explanation for the low survivorship of the newly stocked trout. Stomach content analysis of the three species of fish was carried out to determine diet during the summer months. Discriminate analysis and diet overlap indices were calculated. There appears to be no important overlaps in diet between YOY trout and the other fish in the lake. However, the diet of the two sticklebacks are very similar. Factors such as predation and fishing pressure may be more important than competition for food in determining the survivorship of the newly stock trout.

Effect of Reservoir Operation on Downstream Water Temperatures and Dissolved Oxygen Levels in the Red Deer River¹

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ABSTRACT

Flow regulation of the Red Deer River became possible with construction of the Dickson Dam and creation of a reservoir (Gleniffer Lake) in 1983. In this study, the effects of reservoir operation on water temperatures in the Red Deer River immediately below the dam and dissolved oxygen levels in the downstream Red Deer River in Alberta were examined.

Bottom release from reservoirs with the potential to become thermally stratified, such as Gleniffer, can have a significant effect on water temperatures immediately below the dam. Downstream winter temperatures can be elevated and summer temperatures can be depressed. In addition, the seasonal temperature cycle can be affected. Spring warming and fall cooling can be delayed. The longitudinal extent of the reservoir effect on downstream temperatures depends on flow, weather conditions and channel morphology. These temperature alterations can have a substantial effect on downstream biota.

Gleniffer Lake becomes thermally stratified in mid to late summer and winter. Water is released from the bottom of Gleniffer Lake in the winter months through twin diversion tunnels. Discharge in the spring through fall occurs through the tunnels or from the sub-surface waters over the spillway. Annual spillway operation has varied from only six days in the fall of 1985 to 111 days in 1986. Post-impoundment winter (November to March) temperatures are elevated 1 to 3°C relative to pre-impoundment temperatures both immediately below the dam and at the Anthony Henday Water Treatment Plant (AHWTP) intake, 27 km downstream. Mean daily summer water temperatures immediately below the dam are lower by up to 6°C and much less variable on a daily basis than temperatures immediately upstream of Gleniffer Lake. Post-impoundment spring (April, May), summer (June, July, August) and fall (September, October) temperatures at the AHWTP intake are not significantly different from pre-impoundment temperatures.

¹Note: This paper was presented at the 1988 Applied Aquatic Studies Workshop

Historically, the main water quality problem in the Red Deer River has been the occurrence of depressed dissolved oxygen levels in winter. Dissolved oxygen levels began to decline downstream of the James River confluence and continued to decline towards the Saskatchewan border. The upgrading of sewage treatment plant facilities for the City of Red Deer from 1968 through 1973 resulted in a substantial improvement in winter dissolved oxygen levels below Red Deer. Further improvements were anticipated with winter flow augmentation post-construction of the Dickson Dam.

The extent of dissolved oxygen depletion in the bottom waters of Gleniffer Lake in the summer months is minimal. Winter levels are unknown. Post-impoundment oxygen levels measured 0.4 km downstream from the dam are close to saturation at all times. Average post-impoundment January and February oxygen levels below Red Deer and at Drumheller have increased (from 7.1 to 11.0 mg L⁻¹ at Red Deer, and from 3.7 to 9.4 mg L⁻¹ at Drumheller). Oxygen levels less than 5 mg L⁻¹ still occasionally occur in winter further downstream at the Bindloss Site.

Red Deer River Basin - Physical System¹

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ABSTRACT

The Red Deer River basin is located in central Alberta. Originating in the Rocky Mountains and foothills on the west, the Red Deer River extends across the Alberta plains and drains an area of about 47,000 km² before entering into the province of Saskatchewan to the east.

Flows in the Red Deer River are highly variable. Plains area snowmelt contributed most of the runoff in the March-April period. June and July are normally the highest runoff months in the year and are the results of mountain snowmelt combined with rainfall runoff from the foothill zone. Flows in the winter months are extremely low and generally come from groundwater storage.

Located on the Red Deer River some 40 km southwest of the City of Red Deer, is the Dickson Dam which was completed in 1983. The dam, which has a height of 40 m and a crest length of 650 m, controls runoff from a 5,200 km² area in the mountain and foothill region. The reservoir (Gleniffer Lake) formed by the dam, has a live storage capacity of approximately 203,000 dam³.

The benefits derived from the Dickson Dam project include:

- (a) an assured water supply and improved water quality;
- (b) flood and erosion control;
- (c) recreation; and
- (d) the potential for hydroelectric power generation.

¹Note: This paper was presented at the 1988 Applied Aquatic Studies Workshop.

RED DEER RIVER BASIN - PHYSICAL SYSTEMS

The Red Deer River Basin is located in the south-central part of the Province of Alberta and is the largest river basin in the South Saskatchewan River System.

Originating in the Sawridge Range of the Canadian Rockies, the headwaters of the Red Deer River are fed by meltwaters from the Drummond, Bonnet and St. Bride Glaciers. From its source, the river flows in an easterly direction through four physiographic zones (the mountain, foothills, western prairie and eastern prairie) and drains an area of about 47,000 km² (Fig. 1) before crossing the Alberta-Saskatchewan boundary and joining the South Saskatchewan River.

Figure 1. Red Deer River Basin.

The mountain zone covers an area of about 2,300 km² or 4.9% of the Red Deer River Basin. Topographic relief in the zone ranges from approximately 1,500 to over 3,000 meters above sea level. The zone has a mean annual precipitation in excess of 600 mm, mean annual lake evaporation of approximately 600 mm and a mean annual runoff in excess of 300 mm. Within this zone, the Red Deer River flows in a steeply graded channel set in deep valleys between mountain ridges.

The foothills zone covers an area of approximately 4,000 km² or 8.5% of the Red Deer River Basin. Topographic relief in the zone varies from approximately 1,000 to 1,800 m. The zone has a mean annual precipitation of approximately 550 mm, mean annual lake evaporation of approximately 650 mm and a mean annual runoff of approximately 200 mm. Within this zone, the Red Deer River is typified by frequent braided channels flowing through glacial drift which exhibits large gravel deposits.

The western prairie zone covers an area of approximately 19,000 km² or 40.4% of the Red Deer River Basin. Topographic relief in the zone varies from 850 to 1,000 m. The zone has a mean annual precipitation of approximately 450 mm and a mean annual lake evaporation of 675 mm. Mean annual runoff for the zone is approximately 60 mm but varies from 15 mm for areas adjacent to the eastern prairie to approximately 100 mm for areas adjacent to the foothills. Within this zone the Red Deer River is generally typified by a wide river valley containing a partly entrenched river channel.

The eastern prairie zone covers an area of approximately 21,700 km² or 46.2% of the Red Deer River Basin. Topographic relief in the zone varies from 500 to 850 m with a large number of closed drainage areas. Mean annual precipitation in the zone is approximately 350 mm with isolated pockets as low as 200 mm. Mean annual lake evaporation for the zone is approximately 725 mm while average runoff varies from less than 5 mm to approximately 20 mm. Within this zone, the Red Deer River is typified by gradually enlarged valley flats set amongst semi-arid plains and sinuous channels with occasional islands and sand bars.

Fig. 2 shows the mean annual flow hydrograph for three locations on the Red Deer River. The three locations are: (1) immediately upstream of Dickson Dam so as to represent the mountain and part of the foothill zones; (2) at the City of Red Deer near the eastern edge of the western prairie zone so as to depict the incremental input by the western prairie zone; and (3) at Bindloss near the Saskatchewan boundary so as to depict the incremental input by the eastern

prairie zone. The gauged mountain and foothills zone (lower curve of Fig. 2), while representing only about 8.8% of the basin drainage area to Bindloss, contributes nearly 50% of the gauged flow at Bindloss. Flow from this region commences in early April and peaks in mid-June with the majority of flow being contributed by high elevation snowmelt.

Figure 2. Flow hydrographs for Red Deer River.

The flow hydrograph for the Red Deer River at Red Deer (middle curve in Fig. 2) indicates that the additional $7,600 \text{ km}^2$ (17% of the drainage area at Bindloss) of foothills and western prairie areas between Dickson Dam and the City of Red Deer contributes an additional 32% of the gauged flow at Bindloss. The incremental increase in flow between the Dickson Dam and the City of Red Deer indicates two distinct peaks, one in April-May due to spring snowmelt

in the western prairie zone and a second peak in June-July due to rainfall runoff from the foothills zone.

The flood hydrograph for Bindloss (upper curve in Fig. 2) indicates that the additional 33,100 km² of drainage area (74% of the area at Bindloss) between the City of Red Deer and Bindloss contributed only 18% of the flow at Bindloss half of which is return flow from two irrigation districts which divert waters from the Bow River Basin to the south. The incremental increase in flow between the City of Red Deer and Bindloss indicates that nearly all flow contributions from the large area in the eastern prairie zone occurs during the spring snowmelt period.

DICKSON DAM

In 1980, construction began on the Dickson Dam (see Fig. 1 for location), the only hydraulic structure on the Red Deer River.

The dam was completed and first became operational in 1983.

The purpose and benefits derived from the dam include:

- (a) an assured water supply and improved water quality downstream;
- (b) flood and erosion control downstream;
- (c) recreation; and
- (d) the potential for hydroelectric power generation.

An assured water supply not only includes downstream municipalities and other users within the province, but also includes the ability of the province to meet its requirements under the "Prairie Province Water Board Apportionment Agreement".

The dam is a multi-zone earthfill structure 40 m high and 650 m long. With 3.6 million m³ of fill, it is one of the largest dams of its type in Alberta.

With the exception of Dogpound Creek and the Little Red Deer River, runoff from all of the mountain zone and the majority of the foothills zone, an area of 5,200 km², is controlled by the Dickson Dam. The controlled area, while representing only 10% of the total drainage area of the Red Deer River Basin, generates approximately 50% of the basin's yield.

The body of water formed behind the dam is named Gleniffer Lake and at Full Supply Level (FSL) is approximately 11 km long and 2 km wide, covering 1,750 ha and providing approximately 203 million m³ or 203,000 dam³ of usable storage.

The main components of the dam are shown in Fig. 3.

Figure 3. Dickson Dam inspection report.

Twin, reinforced concrete, low-level diversion tunnels, each approximately 5.5 m in diameter and 0.5 km long are able to provide a base flow to the Red Deer River and are equipped for the installation of hydroelectric power generating facilities, if required. Each tunnel has a design discharge capacity of $42 \text{ m}^3 \text{ s}^{-1}$.

The reinforced concrete service spillway is designed to pass the one in 10,000 year flood ($2,600 \text{ m}^3 \text{ s}^{-1}$). The estimated Probable Maximum Flood (PMF) peak of $5,300 \text{ m}^3 \text{ s}^{-1}$ could be safely handled by a combination of the service spillway and an earthen emergency spillway.

Both low level tunnels and the service spillway release highly oxygenated water to the Red Deer River.

Dam Operation

The service spillway and low level tunnel gates may be operated either manually (by staff at the dam site) or automatically.

Since the primary objective of the Dickson Dam is for low flow augmentation, and operational "rule curve" (or operations guidelines curve) was developed to ensure that a minimum base flow can be achieved most of the time.

The current base flow adopted for dam operations is $16 \text{ m}^3 \text{ s}^{-1}$ which, based on the basins hydrological statistics, can be provided 97% of the time.

The operational rule curve sets the limits of the desirable and permissible operating extremes of the reservoir's drawdown and fill cycles which will ensure water replenishment for the following operating season.

Normally, during times of high inflows, the reservoir is maintained at 2 m below FSL to assist with inflow attenuation and discharge regulation.

In general, releases to the Red Deer River will match the inflow conditions to the reservoir and may be achieved by either the low level tunnels or the service spillway.

Simultaneous operation of the service spillway gates and low level tunnel gates is not normally undertaken. For all practical purposes, the discharge capacity of the tunnels has no significant effect upon the dam's overall discharge when compared to the discharge capacity of the service spillway.

As can be seen in Fig. 4, the dam's operation to date has remained comfortably within the desirable segment of the rule curve.

Reduction of the reservoir's water level below the "desirable" curve (fill and drawdown) would result in the release rate from the reservoir being restricted to a maximum of $16 \text{ m}^3 \text{ s}^{-1}$. Further reduction of the reservoir's water level below the permissible drawdown curve will seriously jeopardize the reservoir's ability to recover and ensure water replenishment for the following operational season.

In a normal operating year, releases of $16 \text{ m}^3 \text{ s}^{-1}$ are made from the reservoir via the low level tunnels during the months of October to April (drawdown period). This essentially entails the discharge of water at a mean temperature of approximately 4°C to the Red Deer River.

Figure 4. Dickson Dam - Gleniffer Lake fill and depletion rule curves.

Observed effects of the winter releases on ice conditions in the Red Deer River include open water for a distance from 11 km to 20 km downstream of the dam. A less obvious, but nevertheless real effect of winter releases is the high oxygenation of the river water caused by air drawn down the air intake shafts of the low level tunnels as a result of the pressure reduction caused by the hydraulic jump within the tunnels.

During the May to September (fill) period the reservoir is subjected to mountain snowmelt and summer rainstorms.

During those times of high inflow, reservoir releases are made via the service spillway. This water is released from the upper portion of the reservoir and is generally at a slightly higher temperature than the receiving river water. Conversely, water released via the low level tunnels during the same period of time is from the lower portion of the reservoir and is generally at a slightly lower temperature than the receiving river water.

An overview of the 1986 operational sequence of low level tunnel and service spillway releases from the Dickson Dam is shown in Fig. 5. During the normal drawdown period, releases (base flow of $16 \text{ m}^3 \text{ s}^{-1}$) were made via the low level tunnels. In mid-May 1986, a tunnel power outage was experienced and released continued via the service spillway. (The increase in discharge rate was due to the lack of operational sensitivity available to the service spillway gates at low flows.)

Figure 5. Dickson Dam 1986 operations.

During mid-May, spring runoff occurred and reservoir pre-spill was required via the service spillway. At the end of June, releases were restored via the low level tunnels such that inflows matched outflows from the reservoir, until, at the beginning of July, severe summer rainstorms occurred and pre-spilling via the service spillway recommenced.

The August-September period (end of fill period) saw inflows matched by outflows via the low level tunnels until, at the end of September unusually late heavy precipitation caused

outflows to be matched by inflows to the reservoir which in turn exceeded the tunnel capacity and required operation of the service spillway.

The October-November period required releases via the low level tunnels with reservoir inflows being matched by outflows until December, when inflows to the reservoir virtually ceased and a base release flow of $16 \text{ m}^3 \text{ s}^{-1}$ was maintained.

IMPACT OF DAM ON LOW FLOWS

As stated earlier, the Dickson Dam first became operational in 1983. Four of the five years (1983-85, and 1987) since the implementation of the dam have been years of below average runoff (see Fig. 6) with 1984 being the second driest single year in 76 years of systemic streamflow records and 1983 to 1985 being the three driest consecutive years on record. In spite of these adverse conditions, mean monthly flows downstream of the dam (for the low flow period of October through March), have been maintained at levels significantly above the long-term average, as indicated in Fig. 7. The same information is presented in the form of October to March flow durations in Fig. 8. As shown in Fig. 8, post-regulation flows for the October to March low flow period are substantially higher than pre-regulation flows for all durations. The October-March flow duration curve for 1985, the second driest year on record, is virtually the same as for the 1984-87 period. In contrast, the October-March curve for 1949-50, a low flow year comparable to 1984, never exceeded the $16 \text{ m}^3 \text{ s}^{-1}$ (570 cfs) post-regulation target flow.

IMPACT OF DAM ON FLOOD PEAKS

Major floods on the Red Deer River generally occur in the mid-May to end of June period and are the result of one or more cold low pressure system lows, moving across the continental divide and drawing warm moist air from the Gulf of Mexico. The warm moist air becomes involved in quite energetic counter-clockwise circulation around the lows and is forced upslope along the foothills and the eastern face of the Rocky Mountains. The rapid uplift of the warm moist air, by both the cold low and the mountain slopes, results in large quantities of rainfall and runoff in the mountain and foothill zones.

As stated earlier, the Dickson Dam is situated such that it intercepts runoff from all of the mountain zone and the majority of the foothills zone. Since major floods in the Red Deer

Figure 6. Frequency curve of annual discharge for Red Deer River at Red Deer.

Figure 7. Flow of the Red Deer River at Red Deer.

Figure 8. Flow of the Red Deer River at Red Deer.

River are the result of large quantities of rainfall and associated runoff during the mid-May and June period in these two zones, the dam has great potential for runoff interception and attenuation of major floods in the Red Deer River. In order to optimize the flood reduction potential of the Dickson Dam, the Gleniffer Lake Reservoir is intentionally maintained at a relatively low level during the mid-May to end of June period (see Fig. 4). With the present ability of 36 hour lead time in forecasting flood peaks, the reservoir can be further drawn down prior to the occurrence of major floods and thus provide a greater degree of peak flow attenuation. The process of flood attenuation is shown in Fig. 9.

The degree of peak flow attenuation which can be achieved by the Dickson Dam has been assessed by Alberta Environment. The results are presented in Table 1.

Figure 9. Flood routing for the Dickson Reservoir 1 in 100 year flood hydrograph.

Table 1. Red Deer River flood frequencies.

Return period (years)	Instantaneous Discharge (cfs)					
	Dickson Damsite		Red Deer		Drumheller	
	Natural	Regulated	Natural	Regulated	Natural	Regulated
200	59,200	39,700	76,000	50,000	80,000	58,300
100	52,000	34,900	67,200	44,200	71,800	52,000
50	44,900	27,100	58,200	35,400	62,800	42,900
20	37,600	22,300	49,100	32,000	52,500	41,000
10	28,100	22,600	37,000	30,000	41,000	37,100

SUMMARY

Draining an area of approximately 47,000 km², the Red Deer River basin has a mean annual flow volume of 1.9×10^6 dam³. Almost 50% of the Red Deer River's mean annual flow is generated in the mountain and foothill zones which represent 10% of the total drainage basin.

Located downstream of the foothills zone, the Dickson Dam was completed in 1983 and has a live storage capacity of 203,000 dam³.

Runoff in almost all years since the construction of the dam has been below average. In spite of these adverse conditions, dam operation has enable a minimum base flow of $16 \text{ m}^3 \cdot \text{s}^{-1}$ (nearly twice the long-term October-March average) to be met at all times.

To date, the full flood attenuation potential of the Dickson Dam has not had to be utilized. However, in-house studies indicated that substantial flood attenuation may be realized.

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